Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them

Envisioning A Chemical-Safe World

Chapter 1 of 12

Global drivers, actors and policies affecting pesticides and fertilizer use

Contents

About ii

1 Global drivers, actors and policies affecting pesticides and fertilizer use 1

Key drivers affecting demand for pesticides and fertilizer use 1

Actors in the value chain affecting pesticides and fertilizer use 3

The international sustainable development context and policy instruments 4

References 8

2 Status and trends of pesticide use

3 The regulatory and policy environment for pesticide management

4 Environmental and health effects of pesticide use

5. The environmental, human health and economic impacts of pesticides

6 Current pesticide risk reduction and risk management

7 Status and trends of fertilizer use

8 The regulatory and policy environment of fertilizer management and use

9 Environmental and health effects of fertilizer use

10 The impact of fertilizer use

11 Current fertilizer risk reduction and risk management

12 Transformative actions to minimize the adverse impacts of pesticide and fertilizers

List of Figures

Figure 1.1	Certified share of cultivated land for selected crops.	3
Figure 1.2	The agriculture and food value chain.	4
Figure 1.3	Concentration of actors in the food production chain.	5
Figure 1.4	Linkages between chemicals and waste and the Sustainable Development Goals.	6

About

In December 2017, Resolution 4 of the 3rd Session of the United Nations Environment Assembly (UNEA 3) requested "the Executive Director to present a report on the environmental and health impacts of pesticides and fertilizers and ways of minimizing them, given the lack of data in that regard, in collaboration with the World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO) and other relevant organizations by the fifth session of the United Nations Environment Assembly". In response to this request, UNEP published a *Synthesis Report on the Environmental and Health Impacts of Pesticides and Fertilizers and Ways to Minimize Them*[1] in February 2022 (United Nations Environment Programme [UNEP] 2022).

The overall goal of the synthesis report is to provide the information base to enable other advocacy actions to be taken by stakeholders to minimize the adverse impacts of pesticides and fertilizers. Specific objectives of the synthesis report are to:

- Update understanding of current pesticide and fertilizer use practices;
- Present major environmental and health effects of pesticides and fertilizers, during their life cycle, and identify key knowledge gaps;
- Review current management practices, legislation and policies aimed at reducing risks in the context of the global chemicals, environmental and health agenda;
- Identify opportunities to minimize environmental and health impacts, including proven and innovative approaches.

This chapter on "Global drivers, actors and policies affecting pesticides and fertilizer use" is the 1st in a series of 12 chapters that make up a comprehensive compilation of scientific information. The chapters were developed to both inform and further elaborate on the information provided in the synthesis report. Please note that the disclaimers and copyright from the synthesis report apply

1 The Synthesis report is available at https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

1 Global drivers, actors and policies affecting pesticides and fertilizer use

Key drivers affecting demand for pesticides and fertilizer use

The world population is growing and dietary habits are changing

The world population is expected to grow from about 7.8 billion in 2020 to 9.8 billion by 2050. Global income is increasing, and the global middle class is expanding.

Demand for food, feed, fibres, fuels and feedstocks is growing

Agricultural crops provide food, feed, fibres, biofuels, feedstocks and other products to support the needs of a growing human population. By 2050 demand for food is projected to grow by 60 per cent, meat production by nearly 70 per cent, aquaculture by 90 per cent and dairy by 55 per cent (IFIF 2021).

Cropland is also increasingly used for other purposes than food production, necessitating greater intensification of production of food crops. For example:

- The global commercial feed manufacturing market has an estimated annual turnover of over USD 400 billion and is expected to grow annually by 3.4 per cent between 2020 and 2027 (Data Bridge Market Research 2020).
- World natural fibre production in 2018 was estimated at 32 million tons,[1] with the size of the global fibres market expected to register high growth rates at least until 2025 (Grandview Research 2021).
- The global market value of biofuels worldwide is expected to grow from USD 136 billion in 2019 to USD 153 billion in 2024 (Statista 2021).

Pesticides and fertilizers provide important functions, and demand for them continues to grow

Pesticides are substances, or mixtures of substances with chemical or biological ingredients, intended to repel, destroy or control any pest or to regulate plant growth. Pesticides are among several tools available to manage pests for agricultural and public health purposes, but also in industrial and domestic settings. They have made an important contribution to the growth of agricultural productivity and food supply. Pesticides are sold and used because of the benefits they are expected to provide. These benefits include reduced crop losses, reduced prevalence of human vector borne diseases, improved shelf life of agricultural commodities, increased livestock yields,

1 In the present report ton indicates a metric ton.

reduced soil disturbance, and better protection of wooden structures, among others.

Fertilizers provide nutrients that are essential to plants (i.e., nitrogen, phosphorus and potassium), maintain soil fertility, and support productive and nutritious crops. They help increase crop yields and therefore contribute to agricultural production (food/feed/fibres). In some cases fertilizers are used to improve the quality of food or feed, so that, for example, they are more nutritious for people and livestock. Without fertilizers there would be a need for more cropland than is currently the case. Through their use, it has been possible to support more people while saving forests and grasslands by reducing their conversion to cropland (Lassaletta *et al*. 2014; Ritchie 2021).

Global demand, production and use of pesticides and fertilizers have expanded steadily in the past decades. By 2016 about 4.1 million tons of pesticide active ingredients were used globally, twice the volume applied in 1990; about 190 million tons of nutrients from inorganic fertilizers were used in agriculture in 2018 (FAO 2020a). Demand for inorganic fertilizers is expected to reach 197 million tons of nutrients by 2024 (International Fertilizer Association 2020). The global pesticide market has increased steadily during the last decade, with a compound annual growth rate of sales value of 4.8 per cent (in both nominal and real values) between 2006 and 2016 (Agrow 2016; Phillips McDougal 2017; Agrow 2019; Agrow 2020). Without fundamental changes in pest and nutrient management, average annual growth in pesticide and fertilizer use is expected to continue in a similar manner. Combined global sales of pesticides and fertilizers continue to grow at about 4.1 per cent per year, and are projected to reach USD 309 billion by 2025 (Grand View Research 2017).

Intensification of the agri-food production system is continuing

Increased demand for crops has led to intensification of agricultural production, which is partly achieved by greater use of pesticides and fertilizers. For example, between 2002 and 2018 the global population increased by about 21 per cent (FAO 2020b) and cereal production by about 44 per cent (FAO 2020c), while pesticide use per hectare of cropland increased by about 30 per cent (FAO 2019). and inorganic fertilizer use per hectare by about 23 per cent for nitrogen, 13 per cent for phosphorus and 56 per cent for potassium (FAO 2020d) (Chapter 2 and Chapter 7). Similarly, livestock intensification has been achieved through the use of concentrated feeds, pharmaceuticals and vaccines (FAO 2012; Van Boeckel *et al*. 2015).

Trade in agricultural commodities is changing

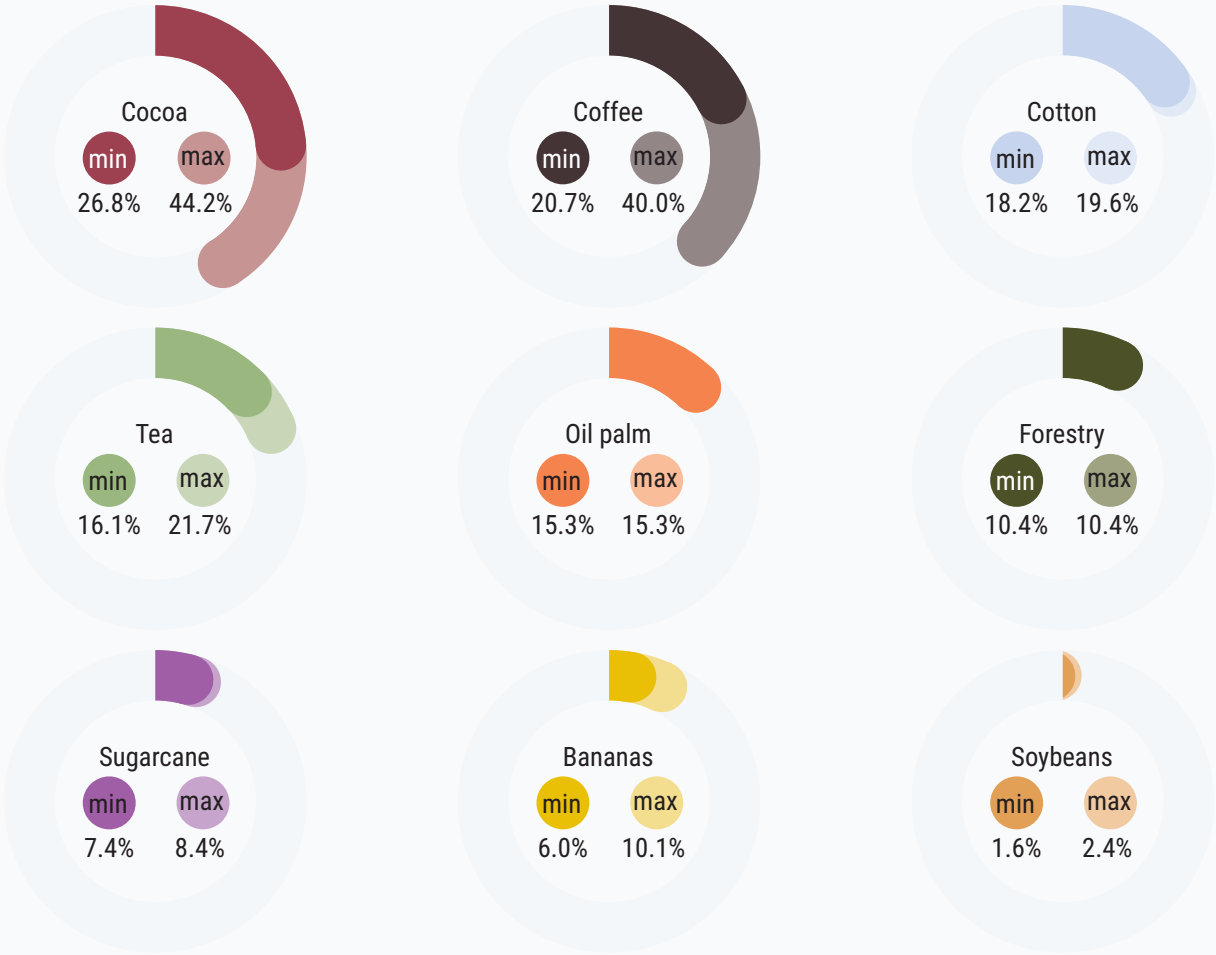
Increasing demand for agricultural products, including from middle income countries, is shaping trade patterns for crops and crop-based goods. This demand therefore affects pesticide and fertilizer use in exporting countries. Since the beginning of the millennium, trade in agricultural products has more than tripled to reach USD 1.33 trillion (Bellman, Lee and Hepburn 2019).

In addition to increases in scale, the geography of global food trade flows is shifting towards South-South trade, which currently accounts for roughly one-quarter of total agricultural trade flows. Most current growth in trade can be attributed to a few countries, including Brazil, China, India and Turkey. By 2030 Brazil is expected to surpass both the European Union and the United States as an agricultural exporter. China has become one of the largest importers of agricultural products, together with India, Mexico and Nigeria. In the future the largest demand is projected to come from Asia, followed by Africa (Bellmann, Lee and Hepburn 2019).

Sustainable agriculture is increasing, but it still covers a small share of cultivated land

While the use of voluntary sustainability standards in the agriculture sector continues to grow, it is estimated that only about 1 per cent of total agricultural cropland has been certified between 2000 and 2011 (Tayleur *et al*. 2017). Crops that tend to be certified at a relatively high rate include major commodities such as coffee, cocoa, tea and oil palm. Staple foods like maize (corn), rice and wheat are rarely certified (Figure 1.1). (Tayleur *et al*. 2017).

Figure 1.1 Certified share of cultivated land for selected crops. Meier *et al.* (2020), p. xiv.



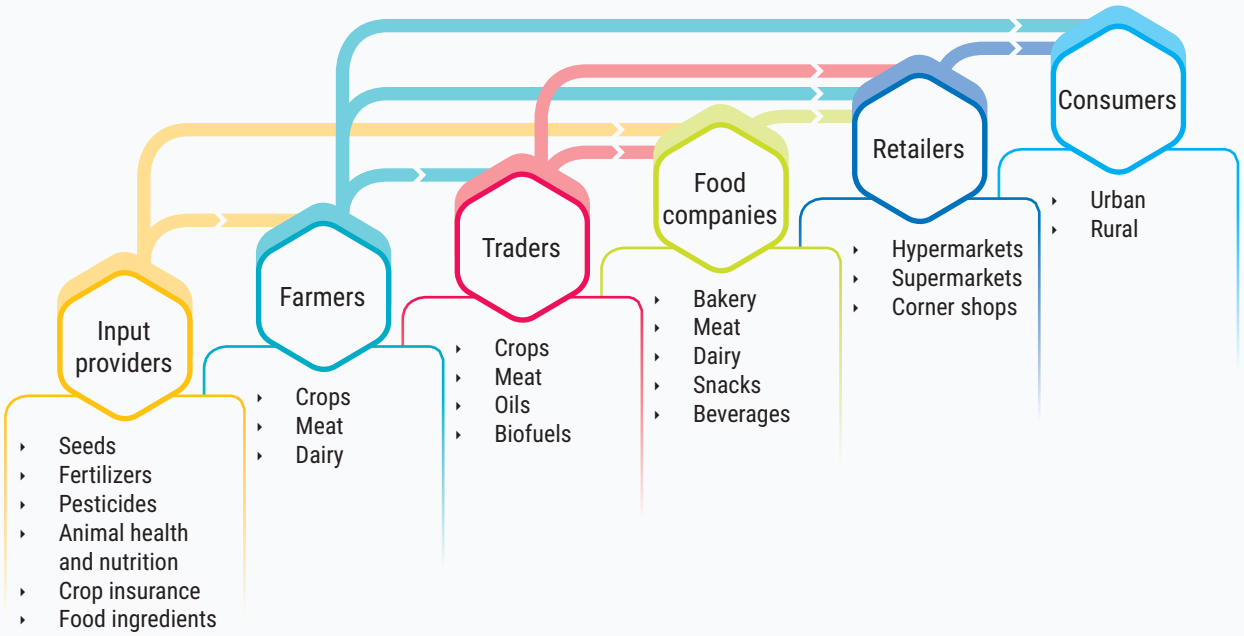
Actors in the value chain affecting pesticides and fertilizer use

Diverse actors affect the demand for and impacts of pesticides and fertilizers

The production and use of pesticides and fertilizers are shaped by diverse actors in the value chain of the global agri-food system and other, non-agricultural value chains. Key actors include consumers, farmers, the pesticide and fertilizer industries, and other corporate actors such as retailers, food companies and textile companies. Figure 1.2 provides an overview of the value chain from input companies (e.g., the pesticide and fertilizer industry) to the consumer.

Given their purchasing power, downstream actors can exert market pressure to foster the sustainable use and management of pesticides and fertilizers. For example, textile companies may require their suppliers to source certified organic cotton and support cotton producers shifting towards sustainable cotton production. While food companies play a particularly important role in driving change, they do not appear to be rising to the sustainability challenge. Only one global food company was among the top 100 most sustainably managed companies identified in a recent *Wall Street Journal* survey which evaluated 5,500 publicly traded companies based on environmental, social and governance (ESG) metrics (Cardello 2019).

Figure 1.2 The agriculture and food value chain.



The global agri-food market is controlled by a small number of corporations

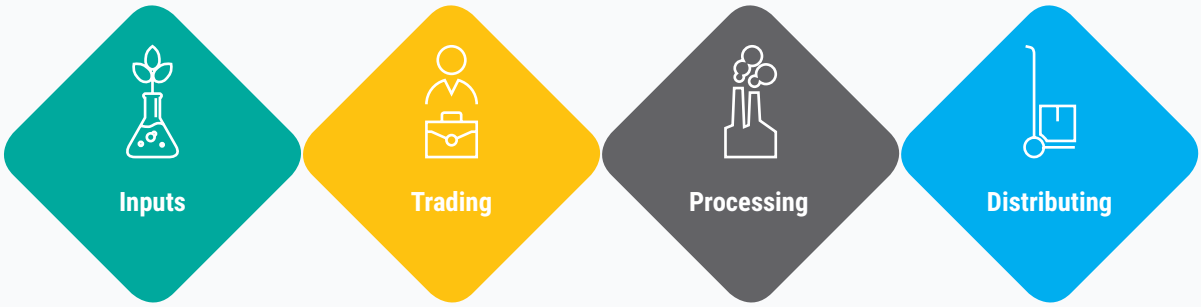
A small number of corporations control important segments of the global agri-food market (Figure 1.3). Three companies control over half the seed market, four traders control over 90 per cent of global grain trade, 10 retailers share 40 per cent of global food sales, and four (conglomerates of) companies currently represent about 65 per cent of the global pesticide market (Bellmann, Lee and Hepburn 2019). The same companies also often have important activities in the areas of seeds and genetically modified crops, leading to a concentration of research, development and marketing capacities with regard to agricultural inputs. At the same time, the share of off-patent ("generic") pesticides has increased from about 40 per cent in the early 2000s to about 70 per cent of the current global pesticide market (AgbioInvestor 2019).

Large and small farms face challenges that require distinct solutions

Of the 570 million farms in the world, 84 per cent are smaller than 2 hectares. In most low and lower-middle income countries farm sizes have decreased during the last 40 years, while they have increased in high income countries (Lowder, Skoet and Raney 2016). About 1 per cent of farms operate 70 per cent of the world's farmland, creating concerns that inequality is growing because agricultural production is increasingly dominated by a few major companies (International Land Coalition 2020). Access to agrochemicals (including pesticides and fertilizers), as well as knowledge about their proper use, are determined to a large extent by income, geographic location, and policies, as is access to relevant technologies.

The ways pesticides and fertilizers are used, and therefore affect the environment and health, greatly differ between smallholder and industrialized agriculture. Due to continued divergence in farm sizes and wealth status (which is particularly prevalent in low and lower-middle income countries), risk reduction measures need to take into account the specific situations and conditions of smallholder and industrial scale farming.

Figure 1.3 Concentration of actors in the food production chain. Bellmann, Lee and Hepburn (2019).



56%

Seeds

3 companies control 56% of the market: Bayer/ Monsanto, Corteva Agriscience (DowDupont), ChemChina/ Syngenta.

60%

Pesticides

4 companies share 60% of global pesticides market: ChemChina/Syngenta, Bayer/Monsanto, BASF, Corteva Agriscience (DowDupont).

90%

Traders

4 major traders control as much as 90% of the global grain trade: Bunge, Archer Daniels Midland, Cargill, Louis Dreyfus.

$1.1 billion

Food and beverages

10 companies own the biggest product brands and manufacture most of consumer shopping carts, collectively generating revenues of more than $1.1 billion a day: Associated British Foods, Coca-Cola, Danone, General Mills, Kellogg's, Mars, Mondelez International, Nestlé, PepsiCo, Unilever.

40%

Retailers

10 retailers share 40% in worldwide sales. In most OECD economies, one retailer controls between 15-30% of national supermarket sales: Tesco (UK), E.Leclerc (France), WalMart (USA), Edeka (Germany).

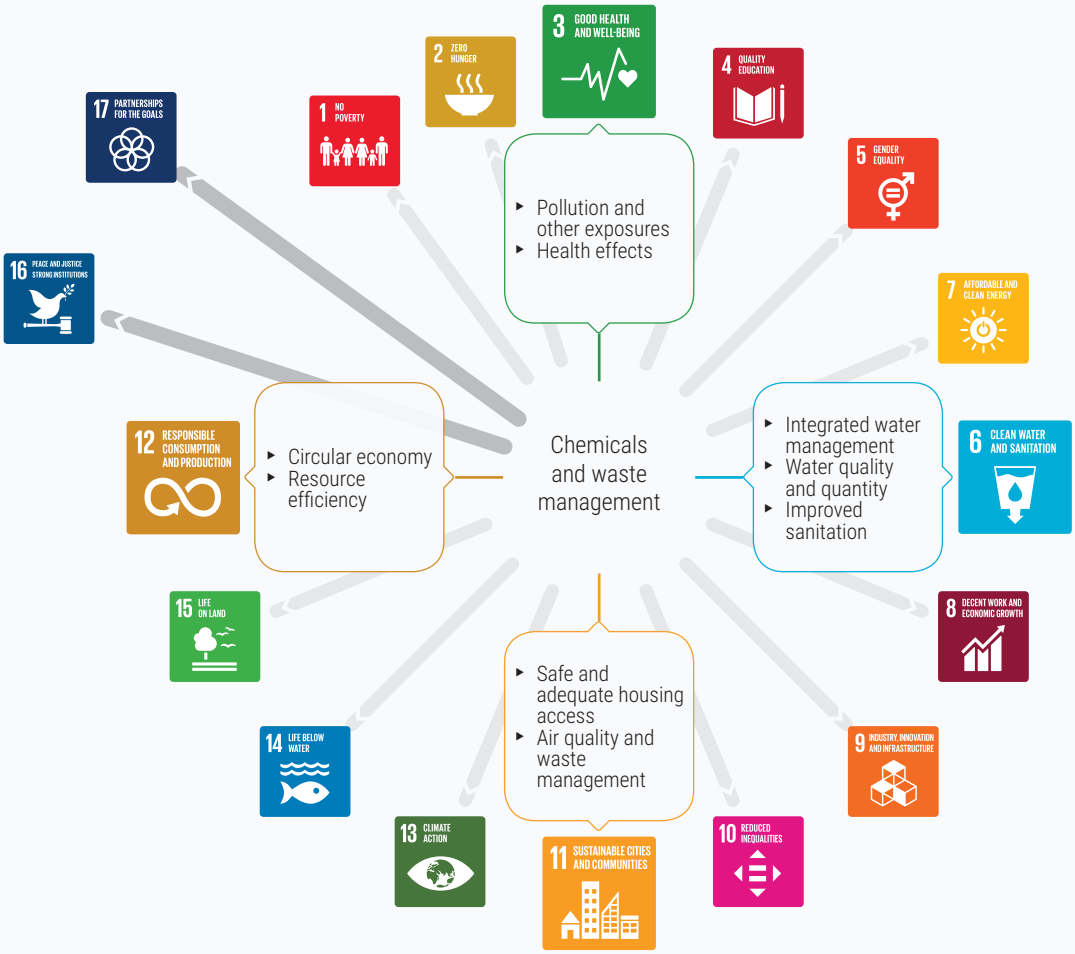
The international sustainable development context and policy instruments

The 2030 Agenda for Sustainable Development: Linkages with pesticides and fertilizers

The 2030 Sustainable Development Agenda was adopted in 2015 as a universal global framework to encourage action by all stakeholders to achieve sustainable development by the year 2030. The 2030 Agenda emphasizes that development needs to be compatible with all three dimensions of sustainability: economic, social and environmental. Sustainable development is therefore integrated, indivisible and needs to be implemented as a whole (rather than in a fragmented manner). Figure 1.4 shows linkages between chemicals and waste and the 17 Sustainable Development Goals (SDGs). Several of the SDGs are directly relevant to the sound management of pesticides and fertilizers. For example, SDG 12 on Sustainable Consumption and Production, Target 12.4, calls for achieving the environmentally sound management of chemicals and all wastes throughout their life cycle by 2020. This goal has not been achieved, according to UNEP's *Global Chemicals Outlook* (GCO-II) (United Nations Environment Programme 2019). In addition, SDG 3 on Good Health and Well-being, Target 3.9, calls for reducing the number of deaths and illnesses from hazardous chemicals by 2030.

SDGs and targets for which the sound management of pesticides and fertilizers is essential include access to food, clean energy (e.g., use of biofuels), health, and safe housing (e.g., use of bio-based building materials and household pesticides). Some SDGs and targets

Figure 1.4 Linkages between chemicals and waste and the Sustainable Development Goals. Based on Inter-Organization Programme for the Sound Management of Chemicals (IOMC) (2018), p. 3.



also help to strengthen an enabling environment for the sound management of pesticides and fertilizers, for example those concerned with access to information, education and financing.

Given its global importance, implementing the 2030 Sustainable Development Agenda presents an opportunity for collaborative action among many diverse actors and at all levels to minimize the adverse impacts of pesticides and fertilizers.

Global instruments and initiatives to manage pesticides and fertilizers

Over many years the international community has put in place a range of policy instruments and initiatives to manage pesticides and fertilizers Insights from the report and the proposed options for action may also provide an opportunity to strengthen relevant international instruments, as appropriate.

Two international legally binding instruments address a limited number of pesticides. The Stockholm Convention on Persistent Organic Pollutants regulates the entire life cycle of the substances listed under it. The Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade includes provisions for using the prior informed consent procedure in regard to the import of the substances listed under it and for facilitating exchange of information on them. In addition, the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposal facilitates sound management and disposal of hazardous wastes including waste pesticides and fertilizers.

Table 1.1 Examples of linkages between pesticides and fertilizers and global conventions and policy frameworks.

Cluster	Global conventions and policy frameworks	Linkages with pesticides and fertilizers
Climate change	United Nations Framework Convention on Climate Change	• Fertilizers contribute to global greenhouse gas emissions • Certain (Intergovernmental Panel on Climate Change) recommendations target fertilizers
Biodiversity	Convention on Biological Diversity	• The Aichi Biodiversity Targets cover excess nutrients, sustainable agricultural production, and elimination of incentives (including subsidies) harmful to biodiversity
Desertification	United Nations Convention to Combat Desertification	• The convention links environment and development to sustainable land management (SLM) • Improving soil organic carbon is one of the SLM options
Food security and nutrition	Rome Declaration on Nutrition (FAO and WHO 2014b); United Nations Decade of Action on Nutrition, 2016-2025 (United Nations 2020)	• Eradicate hunger and prevent all forms of malnutrition • Enhance sustainable food systems • Legislative frameworks for food safety and quality, including for the proper use of agrochemicals

Key voluntary global instruments include the International Code of Conduct on Pesticide Management (FAO and WHO 2014a), which provides guiding principles for sound management of the entire life cycle of pesticides, and FAO's International Code of Conduct for the Sustainable Use and Management of Fertilizers (FAO 2019), which promotes the judicious use of fertilizers and addresses their misuse, overuse and underuse. SAICM focuses on some major global concerns about pesticides not specifically covered by the conventions mentioned above, such as Highly Hazardous Pesticides (Strategic Approach to International Chemicals Management 2021).

Other major global agreements and policies relevant for pesticides and fertilizers

Although not immediately obvious, the adverse impacts of pesticides and fertilizers may compromise the achievement of agreed global goals such as those addressing climate change, biodiversity loss and desertification (Table 1.1).

National and regional policy frameworks relevant to pesticides and fertilizers

Legislation, policies and collaborative platforms are crucial for effective pesticide and fertilizer use and management, at both the national and regional levels. National pesticide legislation and policies now exist in most countries in the world. Fertilizer legislation is less well developed, particularly in low and middle income countries. Increasingly, regional collaboration among countries supports sound management of pesticides, while regional fertilizer use and nutrient management appear to be driven more by non-governmental stakeholders such as scientific bodies.

References

AgbioInvestor (2019). *Trends in Crop Protection and Seed Market Development. Presentation by Dr Matthew Phillips at CropLife America Convention, Antigua, Guatemala, 9 April 2019. AgbioInvestor – Agricultural Market Intelligence, Midlothian, Scotland.* https://agbioinvestor.com. Accessed 7 November 2020.

Bellmann, C., Lee, B. and Hepburn, J. (2019). *Delivering Sustainable Food and Land Use Systems: The Role of International Trade. London: Hoffmann Centre for Sustainable Resource Economy.* https://www.chathamhouse.org/sites/default/files/2019-10-14-HoffmanCentreTradeandFoodSystems.pdf.

Cardello, F. (2019). Food industry is a no-show in new sustainability study, 20 October. *Forbes*. https://www.forbes.com/sites/hankcardello/2020/10/20/food-industry-is-a-no-show-in-new-sustainability-study/#7700fedc1c57. Accessed 6 February 2021.

Data Bridge Market Research (2020). Animal feed market size, share outlook by 2027: Competitors' new strategies, growth, opportunities and techniques development analysis with current and future industry figures. https://www.globenewswire.com/fr/news-release/2020/09/15/2094056/0/en/Animal-Feed-Market-Size-Share-Outlook-by-2027-Competitors-New-Strategies-Growth-Opportunities-and-Techniques-Development-Analysis-with-Current-and-Future-Industry-Figures.html. Accessed 6 February 2021.

Food and Agriculture Organization of the United Nations (2012). *Livestock and Landscapes.* https://www.fao.org/3/ar591e/ar591e.pdf.

Food and Agriculture Organization of the United Nations (2019). *The international Code of Conduct for the Sustainable Use and Management of Fertilizers.* https://fao.org/3/ca5253en/CA5253EN.pdf.

Food and Agriculture Organization of the United Nations (2020a). FAOSTAT: Fertilizers by nutrient. https://www.fao.org/faostat/en/#data/RFN. Accessed 6 February 2021.

Food and Agriculture Organization of the United Nations (2020b). FAOSTAT. Annual population. https://www.fao.org/faostat/en/#data/OA. Accessed 24 October 2020

Food and Agriculture Organization of the United Nations (2020c). FAOSTAT. Crops. https://www.fao.org/faostat/en/#data/QC. Accessed 24 October 2020.

Food and Agriculture Organization of the United Nations (2020d). FAOSTAT. Fertilizers indicators. https://www.fao.org/faostat/en/#data/EF. Accessed 24 October 2020.

Food and Agriculture Organization of the United Nations (2021a). FAOSTAT. Data: Pesticides use. https://www.fao.org/faostat/en/#data/RP/visualize. Accessed 10 January 2022.

Food and Agriculture Organization of the United Nations and World Health Organization (2014a). *The International Code of Conduct on Pesticide Management.* https://www.fao.org/agriculture/crops/thematic-sitemap/theme/pests/code/en/#:~:text=The%20International%20Code%20of%20Conduct%20on%20Pesticide%20Management,approved%20by%20the%20FAO%20Conference%20in%20June%202013.

Food and Agriculture Organization of the United Nations and World Health Organization (2014b). *Rome Declaration on Nutrition. Second International Conference on Nutrition, Rome, 18-21 November 2014.* http://www.fao.org/3/a-ml542e.pdf.

Food and Agriculture Organization of the United Nations. (2017). *The Future of Food and Agriculture – Trends and Challenges.* http://www.fao.org/3/i6583e/i6583e.pdf.

Globe Newswire (2020). The global renewable chemicals market is projected to reach US$125.6 billion by 2025. https://www.globenewswire.com/news-release/2020/09/09/2091027/0/en/The-global-Renewable-Chemicals-market-is-projected-to-reach-US-125-6billion-by-2025.html. Accessed 6 February 2021.

Grandview Research (2017). Agrochemicals market worth $308.92 billion by 2025 | CAGR: 4.1%. https://www.grandviewresearch.com/press-release/global-agrochemicals-market. Accessed 6 February 2021.

Grandview Research (2021). *Natural Fibers Market Size, Share and Trends Analysis Report by Application, by Regional Outlook, by Competitive Strategies, and Segment Forecasts, 2019 to 2025.* https://www.grandviewresearch.com/industry-analysis/natural-fibers-market.

International Feed Industry Federation (2021). Global feed statistics. https://ifif.org/global-feed/statistics. Accessed 6 February 2021.

International Fertilizer Association (2020). *Fertilizer Outlook 2020-2024.* https://www.ifastat.org/market-outlooks. Accessed 6 February 2021.

International Land Coalition (2020). *Uneven Ground: Land Inequality at the Heart of Unequal Societies. Research Findings from the Land Inequality Initiative.* https://www.landcoalition.org/en/uneven-ground/.

Inter-Organization Programme for the Sound Management of Chemicals (IOMC) (2018). *Chemicals and Waste Management: Essential to Achieving the Sustainable Development Goals (SDGs).* http://www.who.int/iomc/ChemicalsandSDGs_interactive_Feb2018_new.pdf.

Lassaletta, L., Billen, G., Grizzetti, B., Anglade, J. and Garnier, J. (2014). 50 year trends in nitrogen use efficiency of world cropping systems: The relationship between yield and nitrogen input to cropland. *Environmental Research Letters 9(10), 105011.* https://iopscience.iop.org/article/10.1088/1748-9326/9/10/105011/meta.

Lowder, S.K., Skoet, J. and Raney, T. (2016). The number, size, and distribution of farms, smallholder farms, and family farms worldwide. *World Development 87, 16-29.* http://dx.doi.org/10.1016/j.worlddev.2015.10.041.

Market Research Future (2020). Renewable chemicals market global information. https://www.marketresearchfuture.com/reports/renewable-chemicals-market-1047. Accessed 6 February 2021.

Meier, C., Sampson, G., Larrea, C., Schlatter, B., Voora, V., Dang, D. *et al. (2020). The State of Sustainable Markets 2020: Statistics and Emerging Trends. Geneva: International Trade Centre (ITC).* https://www.intracen.org/publication/Sustainable-Markets-2020/.

Ritchie H. (2021). Can we reduce fertilizer use without sacrificing food production? 9 September. *Our World in Data.* https://ourworldindata.org/reducing-fertilizer-use. Accessed 7 December 2021.

Statista (2021). Market value of biofuels worldwide in 2019 and *2024. Published by N. Sönnichsen, 27 January.* https://www.statista.com/statistics/217179/global-biofuels-market-size/#:~:text=In%202016%2C%20the%20global%20market,higher%20prices%20for%20alternative%20fuels. Accessed 6 February 2021.

Strategic Approach to International Chemicals Management (SAICM) (2021). Highly Hazardous Pesticides (HHPs). https://www.SAICM.org/Implementation/EmergingPolicyIssues/HighlyHazardousPesticides/tabid/5479/Default.aspx. Accessed 6 February 2021.

Tayleur, C., Balmford, A., Buchanan, G.M., Butchart, S.H.M., Ducharme, H., Green; R.E. *et al. (2017). Global coverage of agricultural sustainability standards, and their role in conserving biodiversity. Conservation Letters 10(5), 610-618.* https://doi.org/10.1111/conl.12314.

United Nations (2020). United Nations Decade of Action on Malnutrition, 2016-2025. https://www.un.org/nutrition/. Accessed 6 February 2021.

United Nations Environment Programme (2019). *Global Chemicals Outlook II – From Legacies to Innovative Solutions.* https://www.unep.org/explore-topics/chemicals-waste/what-we-do/policy-and-governance/global-chemicals-outlook.

United Nations Environment Programme (2022). *Synthesis Report on Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them*. https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

Van Boeckel, T.P., Brower, C., Gilbert, M., Grenfell, B., Levin, S.A., Robinson, T.P. *et al. (2015). Global trends in antimicrobial use in food animals. Proceedings of the National Academy of Sciences of the United States of America (112)18, 5649-5654.* https://doi.org/10.1073/pnas.15n03141112.

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them

Envisioning A Chemical-Safe World

Chapter 2 of 12

Status and trends of pesticide use

Contents

1 Global drivers, actors and policies affecting pesticides and fertilizer use

About vi

2 Status and trends of pesticide use 1

2.1 Overview 1

2.2 Types of pesticides 2

2.2.1 Pesticide definitions 2
2.2.2 Pesticide categories 3
2.2.3 Components of a pesticide 4

2.3 History of pesticide development 7

2.4 Historical and current use of pesticides 10

2.4.1 Pesticide manufacturing 10
2.4.2 Global pesticide use trends – volume 10
2.4.3 Global pesticide use trends 13
2.4.4 Bioprotectants 19
2.4.5 Agricultural pesticide use intensity 20

2.5 Pesticide distribution mechanisms 23

2.5.1 Organization of the pesticide industry 23
2.5.2 Pesticide supply chains 24
2.5.3 Illegal pesticides 26

2.6 Pesticide application technology 29

2.6.1 Application techniques 29
2.6.2 Reducing environmental and occupational exposure 29
2.6.3 Improving the precision of pesticide applications 30
2.6.4 Pesticide formulation 30
2.6.5 Regional differences in pesticide application innovations 32

2.7 Drivers of pesticide use 32

2.7.1 Growth in agricultural production 34
2.7.2 Crop and post-harvest losses due to weeds, pests and diseases 34

2.7.3 Agricultural intensification 35
2.7.4 Genetically modified crops 37
2.7.5 Integrated pest and vector management 40
2.7.6 Organic production 45
2.7.7 Pesticide legislation and policies 46
2.7.8 Pesticide marketing 47
2.7.9 Pesticide prices 47
2.7.10 Commodity prices 48
2.7.11 Fiscal policies 48
2.7.12 Voluntary sustainability standards 50
2.7.13 Food safety 53
2.7.14 Public concerns about human health 53
2.7.15 Vector-borne diseases 54
2.7.16 Climate change 55
2.7.17 Invasive species and pest outbreaks 56
2.7.18 Public environmental concerns 58
2.7.19 Information sources 59
2.7.20 Knowledge, awareness and attitudes 60
2.7.21 Training 61

References 63

3 The regulatory and policy environment for pesticide management

4 Environmental and health effects of pesticide use

5. The environmental, human health and economic impacts of pesticides

6 Current pesticide risk reduction and risk management

7 Status and trends of fertilizer use

8 The regulatory and policy environment of fertilizer management and use

9 Environmental and health effects of fertilizer use

10 The impact of fertilizer use

11 Current fertilizer risk reduction and risk management

12 Transformative actions to minimize the adverse impacts of pesticide and fertilizers

List of Boxes

Box 2.5-1	Illegal pesticides can take different forms	26
Box 2.7-1	Key current drivers of pesticide use. The predominant ways in which these drivers tend to influence pesticide use are presented. However, it is recognized that such influences may be more subtle, as is discussed in the referenced chapters.	33
Box 2.7-2	Concepts of crop yield and crop loss.)	36
Box 2.7-3	Aspects of integrated pest management (IPM).	41
Box 2.7-4	Examples of fiscal instruments that may influence pesticide use.	49
Box 2.7-5	The desert locust upsurge, 2019-2020.	57

List of Figures

Figure 2.2-1	Categorization of pesticides according to the target organisms they are intended control	4
Figure 2.2-2	Bioprotectants covered in this report include semiochemicals, microbials and natural substances, but not invertebrate control agents. Some bioprotectants may be formulated and used similarly to chemical pesticides, in which case they are sometimes referred to as biopesticides.	6
Figure 2.3-1	Chronology and application rates of the main fungicide, insecticide and herbicide groups.	8
Figure 2.3-2	Number of new pesticide active ingredients introduced globally per decade.	9
Figure 2.3-3	Discovery and development costs (nominal values, i.e. not adjusted for inflation) of a crop protection product based on a new active ingredient increased almost 90 per cent between 1995 and 2014..	9
Figure 2.4-1	Global manufacturing of pesticides has increased steadily.	10
Figure 2.4-2	Global use of pesticides in agriculture increased from about 2.3 million tons of active ingredient in the early 1990s to more than 4 million tons in 2016.	11
Figure 2.4-3	Pesticide use in agriculture has exhibited different trends in the major regions of the world	11
Figure 2.4-4	Overall growth in agricultural pesticide use (volumes of active ingredient) between 1990 and 2016 was highest in Latin America and the Caribbean. It was relatively stable in northern America and Europe.	12
Figure 2.4-5	Share of the total volume of pesticides used by countries according to income group in 1991-1995 and 2012-2016.	12
Figure 2.4-6	Average increase in the volumes of pesticides used in countries according to income group between 1991-1995 and 2012-2016.	12
Figure 2.4-7	The nominal value of global pesticide sales has increased steadily since 2006.	13
Figure 2.4-8	Compound annual growth rates for the global pesticides market, based on nominal and real term values (adjusted for variations in USD exchange rates and inflation).	14
Figure 2.4-9	Regional markets for crop protection products and annual growth rates. Market value is expressed as USD million and percentage year-on-year change is from 2018 to 2019.	14
Figure 2.4-10	Major groups of crop protection pesticides as percentage of the total volume used globally.	15
Figure 2.4-11	A major shift has occurred in pesticide use in the United States. Insecticides accounted for 58 per cent of pesticide volume applied in 1960, but only 6 per cent in 2008. Herbicides accounted for 18 per cent of the volume applied in 1960, compared with 76 per cent by 2008.	15

Figure 2.4-12 Change in pesticide use on soybeans and cotton in the United States between 1968 and 2008 (based on total volume of active ingredients applied). 16

Figure 2.4-13 Pesticide use on different crop groups in 2018, based on global sales value. 17

Figure 2.4-14 Trend in global use of vector control insecticides. 18

Figure 2.4-15 Cumulative number of insecticide-treated nets delivered globally from 2004-2019 in sub-Saharan Africa and 2009-2019 in other countries. Almost 2.1 billion insecticide-treated nets were delivered in this period. 18

Figure 2.4-16 The global biopesticide market is a small but rapidly increasing part of the total market for crop protection products (nominal values: USD million per year). 19

Figure 2.4-17 The biopesticide market is dominated by microorganisms, followed by biochemicals such as pheromones and plant extracts. Sales of organic acids and yeasts are relatively small. 19

Figure 2.4-18 Global agricultural pesticide use per unit cropland increased by 75 per cent between 1990 and 2016. 20

Figure 2.4.19 Pesticide use intensity correlates positively with the level of a country's economic development.. 22

Figure 2.5-1 The pesticide industry collaborates in three major international associations representing R&D companies, producers of generic pesticides and biopesticide companies 24

Figure 2.5-2 Simplified pesticides supply chain, where pesticide products may originate from different sources. 25

Figure 2.5-3 Countering the illegal trade in pesticides should address all stages of the pesticide distribution chain to be effective. 28

Figure 2.7-1 Growth rates of total and per capita food consumption, 2018-2027. 34

Figure 2.7-2 Herbicide use in soybean (kg/ha) in the United States following the introduction in 1996 of GM herbicide tolerant soybean. Glyphosate has progressively replaced other herbicides. 39

Figure 2.7-3 IPM projects and programmes in Asia and Africa result in increased crop yields and reduced pesticide use. Shown are 85 projects in 24 countries. 44

Figure 2.7-4 Distribution of main organic land use types and crop categories in 2018. Globally, 71.5 million ha were under organic production in 2018. 45

Figure 2.7-5 The global area of cropland under VSS certification has been increasing rapidly. 51

Figure 2.7-6 Public health concerns in the news. Shown are the number of newspaper articles about food and pesticides as well as cancer and pesticides, published annually over the last 20 years. 54

Figure 2.7-7 Environmental concerns of European citizens. The percentage of respondents who considered "the use of pesticides, fertilizers, etc" an important environmental issue has slightly increased since the mid-2000s. 59

Figure 2.7-8 Environmental concerns in the news. Shown are the number of newspaper articles about the environment and pesticides, and honeybees and pesticides, published annually in the last 20 years. 59

List of Tables

Table 2.2-1	Specific groups of pesticides included in countries' pesticide definitions.	3
Table 2.2-2	Major classes of synthetic chemical insecticides, herbicides and fungicides and examples of well-known individual compounds.	5
Table 2.2-3	Examples of common biopesticides on the market.	6
Table 2.4-1	Average reported insecticide use for vector control according to method of application and class of insecticide by WHO region (2000–2009), in metric tons of active ingredient per year.	17
Table 2.5-1	Top 10 pesticide companies and their share of the total agricultural pesticide market in 2018.	23
Table 2.5-2	Recent reports on the scale of the trade in illegal pesticides.	27
Table 2.7-1	Estimates of global yield losses in major crops caused by pests, diseases and weeds.	35
Table 2.7-2	Global share of approvals for cultivation of genetically modified crops according to the main commercial GM traits. Approved events may be single traits or combinations of traits.	37
Table 2.7-3	Impact of growing genetically modified crops on volume of pesticide use.	38
Table 2.7-4	Integrated pest management (IPM) and integrated vector management (IVM) have become central policies for pest and vector management at national and international levels: some examples.	40
Table 2.7-5	Effects of integrated pest management (IPM) on pesticide use (global and regional reviews conducted since 2010).	42
Table 2.7-6	Examples of pesticides allowed in organic agriculture.	46
Table 2.7-7	Selected pest and pesticide management requirements of major voluntary sustainability standards.	52
Table 2.7-8	Some examples of recent serious invasive pests.	56

About

In December 2017, Resolution 4 of the 3rd Session of the United Nations Environment Assembly (UNEA 3) requested "the Executive Director to present a report on the environmental and health impacts of pesticides and fertilizers and ways of minimizing them, given the lack of data in that regard, in collaboration with the World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO) and other relevant organizations by the fifth session of the United Nations Environment Assembly". In response to this request, UNEP published a *Synthesis Report on the Environmental and Health Impacts of Pesticides and Fertilizers and Ways to Minimize Them*[1] in February 2022 (United Nations Environment Programme [UNEP] 2022).

The overall goal of the synthesis report is to provide the information base to enable other advocacy actions to be taken by stakeholders to minimize the adverse impacts of pesticides and fertilizers. Specific objectives of the synthesis report are to:

- Update understanding of current pesticide and fertilizer use practices;
- Present major environmental and health effects of pesticides and fertilizers, during their life cycle, and identify key knowledge gaps;
- Review current management practices, legislation and policies aimed at reducing risks in the context of the global chemicals, environmental and health agenda;
- Identify opportunities to minimize environmental and health impacts, including proven and innovative approaches.

This chapter on "Status and trends of pesticide use" is the 2nd in a series of 12 chapters that make up a comprehensive compilation of scientific information. The chapters were developed to both inform and further elaborate on the information provided in the synthesis report. Please note that the disclaimers and copyright from the synthesis report apply

1 The Synthesis report is available at https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

Status and trends of pesticide use

2.1 Overview

In this report the term pest designates any type of organism targeted by a pesticide. When specific groups of pesticides such as insecticides or herbicides are addressed, they are identified as such. Pesticides are used to control arthropod (invertebrate) pests such as insects and mites, as well as diseases caused by fungi and bacteria. They are also used to control weeds, molluscs, nematodes, rodents, and other organisms that may damage crops or trees, transmit human diseases, overgrow roads, damage buildings, or are otherwise considered a nuisance or danger (Matthews 2018; Philipps McDougal 2018a).

Pesticide use has steadily increased since the introduction of synthetic organic pesticides in the 1940s (Matthews 2018). By 2016 about 4.1 million tons of pesticide active ingredients per year were used globally, double the quantity applied in 1990 (FAOSTAT 2019). The total value of the pesticide market was estimated at about United States dollars (USD) 65 billion in 2018 (Agrow 2019). The most pesticides by volume are used in Asia and South and Central America, the regions that have shown the highest growth in use during the last 25 years (FAO 2019a). [Chapters 2.4.2 and 2.4.3]

The large majority of pesticides are used in agriculture. Non-agricultural uses, including domestic or industrial applications and vector control, represent only 10-15 per cent of the global market by value (Phillips McDougal2017); Agrow 2019; Agrow 2020). About 60 per cent of the volume of all agricultural pesticides applied consists of herbicides, with the other 40 per cent almost equally divided between fungicides and insecticides (Phillips McDougal 2018b). [Chapter 2.4.3] Biological pest control agents (or bioprotectants) represent about 7 per cent of the value of the total crop protection market (Agrow 2018). Although this is a small fraction of the total pesticide market, sales of bioprotectants are growing rapidly (at 15-20 per cent per year), considerably faster than sales of synthetic chemical pesticides (Glare *et al*. 2012; DunhamTrimmer 2019). [Chapter 2.4.4]

The intensity of global agricultural pesticide use, measured as kilogram (kg) of pesticide active ingredient applied per hectare (ha) of cropland, increased by about 75 per cent between 1990 and 2016. Agricultural productivity has also increased in the same period. As a result, the quantity of pesticide required per unit agricultural production has remained approximately unchanged. Nevertheless, the biological activity ("pest control power per unit product") of modern pesticides is significantly higher than that of the older groups of pesticides (FAO 2019a). Pesticide use intensity per ha cropland and per unit agricultural production is positively correlated with per capita gross domestic product: the richer the country, the more pesticides are used. A slight decrease in use intensity per hectare (but not in pesticide use intensity per unit

production) has been seen in the world's richest countries. High income countries therefore do not appear to use pesticides more efficiently than lower income countries. However, differences exist between crops and between regions (Schreinemachers and Tipraqsa 2012). [Chapter 2.4.5]

The pesticide industry has experienced important mergers and acquisitions during the last decade. Currently four (conglomerates of) companies represent about 60 per cent of the global agricultural pesticide market. The same companies often also have important activities involving seeds and biotech crops, leading to a concentration of research, development and marketing capacities with regard to agricultural inputs (Yuan 2019). At the same time, the share of off-patent ("generic") pesticides has increased from about 40 per cent of the global value of the pesticide market in the early 2000s to about 70 per cent today (AgbioInvestor 2019). [Chapters 2.5.1 and 2.7.9]. The growth of the global pesticide market has been accompanied by an important rise in the trade of illegal pesticides. They include banned or otherwise non-authorized pesticides, as well as counterfeit, fake, and illegally labelled or packaged products. Illegal pesticides can damage crops, harm human health and contaminate the environment. Although no precise estimates are available, the value of illegal pesticide sales is believed to represent 10-15 per cent of the legitimate global pesticide market (United Nations Interregional Crime and Justice Research Institute [UNICRI] 2016). [Chapter 2.5.3]

Application equipment and methods can greatly influence environmental and human exposure to a pesticide, as can the way pesticides are formulated and marketed. Developments in pesticide application technology have resulted in vehicle-mounted and tractor sprayers with various engineering controls which minimize occupational and environmental exposure (Jensen and Olesen 2014; Matthews 2020). Such technology is not available or much used in low and middle income countries, where simple hand-held sprayers dominate with associated higher risks for operators and the environment (Matthews, Bateman and Miller 2014; Horne 2019). [Chapter 2.6]

Many factors influence the use of pesticides, either positively or negatively. Drivers can be agronomic, economic or regulatory. Pesticide use can also be influenced by public health, environmental or information considerations. Key drivers that tend to increase pesticide use are current practices of agricultural intensification, pesticide resistance, genetically modified crops (mainly for herbicide resistance), pesticide marketing practices, and commodity prices. Pesticide use is limited mainly by national legislation and policies, as well as by environmental and human health (including food safety) concerns. The type of information and training provided to pesticide users can lead to increasing or decreasing use of pesticides. [Chapter 2.7]

2.2 Types of pesticides

2.2.1 Pesticide definitions

An internationally agreed definition of pesticide is provided in the International Code of Conduct on Pesticide Management. It defines a pesticide as "any substance, or mixture of substances of chemical or biological ingredients intended for repelling, destroying or controlling any pest, or regulating plant growth", where a pest is defined as: "any species, strain or biotype of plant, animal or pathogenic agent injurious to plants and plant products, materials or environments and includes vectors of parasites or pathogens of human and animal disease and animals causing public health nuisance" (Food and Agriculture Organization of the United Nations [FAO] and World Health Organization [WHO] 2014). According to the definition in the International Code of Conduct, pesticides can be of chemical or biological origin; they may not only kill pests, but also repel or otherwise influence them, and they may include plant growth regulators.

Table 2.2-1 Specific groups of pesticides included in countries' pesticide definitions. Based on an analysis by the Secretariat of the Rotterdam Convention (Rotterdam Convention 2019).

Pesticide group		Intended uses	Number of countries (out of a total of 122)
A	Pesticides used on plants	a. Prevent, destroy or control undesirable species	114 (93 per cent)
		b. Promote or inhibit plant growth, modify physiology, or influence the life processes of plants	
		c. Protect commodities from deterioration during storage and transport by application to crops prior to or following harvest	
B	Pesticides used on animals	a. Prevent, destroy or control insects, arachnids or other pests in or on animals	83 (68 per cent)
		b. Prevent, destroy or control pests on hosts' bodies (ectoparasites)	
		c. Prevent, destroy or control vectors/transmitters of animal disease	
C	Pesticides used in public health and hygiene	a. Control vectors of human disease/ectoparasites on humans	76 (62 per cent)
		b. Domestic/household use	
D	Pesticides used on inanimate objects or in the environment	a. Protect food, feed and fodder	72 (59 per cent)
		b. Protect wood, clothes, textiles and fabrics	
		c Prevent or control organisms harmful to roadways, railways, dams, airports, ports, vessels, floats, buildings or, for example, in industrial or aquatic environments	

Included in the survey were 122 Parties to the Convention. In most cases the term pesticide was defined in national legislation.

While the term pesticide is generic in the sense that it covers all types of pests, several other terms are regularly used which tend to refer to specific pesticide uses. These include plant protection product, agrochemical, agricultural remedy and phytosanitary product (for pesticides used in agriculture), and biocide, public health pesticide and domestic pesticide (for non-agricultural pesticides).

Pesticides are defined differently by individual countries, often influenced by the ways they are intended to be used. A recent review by the Secretariat of the Rotterdam Convention of definitions of the term pesticide showed that a large majority of pesticide definitions referred to products intended for use in protecting plant health (Rotterdam Convention 2019) (Table 2.2-1). However, pesticide definitions in 30-40 per cent of the countries included in the review did not include products intended to protect animal or human health, or to protect inanimate objects or the environment. As many of these definitions are part of pesticide legislation, the review suggested that non-agricultural pesticides are less well regulated than agricultural pest control products, particularly in low and middle income countries (see also Chapter 3.4).

2.2.2 Pesticide categories

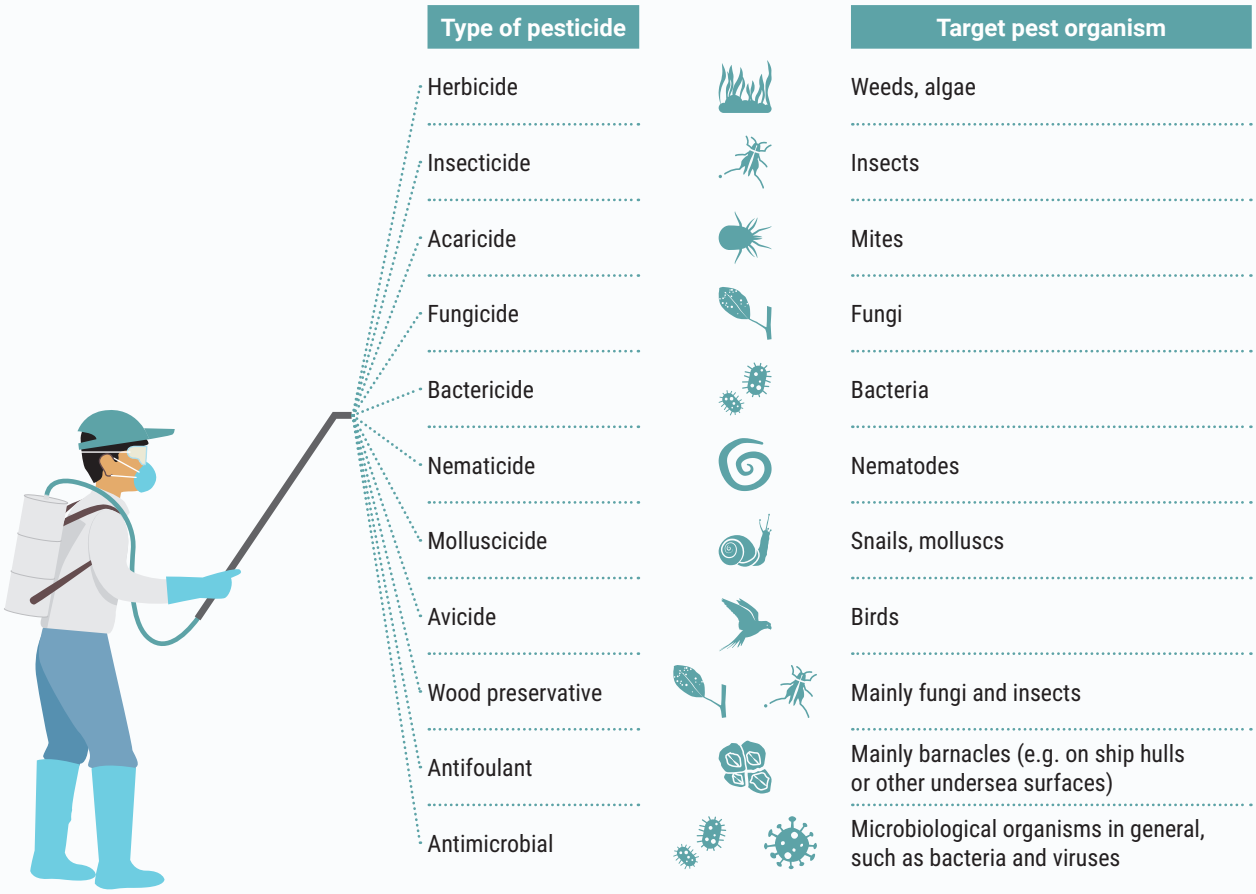
It is common to categorize pesticides according to the target organisms they are intended control. These categories include insecticides, fungicides, herbicides and rodenticides (Figure 2.2-1).

Pesticides can also be categorized according to their origin (chemical, biological or plant-incorporated).

Chemical pesticides

Some chemical pesticides (e.g., sulphur- and copper-based fungicides) are inorganic. However, a large majority of chemical pesticides on the market are synthetic organic compounds

Figure 2.2-1 Categorization of pesticides according to the target organisms they are intended control



Note: The definition of pesticides under the International Code of Conduct on Pesticide Management includes plant growth regulators, however this is not included in this figure as this group of pesticides does not target pest organisms

designed, synthesized and manufactured by the specialized pesticides industry. These pesticides are generally categorized according to the chemical groups to which they belong (e.g., organophosphates, neonicotinoids, triazoles and dinitroanilines) (Table 2.2-2) or their mode of action (e.g., cholinesterase inhibitors, fumigants, photosynthesis inhibitors).

Bioprotectants

Bioprotectants are products that originate in nature, can be sourced from nature, or are identical to their natural origin if synthesized (International Biocontrol Manufacturers Association [IBMA] 2019). They are also referred to as biological pest control agents (FAO and WHO 2017). Bioprotectants include semiochemicals (e.g., pheromones) and microbials (e.g., bacteria, fungi and viruses); natural substances such as botanical products; and invertebrate biocontrol agents (Figure 2.2-2). The environmental and human health risks of invertebrate biocontrol agents (or macrobials) in pest control are not covered in this report (but see, for example, van Lenteren *et al*. 2006; van Lenteren *et al*. 2018).

The term biopesticide is used to describe bioprotectants formulated and applied similarly to a conventional chemical pesticide (FAO and WHO 2017). Unlike chemical pesticides, many (although not all) biopesticides are very target-specific and will generally only control a limited range of pests, diseases or weeds (Table 2.2-2).

2.2.3 Components of a pesticide

A pesticide placed on the market is generally referred to as a pesticide product, pesticide

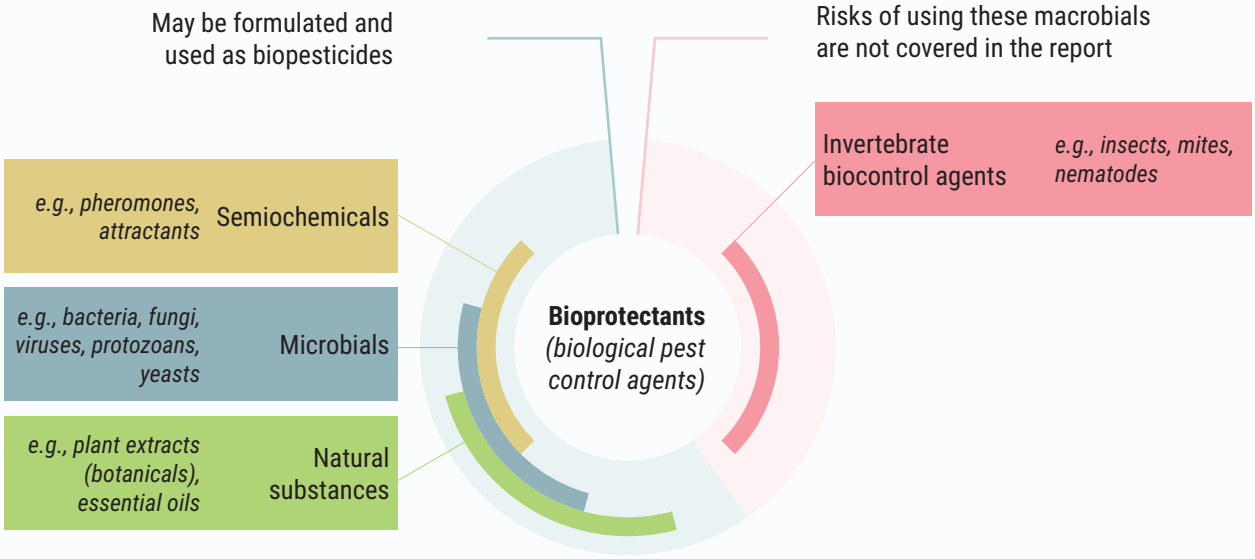
Table 2.2-2 Major classes of synthetic chemical insecticides, herbicides and fungicides and examples of well-known individual compounds. Adapted from Matthews (2018); Philipps McDougal (2018a); Wood (2019).

Insecticides	Herbicides	Fungicides
Organochlorines	Triazines	Dithiocarbamates
dichlorodiphenyltrichloroethane (DDT)	ametryn	mancozeb
endosulfan	atrazine	maneb
	cyanazine	thiram
Organophosphates	Phenoxy herbicides	Triazoles
chlorpyrifos-ethyl	2,4-D	difenoconazole
malathion	fluazifop-P	hexaconazole
pirimiphos-methyl	MCPA	tetraconazole
Carbamates	Chloroacetanilides	Strobilurines
bendiocarb	acetochlor	azoxystrobin
carbofuran	metolachlor	pyraclostrobin
oxamyl		
Pyrethroids	Dinitroanilines	Imidazoles
deltamethrin	pendimethalin	iprodione
tralomethrin	trifluralin	imazalil
Neonicotinoids	Quaternary ammonium herbicides	Carbamates
clothianidin	diquat	thiophanate-methyl
imidacloprid	paraquat	
thiamethoxam		
Pyrazoles	Organophosphonate herbicides	Benzimidazoles
fipronil	glyphosate	carbendazim
Diamides	Phenylureas	Aromatic fungicides
chlorantraniliprole	diuron	chlorothalonil
Avermectins	Sulfonylureas	Amides
abamectin	bensulfuron-methyl	metalaxyl
	metsulfuron	
	nicosulfuron	
Pyrroles		
chlorfenapyr		
Juvenile hormone mimics		
pyriproxifen		

formulation or formulated product. It consists of an active ingredient (or active substance) and one or more co-formulants. The active ingredient is the part of the product that provides the pesticidal action (FAO and WHO 2014). The manufactured active ingredient (known as the technical material) consists of the active ingredient, together with associated impurities, and sometimes small amounts of necessary additives (FAO and WHO 2016).

Apart from active ingredients, plant protection products contain co-formulants which give the product the necessary properties for application. These are the non-active ingredient components of a formulated product (FAO and WHO 2014), sometimes also referred to as inert ingredients or inerts. Co-formulants make plant protection products easy to handle, apply and store. They can improve operator safety, help disperse the active ingredient evenly in the spray liquid,

Figure 2.2-2 **Bioprotectants covered in this report include semiochemicals, microbials and natural substances, but not invertebrate control agents. Some bioprotectants may be formulated and used similarly to chemical pesticides, in which case they are sometimes referred to as biopesticides.**
Based on DunhamTrimmer (2019); IBMA (2019).



or facilitate spreading on plants. Examples of co-formulants are wetting agents, solvents, emulsifiers, carriers, anti-evaporants, synergists, dyes, stabilizers and safeners (German Federal Office of Consumer Protection and Food Safety 2019). However, co-formulants can be hazardous substances whose risks also need to be evaluated. Some co-formulants have been shown to be more toxic than the pesticide active ingredient, and certain regulatory agencies have established lists of prohibited co-formulants.

Chemicals added to a pesticide formulation to increase its effectiveness are referred to as adjuvants; these are generally added separately to the pesticide product in the spray tank.

Table 2.2-3 **Examples of common biopesticides on the market.** Adapted from DunhamTrimmer (2019); IBMA (2019).

Biochemicals		
Example	Group	Target organism
Trimedlure	Pheromone	*Mediterranean fruit fly*
(E,Z)-tetradeca-9, 12-dienyl acetate	Pheromone	Beet armyworm
Azadirachtin A	Botanical	Wide range of insects

Microbials		
Example	Group	Target organism
Bacillus thuringiensis subsp. kurstaki	Bacteria	Lepidopteran larvae
Bacillus thuringiensis subsp. israelensis	Bacteria	Mosquito larvae
Metarhizium anisopliae (various strains)	Fungi	E.g. Coleoptera larvae, termites)
Metarhizium acridum strain IMI 330189	Fungi	Locusts and grasshoppers
Trichoderma atroviride	Fungi	Various fungal pathogens
Helicoverpa zea single-enveloped nucleopolyhedrovirus (HzSNVP)	Virus	American cotton bollworm
Bacillus amyloliquefaciens D747	Bacteria	Fungal pathogens

2.3 History of pesticide development

Since the early stages of agriculture, pesticides have been used to protect crops. Reportedly, the first known pesticide was elemental sulphur used about 4,500 years ago in Mesopotamia. The Rig Veda, an ancient Indian collection of Vedic Sanskrit hymns composed some 3,500 years ago, mentions the use of poisonous plants for pest control (Pandya 2018). The Greek poet Homer described the benefits of sulphur 3,000 years ago as a "pest averting" substance. Pyrethrum was known in Persia around 400 B.C., where it is thought to have been used to protect stored agricultural products (Matthews 2018).

In the 17th and 18th centuries substances such as vinegar, brine, tobacco extracts, sulphur, oil soap, arsenic and copper were used as pesticides. In the 19th century Bordeaux mixture (copper sulphate, water and lime) was, apparently accidentally, found to be effective against downy mildew on grapevines. Paris green (or Schweinfurt green), containing copper acetoarsenite, was used from about 1860 onwards as an insecticide and rodenticide (Matthews 2018).

In the early 20th century plant extracts such as pyrethrum, tobacco and rotenone, and inorganic chemicals such as arsenic, copper and sulphur were the main compounds used as insecticides and fungicides.

Until the 1940s pest management was mainly based on agronomic measures such as crop rotation, intercropping, mechanical measures (e.g., hand picking of pest insects), field hygiene (e.g., removal of crop residues), and promotion of pests' natural enemies (Chittenden 1899; Vayssière and Mimeur 1926). Pesticides only played a limited role.

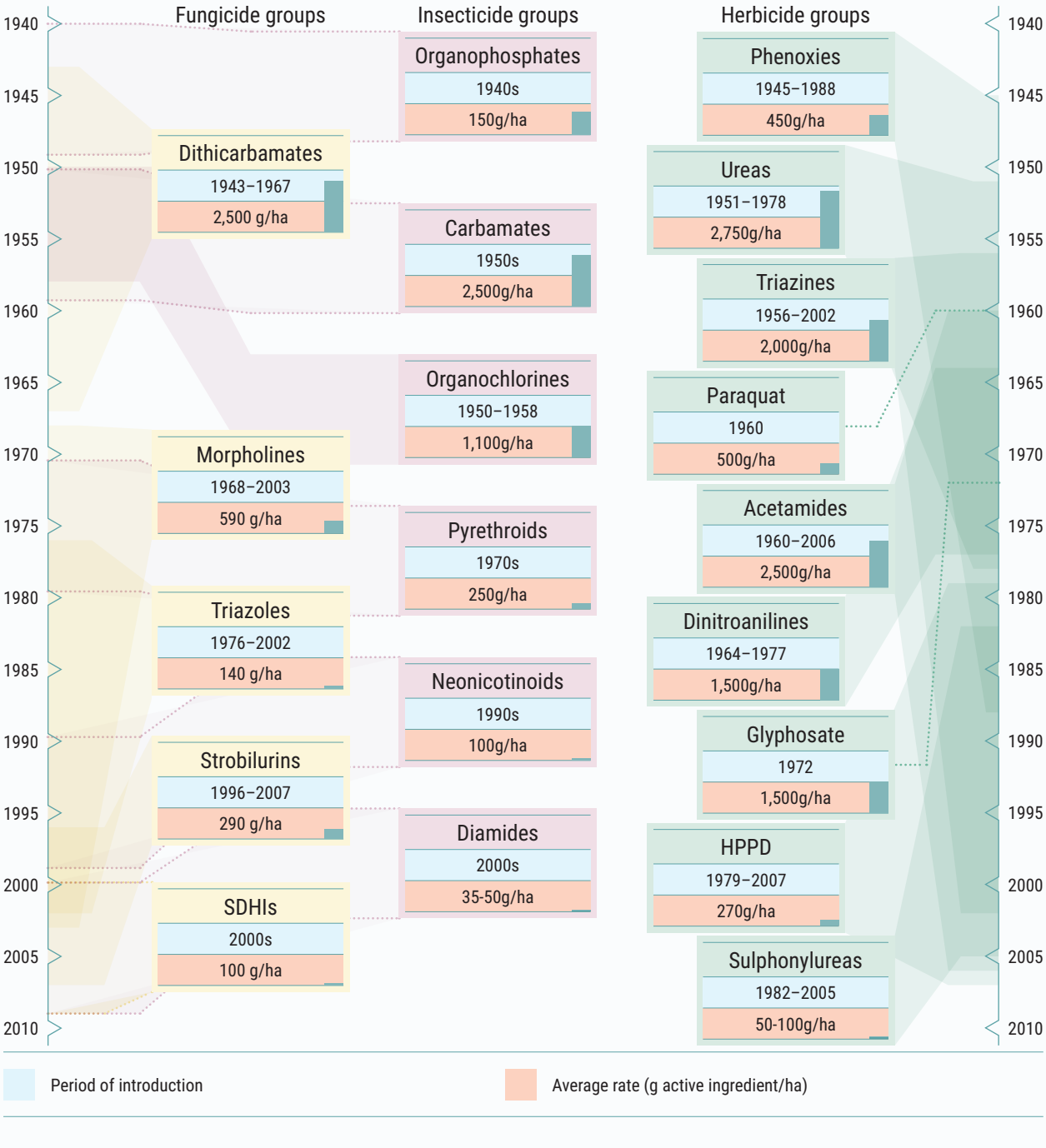
In the 1940s organic chemicals began to be synthesized which could be used as insecticides, herbicides and fungicides, representing a revolutionary change in pesticide development. Organochlorine and organophosphorus insecticides and acaricides, phenoxy herbicides, and dithiocarbamate fungicides all came on the market in the 1940s and 1950s. Some of these chemicals are still used today. Many other synthetic organic pesticides followed (Matthews 2018; Phillips McDougal 2018a).

More recently developed classes of insecticides include neonicotinoids, phenyl-pyrazoles, pyrroles, diamides, spinosyns, avermectins and ketoneols. Modern fungicides belong to groups such as the strobilurins and carboximides (succinate dehydrogenase [SDH] inhibitors). On the other hand, few new herbicide groups have come on the market in the last 30 years, and more recently developed herbicides belong to already known groups such as the pyrimidindiones, triazolones and benzoylpyrazoles (4-Hydroxyphenylpyruvate dioxygenase [HPPD] targeting herbicides) (Matthews 2018; Phillips McDougal 2018a) (Figure 2.3-1).

Over time new pesticide groups have become more effective for each gram of active ingredient used. Application rates in the 1950s averaged 1,000-2,500 grams of active ingredient per hectare (g/ha); by the 2000s they fell to 40-100 g/ha (Lamberth *et al*. 2013; Phillips McDougal 2018a) (Figure 2.3-1). Greater biological efficacy means farmers need a lower volume of pesticides to control the same pests. However, with the increased bioefficacy of modern pesticides has come higher toxicity to certain groups of non-target organisms.

In the 1960s there were about 100 active ingredients on the market, compared with some 600 synthetic chemical active ingredients today. In addition, there are currently around 300 biopesticide active substances and organisms (Phillips McDougal 2018a). The number of introductions of new pesticide active ingredients increased until the 1990s, but these introductions have significantly declined during the last two decades (Figure 2.3-2). While major companies annually invest 7-10 per cent of their sales in research and development (R&D), it has become increasingly difficult to develop and register new pesticide active ingredients. This is partly due to stricter environmental and human health requirements in large economies such as those

Figure 2.3-1 **Chronology and application rates of the main fungicide, insecticide and herbicide groups.**
Phillips McDougall (2018a).



in North America and the European Union [EU]). However, the number of compounds that need to be synthesized and screened to deliver one new market introduction has also increased considerably, from around 52,500 in 1995 to around 140,000 in 2005 (Lamberth *et al*. 2013).

For example, following the introduction of the EU-wide registration system for plant protection products in 1991 the authorizations of more than half the 1,000 active ingredients on the market were not renewed (European Union [EU] n.d.). This was partly because the pesticides did not meet the more stringent environmental and human health criteria, and partly because the pesticide industry did not consider the generation of additional data economically worthwhile and therefore withdrew applications for renewal.

Figure 2.3-2 Number of new pesticide active ingredients introduced globally per decade. Phillips McDougal (2018a).

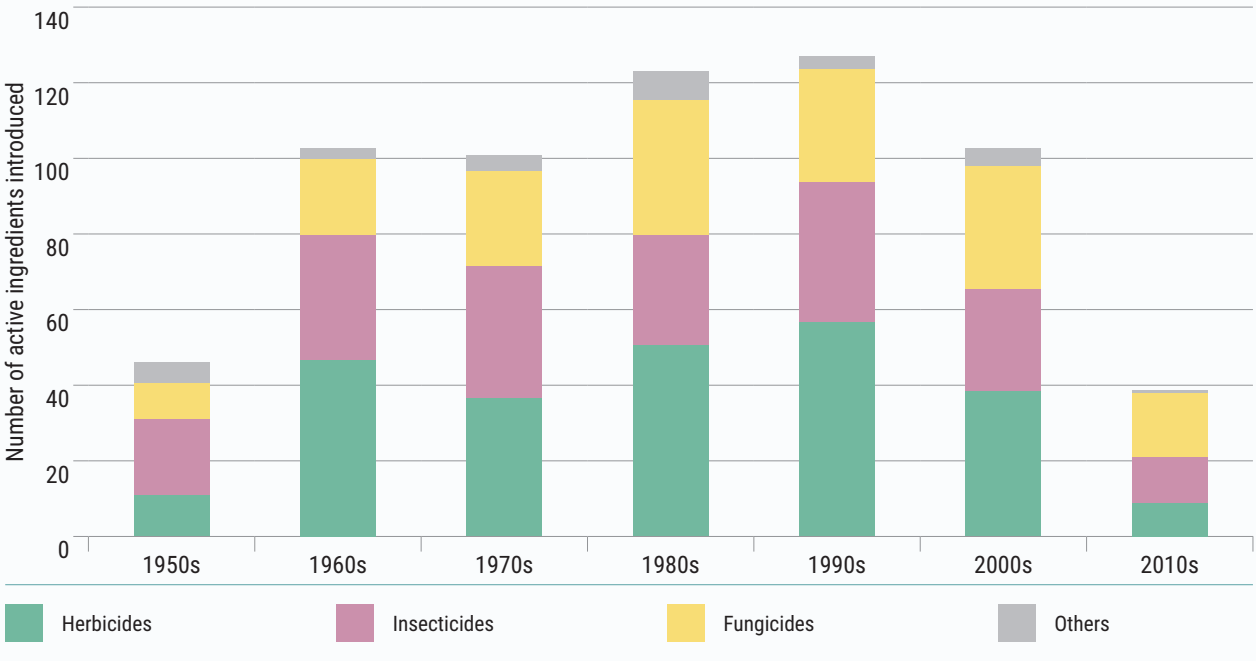
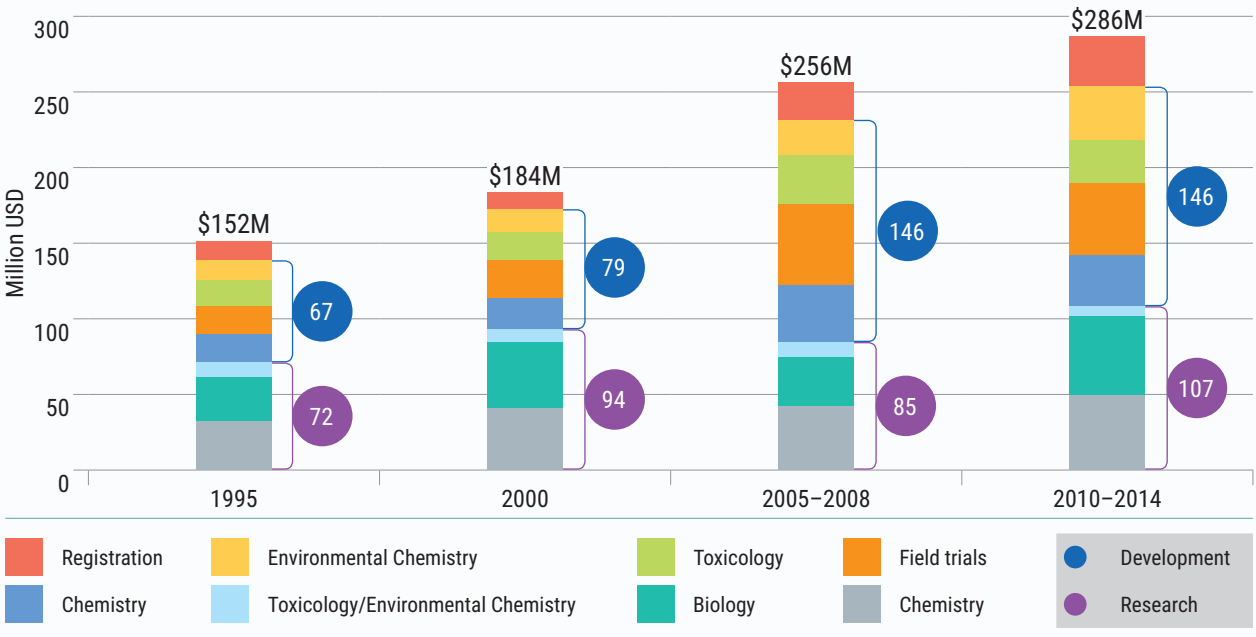


Figure 2.3-3 Discovery and development costs (nominal values, i.e. not adjusted for inflation) of a crop protection product based on a new active ingredient increased almost 90 per cent between 1995 and 2014. Phillips McDougal (2018a).



Note: Estimates are for five leading multinational pesticide companies. As a comparison, USD cumulative inflation over the same period was about 55% (CPI 2019).

As a result of the above and other factors, the overall nominal costs of the discovery and development of a new active ingredient intended for use in crop protection almost doubled between 1995 and 2010-2014, from United States dollars (USD) 152 million to USD 286 million (Figure 2.3-3).

About one-third of these costs are regulatory, i.e., related to registration of the pesticide (represented in the figure as the combined costs of registration, environmental chemistry and toxicology). In the same period the time required to develop and introduce a new pesticide active ingredient increased from about eight to about 11 years (Phillips McDougal 2018a).

In the last 20 years the number of new biopesticide introductions has frequently exceeded introductions of conventional pesticides. This trend is likely to continue. 2017 was the first year in which there were more patents for biopesticides than for conventional crop protection products: 173 compared with 117 (Phillips McDougal 2018a). At the same time, it should be remembered that not all patents result in a commercial product.

2.4 Historical and current use of pesticides

2.4.1 Pesticide manufacturing

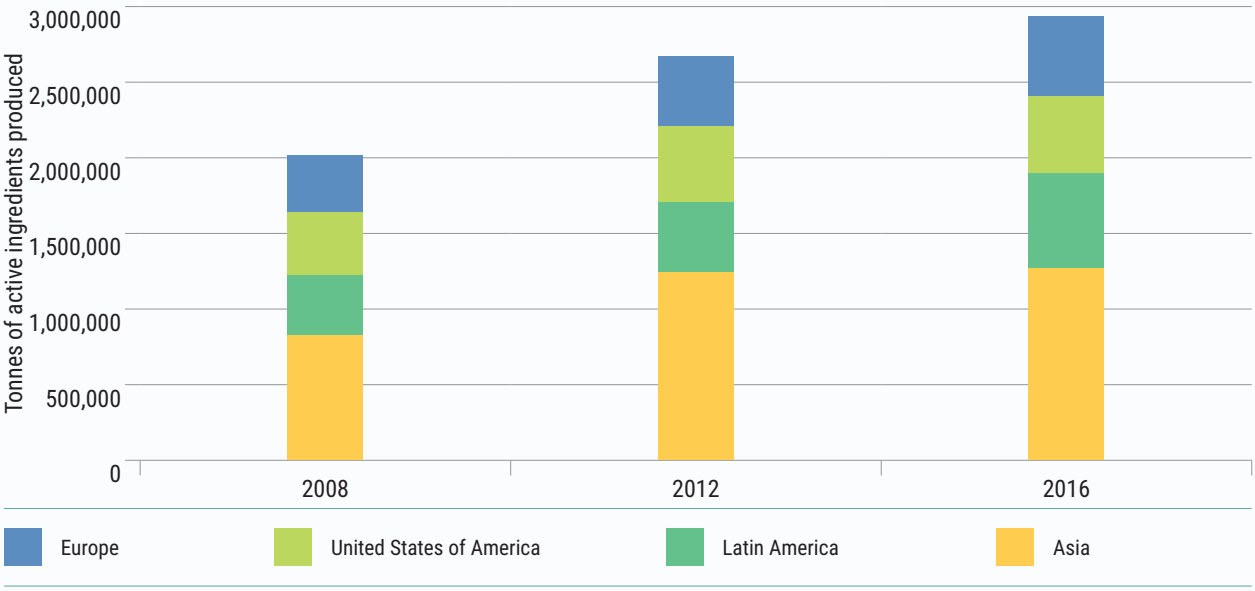
Global manufacturing of pesticide active ingredients has shown continuous growth in the last decade (Figure 2.4-1). Asia is the largest pesticide producer, with the greatest manufacturing capacity in China (where production more than doubled between 2008 and 2016). Growth of manufacturing during this period was highest in China (10 per cent compound annual growth rate), India (8 per cent), other parts of Asia (6 per cent) and Latin America (6 per cent). Pesticide production in other regions also increased, albeit at a slower pace (Oliver 2018).

2.4.2 Global pesticide use trends – volume

Pesticide use increased steadily following the introduction of synthetic organic pesticides in the 1940s and 1950s. Since the early 1990s global pesticide use in agriculture has almost doubled (Figure 2.4-2), amounting in 2016 to 4.1 million tons of active ingredients (FAOSTAT 2019).

Marked differences exist among world regions. The highest overall growth in pesticide consumption is in South and Central America and the Caribbean (an almost four-fold increase over

Figure 2.4-1 Global manufacturing of pesticides has increased steadily. Based on Oliver (2018).



25 years) (Figures 2.4-3 and 2.4-4). Asia uses more pesticides than any other region and has experienced the highest absolute growth in volume. Oceania has the lowest use of pesticides, but has shown the second highest growth rate. Pesticide use in northern America has only increased by about 10 per cent, while in Europe a slight decrease has been seen.

Estimates of pesticide use (Figure 2.4-2) are 25-40 per cent higher than those of pesticide manufacturing (Figure 2.4-1). This may be attributable to reliance on different sources: data from the FAO's FAOSTAT database for pesticide use and United Nations (UN) trade statistics and industry data for manufacturing, which have somewhat different geographical coverage.

Difference in pesticide use is also associated with per capita gross domestic product (GDP). Between 1991 and 1995 the majority of the volume of pesticides was used by what were then low income and high income countries, while in 2012-2016 the majority was used by upper-middle income countries (Figure 2.4-5). This change has been heavily influenced by China's development from a low income country in 1995 to an upper-middle income country in 2016. Even without taking into account pesticide use in China, however, upper-middle income

Figure 2.4-2 Global use of pesticides in agriculture increased from about 2.3 million tons of active ingredient in the early 1990s to more than 4 million tons in 2016. FAO (2019a).

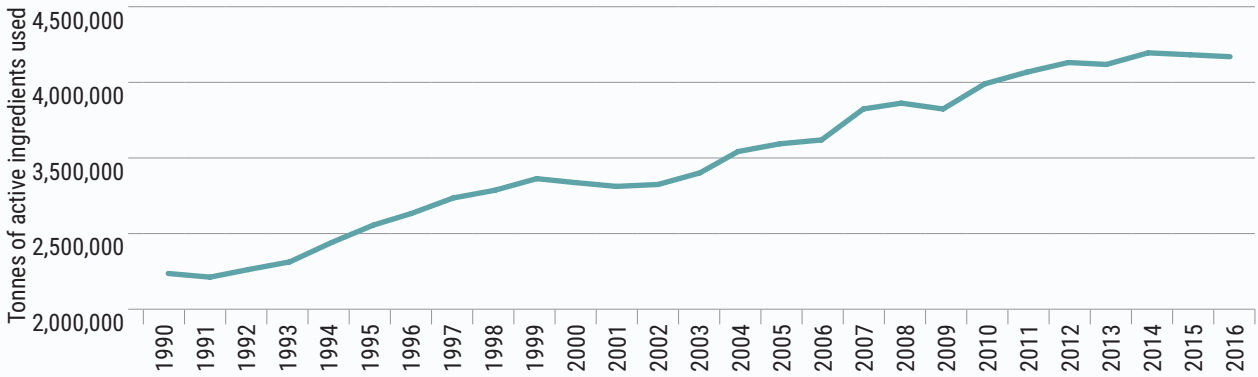


Figure 2.4-3 Pesticide use in agriculture has exhibited different trends in the major regions of the world. FAO (2019a).

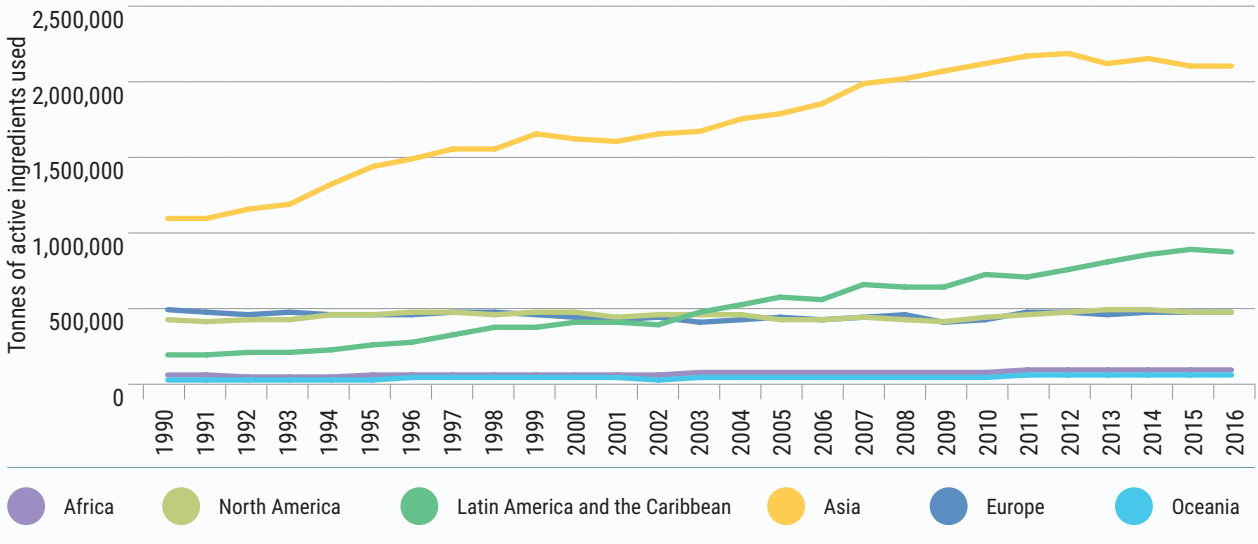


Figure 2.4-4 **Overall growth in agricultural pesticide use (volumes of active ingredient) between 1990 and 2016 was highest in Latin America and the Caribbean. It was relatively stable in northern America and Europe.** FAO (2019a).

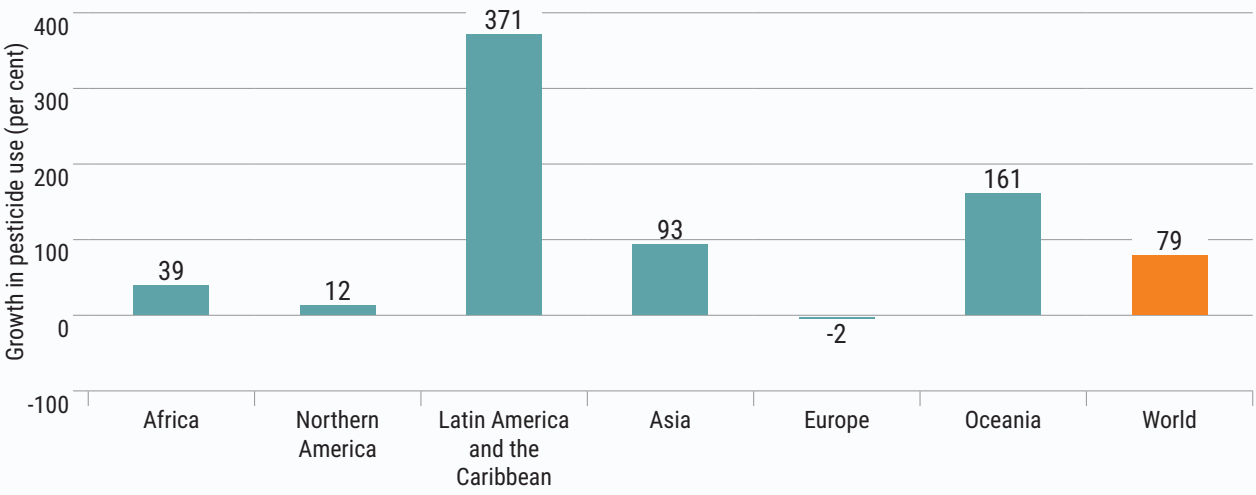
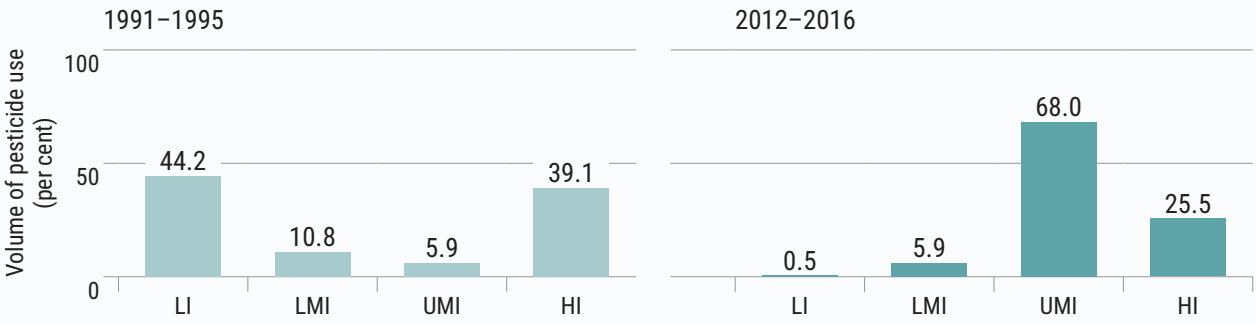
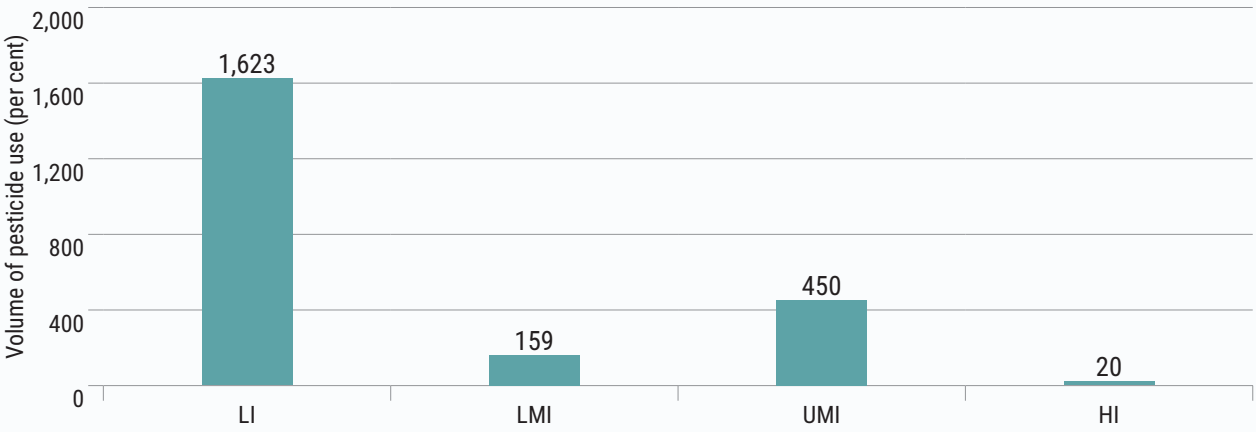


Figure 2.4-5 **Share of the total volume of pesticides used by countries according to income group in 1991-1995 and 2012-2016.** FAO (2019a).



LI = Low-income countries; LMI = Lower Middle-income countries; UMI = Upper Middle-income countries; HI = High-income countries
Note: For this analysis, 101 countries are included for which pesticide use data were updated on a regular basis in FAOSTAT (FAO 2019a); countries were omitted from the analysis if pesticide use data had not been updated for any 10 years or more during the 26 year time series. Income groups are as classified by World Bank (2020a) for calendar years 1995 and 2016

Figure 2.4-6 **Average increase in the volumes of pesticides used in countries according to income group between 1991-1995 and 2012-2016.** FAO (2019a).



Note: Data and abbreviations as in Figure 2.4-5.

countries are responsible for a major share of global pesticide use, likely linked to large-scale agricultural extension and intensification in these countries. Use in low income and lower-middle income countries in 2012-2016 did not exceed 7 per cent of total global pesticide use.

Despite the relatively limited use of pesticides in low income countries, it is in those countries that the relative increase in pesticide use since the early 1990s has been greatest (Figure 2.4-6).

2.4.3 Global pesticide use trends

Value

The total value of the pesticide market in 2018 was estimated at USD 65 billion, of which USD 57.6 billion (88 per cent) for crop protection products and USD 7.5 billion for non-crop protection purposes (e.g., home and garden, pasture, wood preservatives, public health, industrial) (Agrow 2019). The nominal value of the pesticide market (sales) has shown an overall steady increase since 2006 for crop protection products and non-crop pesticides (Figure 2.4-7). Over time, crop protection products represent a stable share (86-90 per cent) of the total pesticide market.

Although the value of the pesticide market has fluctuated from year to year, during the last decade there has never been a shrinking market for more than two years in a row (Figure 2.4-8). Between 2006 and 2018 the pesticide market grew by 4.8 per cent per year on average in both nominal and real terms.

In 2019 the largest regional markets for crop protection products were Asia and the Pacific and Latin America (Figure 2.4-9). The highest compound annual growth rate (2018 to 2019) was in Latin America (8 per cent). All other regions experienced a shrinking crop protection market.

Targets

The majority of pesticides used are herbicides, which amounted to some 61 per cent of the total volume of crop protection products applied globally in 2016. The share of herbicides has been growing since the 2000s (Figure 2.4-10). Fungicide use has also increased in the last few decades because of growing demand for fruits and vegetables, among other reasons. Insecticide use has remained fairly stable in volume terms, but has decreased as a percentage of the total crop protection product market.

Figure 2.4-7 The nominal value of global pesticide sales has increased steadily since 2006. Agrow (2016); Phillips McDougal (2017); Agrow (2019); Agrow (2020).

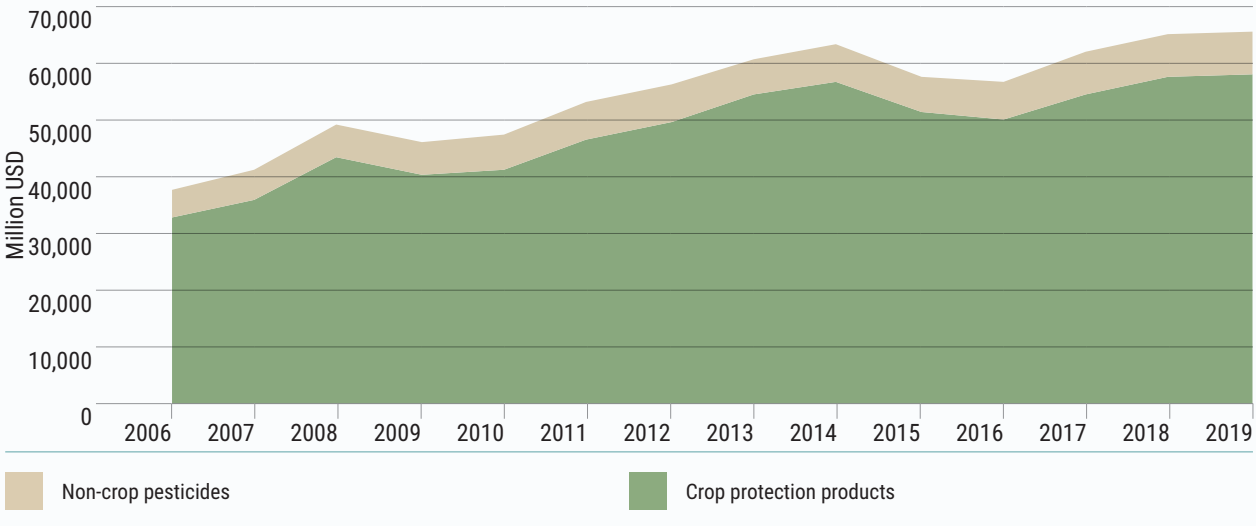
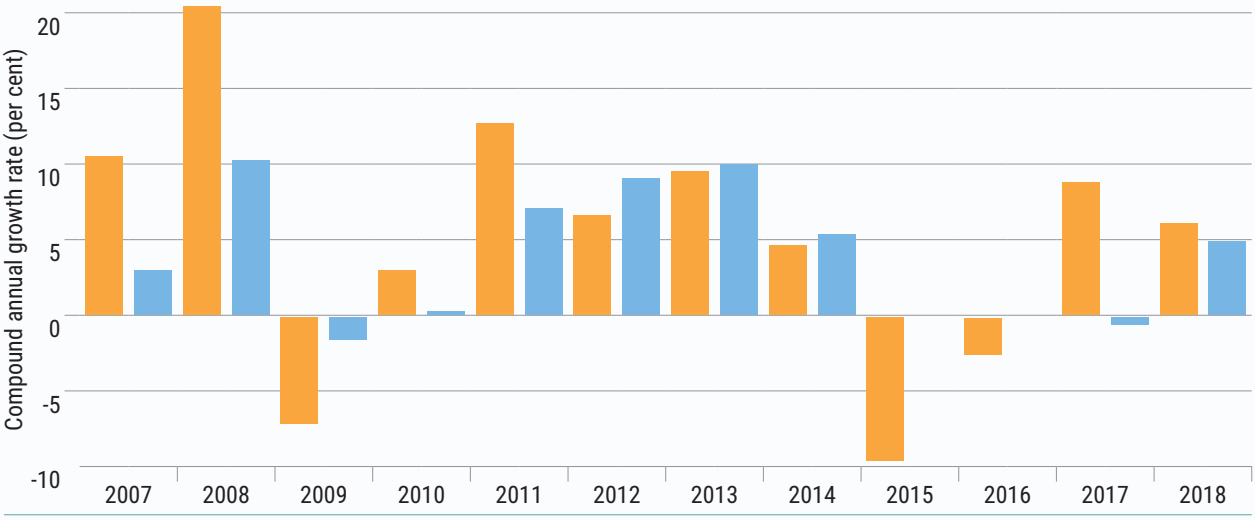


Figure 2.4-8 Compound annual growth rates for the global pesticides market, based on nominal and real term values (adjusted for variations in USD exchange rates and inflation). Agrow (2016); Phillips McDougal (2017); Phillips McDougal (2018a); Agrow (2019).



Note: Real term values for 2015 and 2016 are missing.

As an example of changing pesticide use patterns, farmers in the United States used very different types of pesticides on major crops in 2008 compared with 1960 (Figure 2.4-11). The growth of herbicide use in that country is also illustrated by the percentage of crop area treated. Approximately 5-10 per cent of corn (maize), wheat and cotton crops were treated with herbicides in

Figure 2.4-9 Regional markets for crop protection products and annual growth rates. Market value is expressed as USD million and percentage year-on-year change is from 2018 to 2019. Phillips McDougal (2019a), IHS Markit (personal communication).

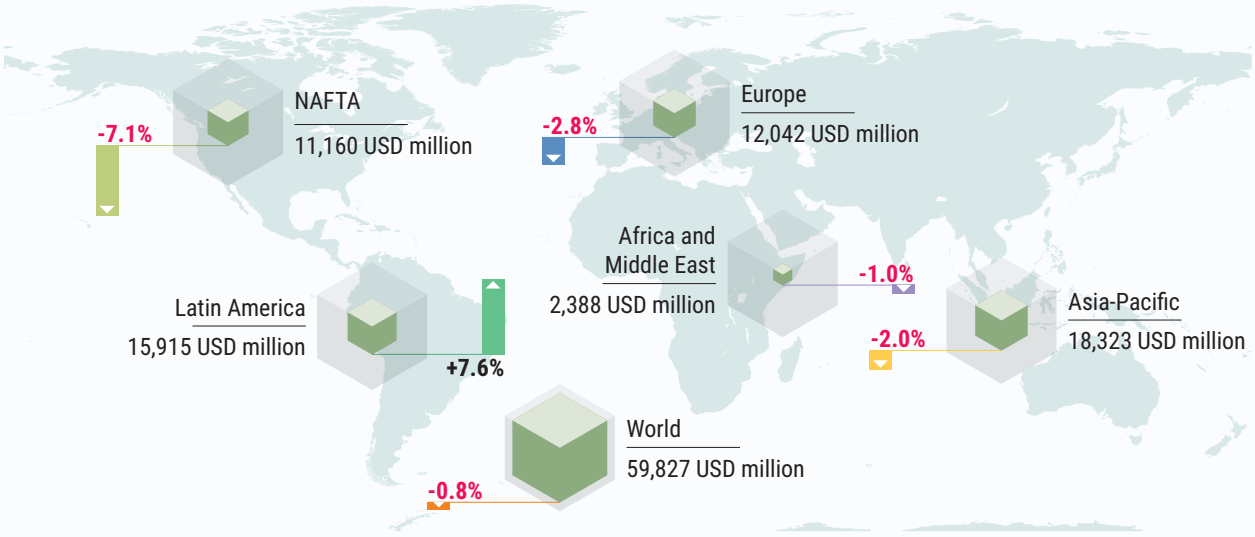


Figure 2.4-10 **Major groups of crop protection pesticides as percentage of the total volume used globally.**
Based on Phillips McDougal (2018b).

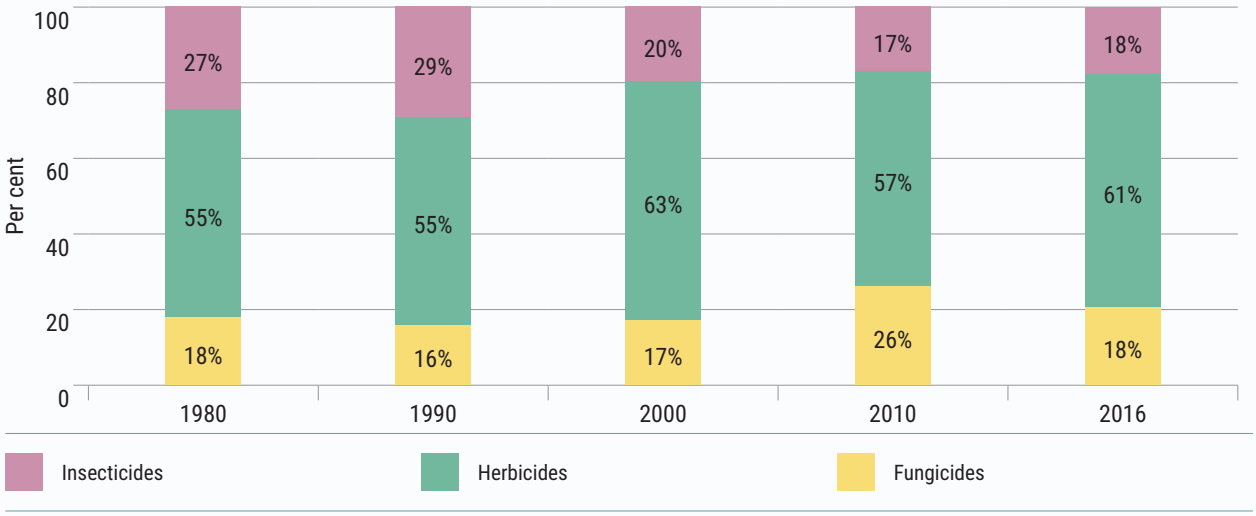
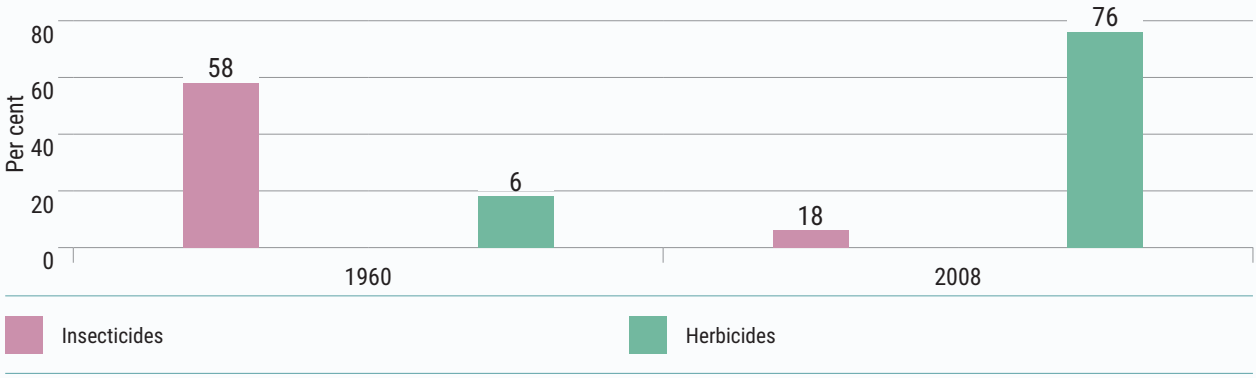


Figure 2.4-11 **A major shift has occurred in pesticide use in the United States. Insecticides accounted for 58 per cent of pesticide volume applied in 1960, but only 6 per cent in 2008. Herbicides accounted for 18 per cent of the volume applied in 1960, compared with 76 per cent by 2008.** Fernandez-Cornejo *et al.* (2014).



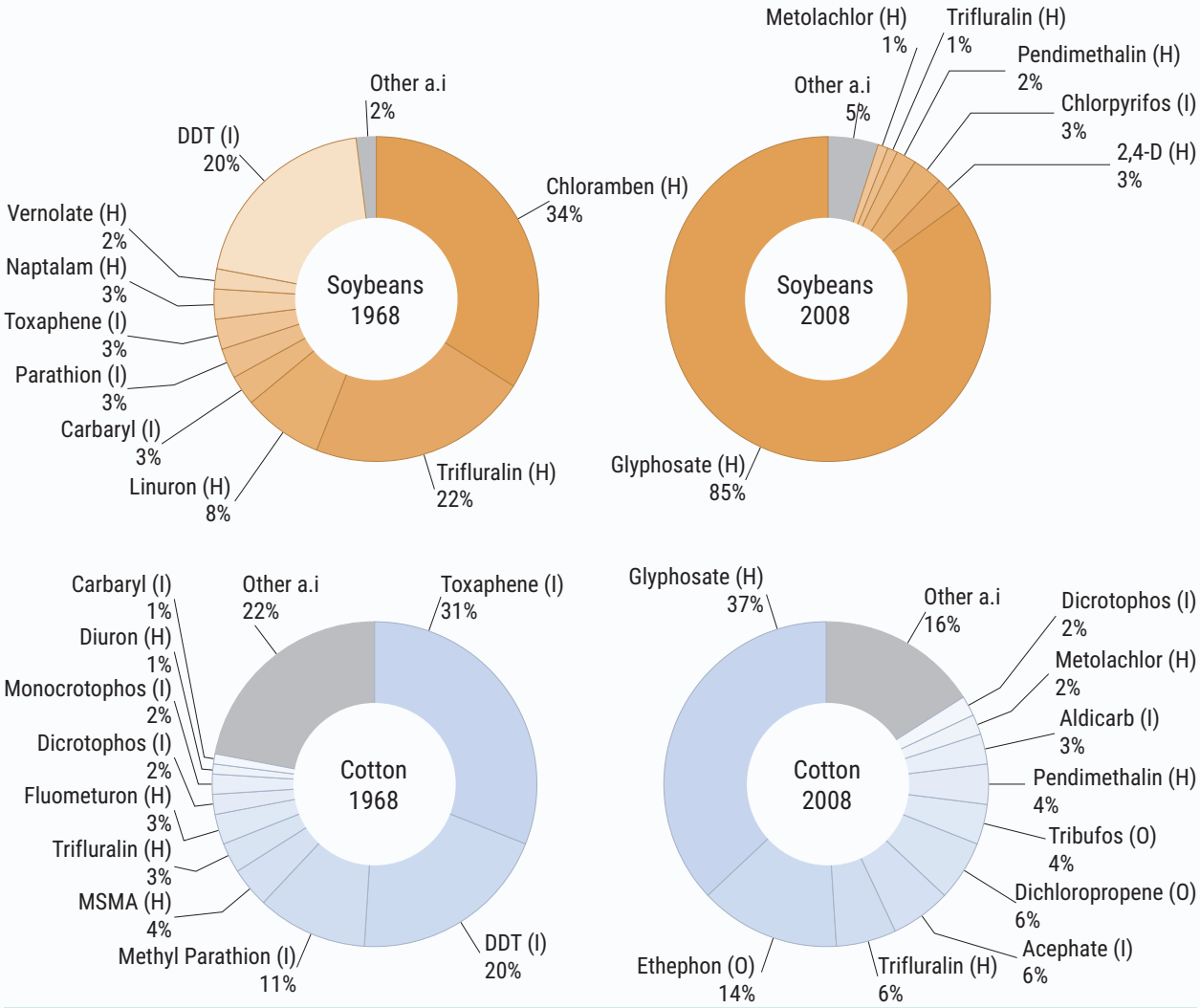
the early 1950s; by 1980 herbicides were used on 90-99 per cent of the area planted with corn, cotton and soybeans (Fernandez-Cornejo *et al.* 2014). This is further supported by another study which shows that between 1993 and 2015 the total applied mass of insecticides has decreased while the total applied mass of herbicides has increased in the United States (Schulz *et al.* 2021). These changing use patterns affect the environmental and human health risks posed by crop protection products (Chapter 4).

The spectrum of pesticide active ingredients used today is also very different from a few decades ago. For example, the dominant pesticides in soybean production in the United States in 1968 were the herbicides chloramben and trifluralin and the insecticide dichlorodiphenyltrichloroethane (DDT), while in 2008 the herbicide glyphosate dominated pesticide inputs (Figure 2.4-12). Similarly, the predominant pesticides used on cotton in 1968 were the insecticides toxaphene, DDT and methyl parathion; 40 years later they were the herbicide glyphosate and the plant growth regulator ethephon. Of the 17 active ingredients listed for 1968 in Figure 2.4-12, the use of eight had been prohibited in the United States by 2008.

Crops

Although pesticides are applied on many different crops, a large share of their use is concentrated

Figure 2.4-12 Change in pesticide use on soybeans and cotton in the United States between 1968 and 2008 (based on total volume of active ingredients applied). Fernandez-Cornejo *et al.* (2014).



a.i. = active ingredient; (H) = herbicide; (I) = insecticide; (O) = other.

on a limited number crop groups (Figure 2.4-13). Cereals are the largest market for pesticides globally, with maize, rice and other cereals representing 38 per cent of sales value. Fruits and vegetables represent one-quarter of the market and soybean another 15 per cent.

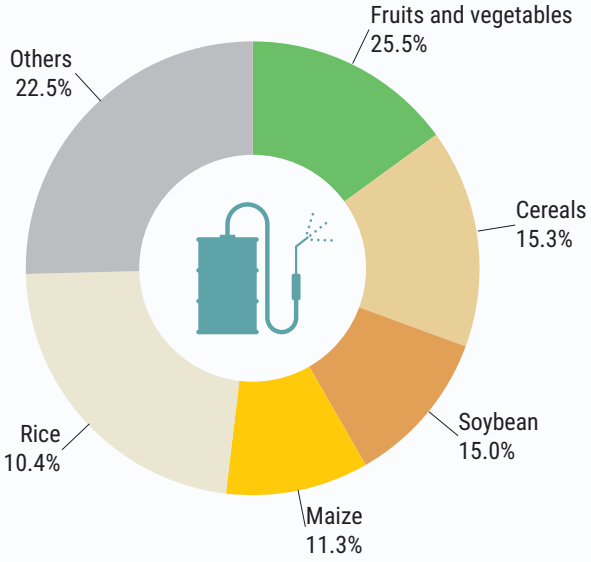
Non-agricultural pesticides

Non-agricultural pesticides include household and garden products, public health pesticides, products for disinfection of aircraft, hospitals and restaurants, and industrial pesticides, among others.

While both the sales and use of agricultural pesticides are relatively well monitored, much less is known about the use of non-agricultural pesticides. Such products tend to be used in or close to human habitations and therefore pose different (and sometimes higher) risks than agricultural pesticides. This is especially the case for domestic and public health pesticides (mainly pesticides used for vector control, in and around households, and by professional pest control operators). Sensitive groups such as children may also be exposed to such pesticides.

Non-agricultural pesticides represent about 12 per cent of the value of the global pesticide market (Figure 2.4-7). A considerable share of these pesticides is applied domestically, although no recent estimates of use volumes are available.

Figure 2.4-13 Pesticide use on different crop groups in 2018, based on global sales value. Phillips McDougal (2019a).



The World Health Organization (WHO) monitors the use of pesticides for the control of human disease vectors such as malaria, dengue, zika, leishmaniasis and Chagas disease. The latest review available is for the period 2000-2009 (WHO 2011; Van den Berg *et al*. 2012). On average during this period, 7,100 tons of pesticides were used annually for vector control, the equivalent of about 0.2 per cent of total agricultural pesticide use in those years. Of the total volume of pesticides applied for vector control, most was used in Southeast Asia (Table 2.4-1).

DDT, the only organochlorine pesticide covered in the WHO review, was used in larger quantities than any other insecticide class. It was exclusively applied for indoor residual spraying for malaria control. Of the global use of DDT in the period 2015-2017, 95 per cent was in India alone; the remainder was used in Africa

Table 2.4-1 Average reported insecticide use for vector control according to method of application and class of insecticide by WHO region (2000–2009), in metric tons of active ingredient per year. van den Berg *et al*. (2012).

WHO region[a]	Residual spraying				Space spraying		Treatment of nets (PY)[b]	Larviciding[c]	
	OC	OP	C	PY	OP	PY		OP	PY
Africa	805	19	19	24	0	0	12	1	0
Americas	0	97	4	164	276	66	0	82	0
Eastern Mediterranean	0	26	5	15	2	5	1	20	1
Europe	0	2	0	1	0	1	0	1	0
South-East Asia	3,623	483	2	39	15	1	4	49	0
Western Pacific	0	1	0	39	292	27	14	9	0
All	4,429	627	30	282	584	100	31	163	2

Abbreviations: C, carbamates; OC, organochlorines (DDT only); OP, organophosphates; PY, pyrethroids. ***a*** Canada and the United States (Americas region) and Australia and Japan (Western Pacific region) were not targeted, whereas in the European region, only Armenia, Azerbaijan, Georgia, Kyrgyzstan, Tajikistan, Turkey, Turkmenistan, and Uzbekistan were targeted. ***b*** Conventional application of insecticides to treat bed nets or curtains (excluding insecticides used in factory-made LNs). ***c*** The use of insecticides to treat aquatic breeding sites of mosquitoes.

Figure 2.4-14 **Trend in global use of vector control insecticides.** Based on van den Berg *et al.* (2012).

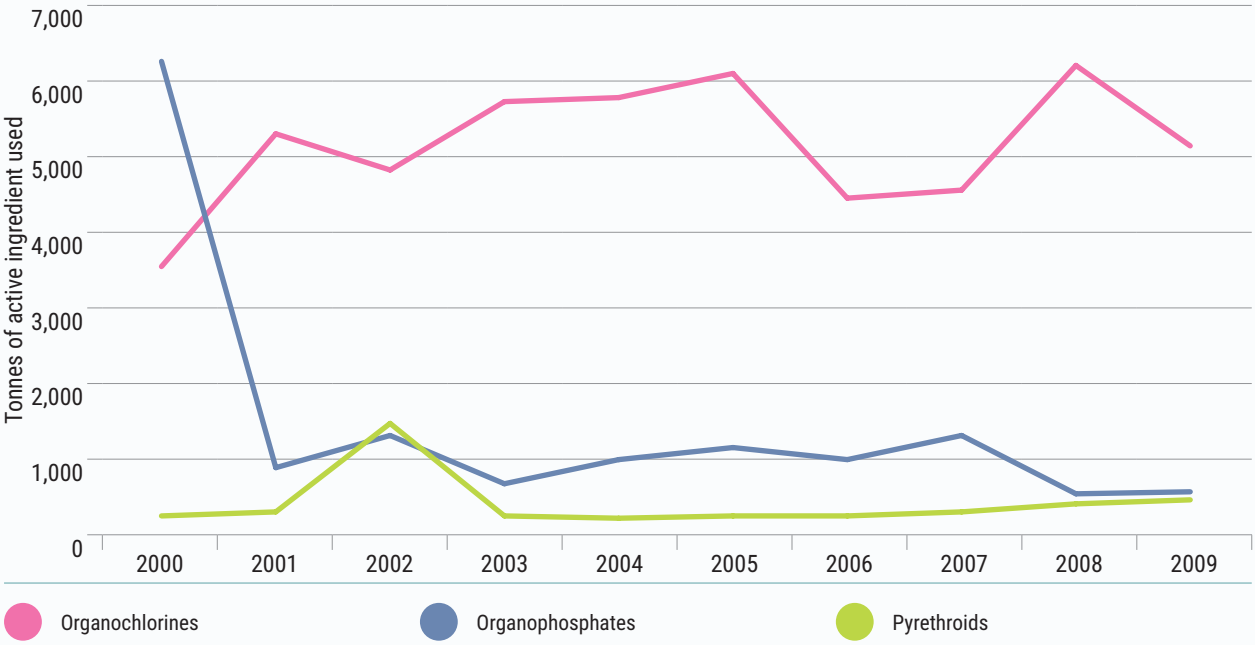
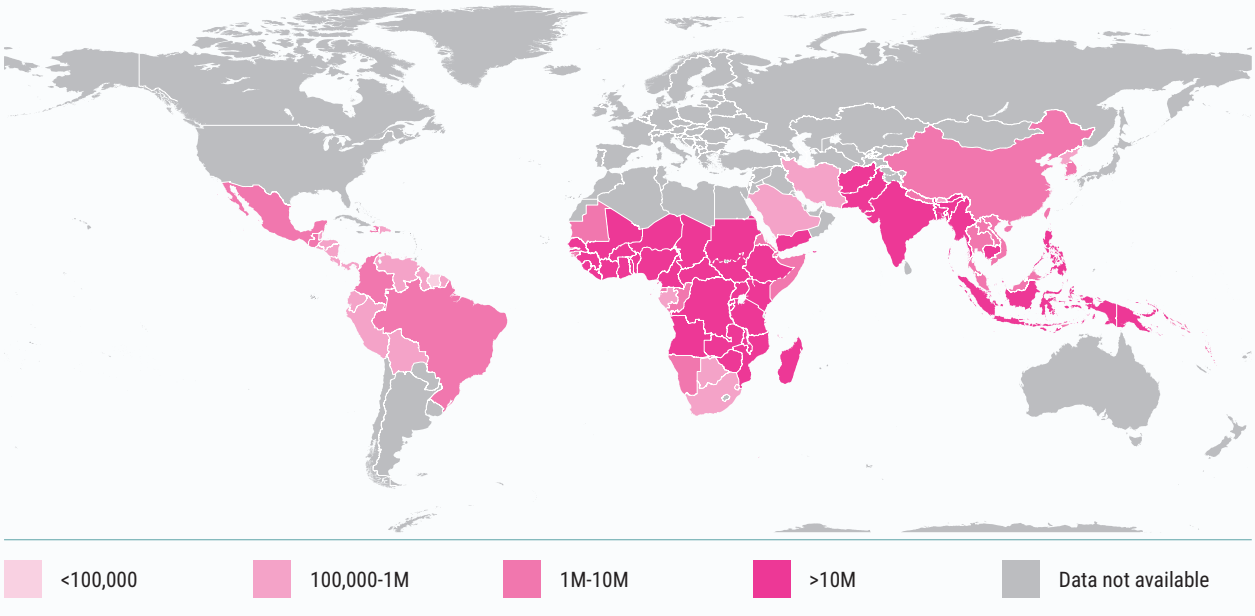


Figure 2.4-15 **Cumulative number of insecticide-treated nets delivered globally from 2004-2019 in sub-Saharan Africa and 2009-2019 in other countries. Almost 2.1 billion insecticide-treated nets were delivered in this period.** Alliance for Malaria Prevention (2020).



(United Nations Environment Programme [UNEP] 2019). Use of DDT, while much lower than in earlier decades, remained fairly stable in the period 2000-2010 (UNEP 2017 and Figure 2.4-14), but declined by about 50 per cent between 2010 and 2017 (UNEP 2019). Use of organophosphates had decreased considerably since the 1990s. Pyrethroids did not constitute a major global share in terms of tons applied. Due to their higher biological activity and much lower effective dosage against mosquito vectors, however, pyrethroids accounted for 81 per cent of the global spray utility

in vector control (i.e., the surface area covered by an active ingredient) (WHO 2011; Van den Berg *et al*. 2012).

In addition, pyrethroids are used almost exclusively in insecticide-treated bed nets. The above estimates of vector control insecticides do not include ready-for-use, long-lasting insecticidal nets (LLINs), more than 2 billion of which have been distributed in malaria endemic countries over the last 15 years (Alliance for Malaria Prevention 2020) (Figure 2.4-15).

2.4.4 Bioprotectants

In 2018 global sales of all bioprotectants were estimated at about USD 3.8 billion, almost 7 per cent of the total crop protection product market (DunhamTrimmer 2019). Although they represent only a small share of the total pesticide market, sales of biocontrol products are growing rapidly at 15-20 per cent per year (Glare *et al*. 2012; DunhamTrimmer 2019) (Figure 2.4-16).

Figure 2.4-16 The global biopesticide market is a small but rapidly increasing part of the total market for crop protection products (nominal values: USD million per year). Based on Phillips McDougal (2018a); Agrow (2019); DunhamTrimmer (2019).

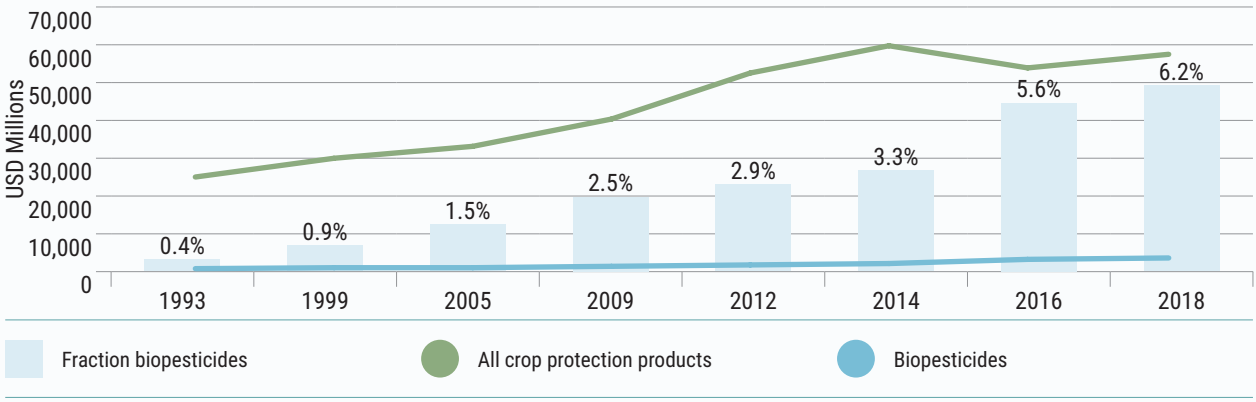
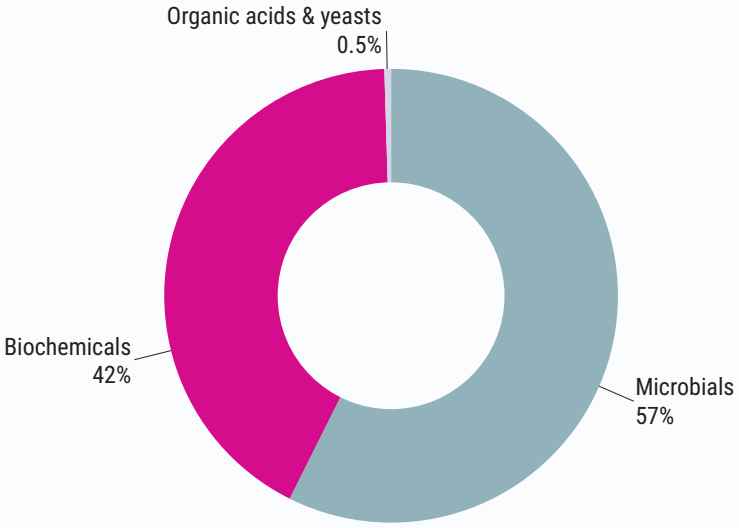


Figure 2.4-17 The biopesticide market is dominated by microorganisms, followed by biochemicals such as pheromones and plant extracts. Sales of organic acids and yeasts are relatively small. DunhamTrimmer (2019).



Biopesticides represent about 93 per cent of the bioprotectant market. The remainder consists of macrobials and non-formulated bioprotectants. Microorganisms account for more than half the biopesticide market; the rest is made up almost entirely of biochemicals (plant extracts and semiochemicals) (Figure 2.4-17). Bioinsecticides and biofungicides dominate the biopesticide market (DunhamTrimmer 2019).

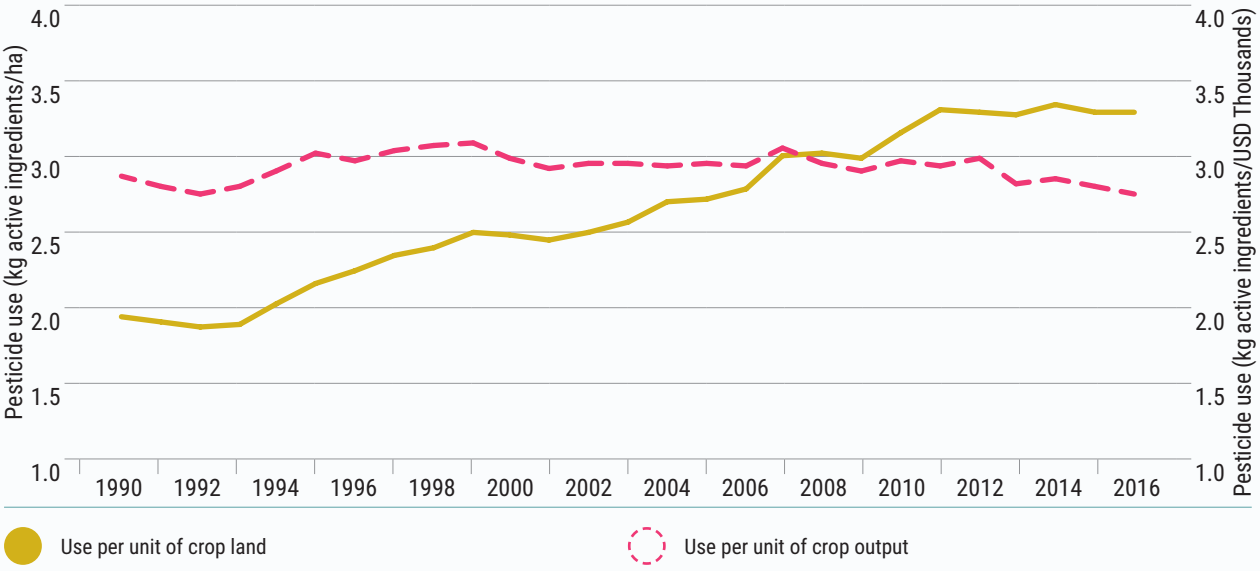
About three-quarters of the total bioprotectant market is accounted for by fruit and vegetable production. In this sector biocontrol (biopesticides and macrobials) represents about 18 per cent of the total crop protection market, considerably higher than the overall share of 7 per cent of the total crop protection product market. North America and Europe account for two-thirds of the biopesticide market, followed by Asia and the Pacific. Latin America is the fastest growing region for biopesticide sales (Agrow 2018).

The United States has long led in biopesticides registration, with over 350 biopesticide active ingredients presently registered. However, other world regions appear to be catching up (e.g., in Brazil only one biopesticide was registered in 2009 compared with almost 80 in 2018) (Agrow 2018). Europe has lagged behind in authorizing new biopesticides, but recently the number of new introductions of biopesticides has surpassed those of conventional chemical pesticides. Between 2011 and 2018 the number of registered biopesticides grew from 123 to 182, an increase of 48 per cent compared to 13 per cent for conventional pesticides in the same period (Robin and Marchand 2018).

2.4.5 Agricultural pesticide use intensity

While global pesticide use has steadily increased in the last 25 years, from an agronomic and environmental point of view it is more relevant to assess trends in pesticide use intensity (e.g., changes in pesticide use per unit area of cropland or per unit of agricultural output). Pesticide use per unit cropland is particularly relevant for environmental and human health risks. Pesticide use per unit crop output, on the other

Figure 2.4-18 Global agricultural pesticide use per unit cropland increased by 75 per cent between 1990 and 2016. FAOSTAT (2019a).



Note: ***Continuous line:*** overall pesticide use intensity in kg a.i. per ha crop land, for 108 countries for which pesticide use data were updated on a regular basis in FAOSTAT (2019a); countries were omitted from the analysis if pesticide use data had not been updated for any 10 years or more during the 27 year time series. ***Hatched line:*** overall pesticide use intensity in kg a.i. per 1000 constant International dollars of crop output, for the same countries. Be aware that this graph will be different from the pesticide sustainability indicator shown in FAOSTAT because of the larger number of countries included in the latter.

hand, is a more relevant indicator for agricultural production and intensification.

Figure 2.4-18 shows steady growth in pesticide use intensity. While an average of 1.9 kg active ingredient (a.i.) per ha of cropland was applied globally in the early 1990s, this increased to about 3.3 kg a.i./ha in the mid-2010s, a rise of almost 75 per cent. That global average is based on total area cropland. It disregards the fact that larger quantities of pesticides will be used on certain crops, while little or none will be used on others. This means the global average use intensity shown here underestimates real pesticide use per unit cropland for the crops on which they are actually applied.

Pesticide use per unit crop output has remained stable during the last 25 years, at approximately 2.9 kg a.i. of pesticide applied for every 1,000 International dollars of crop production (Figure 2.4-18). International dollars are often used as a proxy for the volume of crop production, which allows aggregating different crops (FAO 2019a). This indicator shows that the increase in pesticide use per unit cropland, on average, was associated with a similar increase in crop production. However, as indicated in Chapter 2.3, every kilogram of pesticide active ingredient has become more biologically active over time. While pesticide use intensity per unit output remained stable, the active load of pesticides on target pests (and non-target organisms sensitive to the same pesticide mode of action) is continuing to increase.

It is important to note that global average pesticide use per unit output does not represent an evaluation of the costs and benefits of pesticide use (Chapter 5).

Pesticide use can be very different between countries and regions. Schreinemachers and Tipraqsa (2012) analysed pesticide use data from the mid-1990s to the mid-2000s. They found that pesticide use increased with the income group to which a country belongs (i.e., there was higher pesticide use, both per unit of cropland and per unit of crop output, in countries with higher income levels). Only at the highest per capita GDP did pesticide use intensity appear to level off.

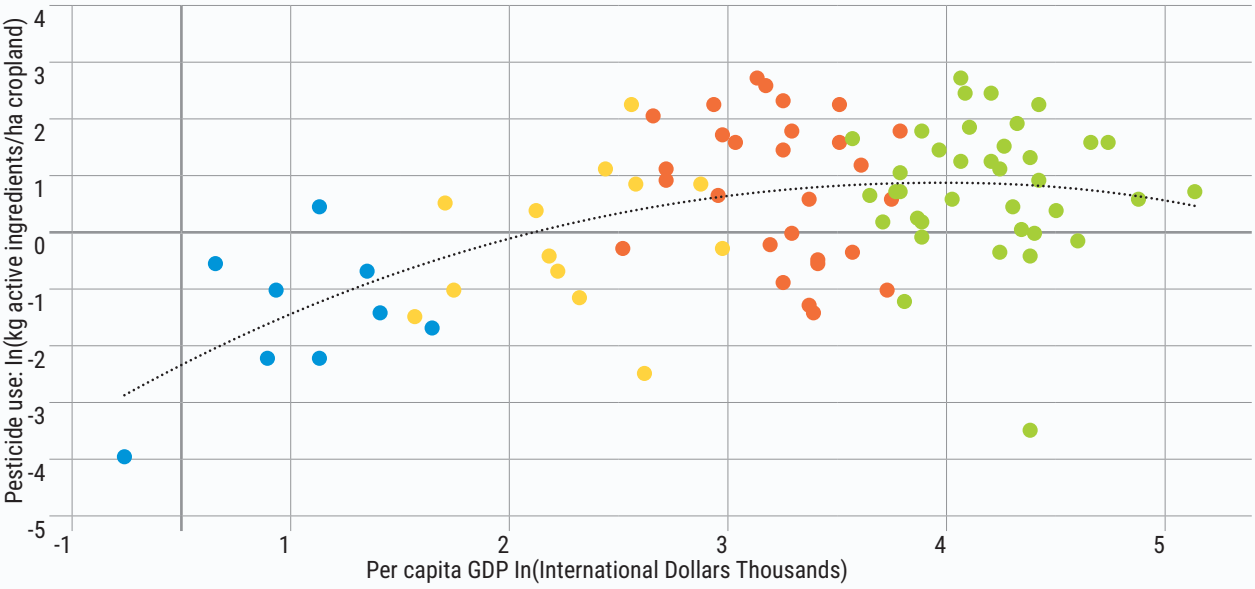
Based on a similar approach, pesticide use intensity was assessed for the most recent period on which data were available in FAOSTAT (2019a). Pesticide use per ha cropland increased with a country's per capita GDP and then showed a slightly decreasing trend for high income countries (Figure 2.4-19a). This may be explained by the fact that in certain high income countries, especially in Europe, there have recently been significant reductions in pesticide sales (Eurostat 2018), partly as a result of specific policies to reduce reliance on pesticides in agriculture (Chapter 3.4). Another explanation for apparent reductions in use intensity in high income countries is that they may be introducing new active ingredients, with lower application rates, more rapidly than lower income countries.

Similarly, pesticide use per unit crop output increases with per capita GDP and levels off in high income countries, but does not show a clear reduction (Figure 2.4-19b). More pesticides are thus applied per unit of crop production in richer than in poorer countries. It has been suggested that pesticide use intensity will decrease in high income countries, where pest management is most effective and environmental and health measures limit the use of pest control products. Somewhat surprisingly, this does not appear to be the case, nor does the situation seem to have changed since the assessment by Schreinemachers and Tipraqsa (2012).

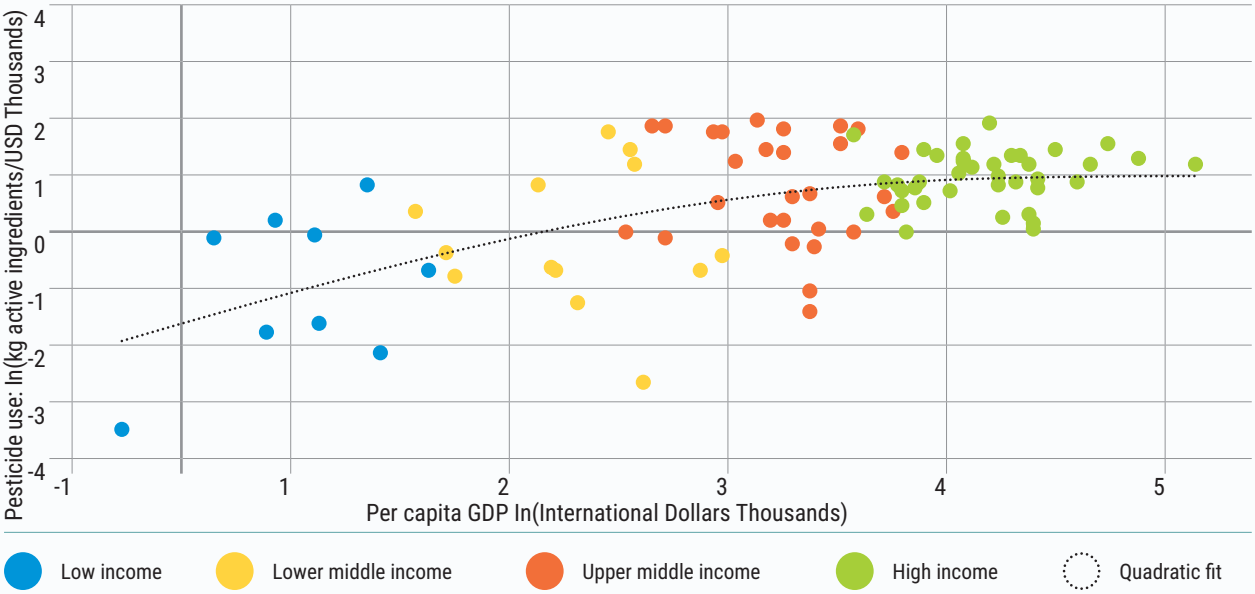
It is not entirely clear why higher income countries should apply more pesticides per unit crop output than lower income ones. This might partly be explained by relatively higher commodity prices, which are known to stimulate pesticide use (Chapter 2.7.10). Agricultural production in high income countries may also be more intense (e.g., larger areas of monocropping, limited genetic variety in crops), which can increase pest pressure and subsequent pesticide use. A longer history of relatively high levels of pesticide use may also increase pest resistance and resurgence, leading to increased use of pesticides (Chapter 4.4.3). However, this analysis seems to suggest that, contrary to expectation, the effectiveness of pesticide use in increasing or maintaining crop outputs in higher income countries is lower than in lower income countries.

Figure 2.4.19 Pesticide use intensity correlates positively with the level of a country's economic development. FAOSTAT (2019a).

A. Pesticide use intensity per area cropland increases with higher per capita GDP and slightly decreases again in high income countries.



B. Pesticide use intensity per unit crop output increases with higher per capita GDP, but does not decrease in high income countries.



Pesticide use, areas of cropland and crop output are annual average values for the period 2014-2016 in FAOSTAT (FAO 2019a); pesticide use intensity per surface area is expressed as kg a.i./ha cropland; pesticide use intensity per unit crop output is expressed as kg a.i./1,000 international dollars of crop output; per capita GDP (in current International dollars) is for 2015 (World Bank 2020b).

This assessment is global and will very likely be different in specific cropping systems. Trends in pesticide use intensity and pesticides' associated effectiveness in specific crops or regions merit further research.

2.5 Pesticide distribution mechanisms

2.5.1 Organization of the pesticide industry

The pesticide industry has traditionally been divided into two main groups of companies, those developing and marketing new pesticide active ingredients (research and development [R&D] companies) and those marketing generic off-patent or post-patent pesticides ("generics companies"). R&D companies invest a considerable part of their profits in discovering and developing new molecules and products; generics companies may develop new products, but on the basis of off-patent active ingredients.

Currently that dichotomy is no longer as strict as in the past. R&D companies attempt to maintain off-patent pesticides as proprietary products by developing new innovative formulations or introducing new uses. The combination of pesticides, seeds and biotech crops in the same company, which has been a trend during the last 10-15 years, has opened up new market possibilities for R&D companies. More recently, small biocontrol companies are being acquired by the larger agrochemical firms. Furthermore, some of the main R&D companies have acquired producers of generics with the aim of servicing a broader market.

At the same time, generics companies are investing in developing new formulations and some have entered the field of identification and development of new or modified active ingredients.

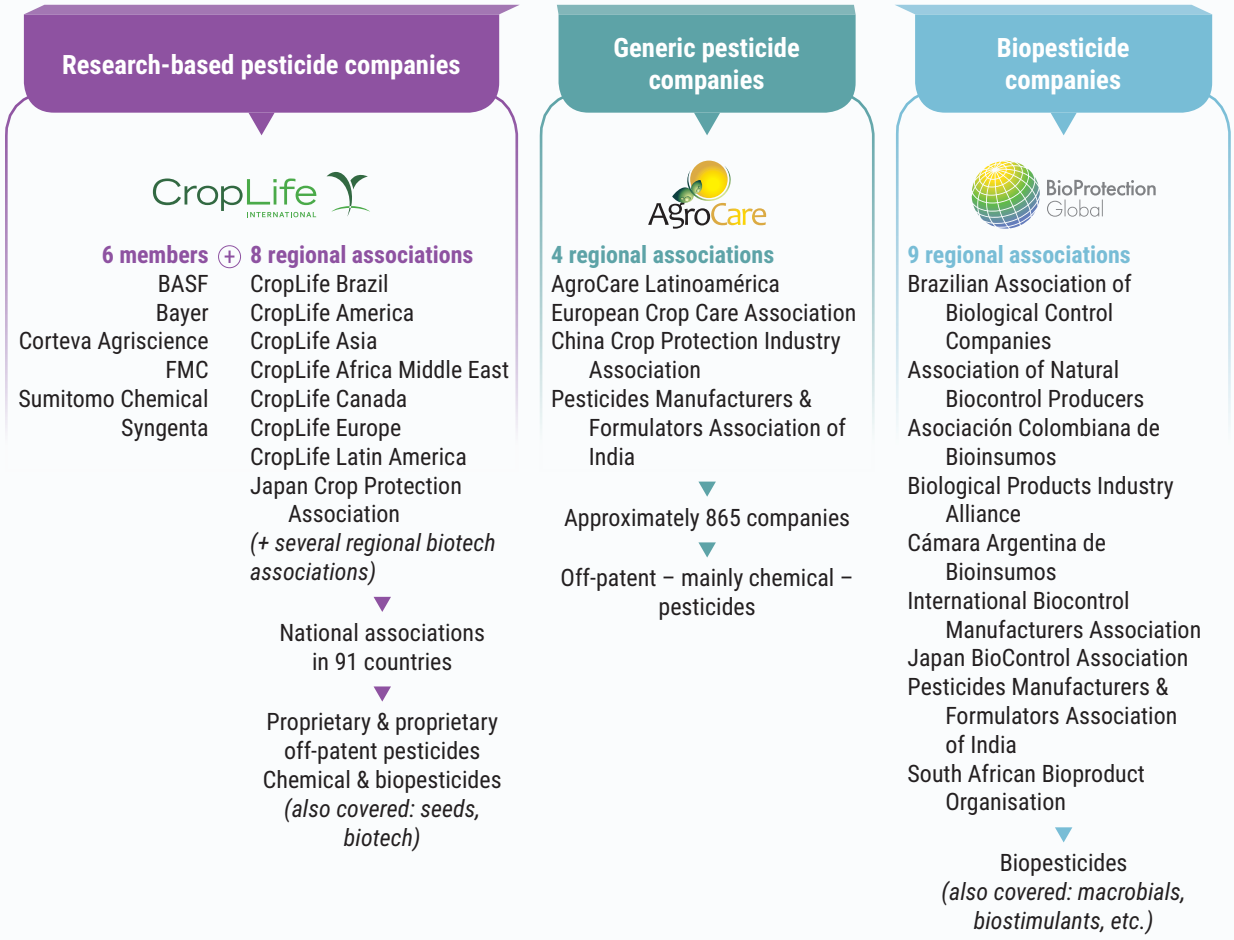
The broader plant science industry (including pesticides, biotechnology, seeds and biocontrol) has been much in flux over the last decade. Several important and many more smaller mergers and acquisitions have taken place. For instance, Bayer CropScience acquired Monsanto; United Phosphorus Ltd (UPL) acquired what used to be Arysta LifeScience; ChemChina acquired both Syngenta and ADAMA (and will be merging its agricultural assets with Sinochem); and Dow AgroSciences merged with DuPont to become Corteva Agriscience. (See Phillips McDougal 2019b for a graphical representation of key agrochemical mergers and acquisitions.)

This activity has led to a concentration of pesticide manufacturing and marketing. Today the global pesticide market is dominated by four major conglomerates or companies: ChemChina (with subsidiaries Syngenta and ADAMA), Bayer Crop Science, BASF and Corteva Agriscience, which together represent about 60 per cent of the agricultural pesticide market (Table 2.5-1).

Table 2.5-1 Top 10 pesticide companies and their share of the total agricultural pesticide market in 2018.
Yuan (2019).

Company	Agricultural pesticide sales in 2018	Share of total agricultural pesticide market in 2018
Syngenta	9,909	17%
Bayer CropScience	9,641	17%
BASF	6,916	12%
Corteva AgriScience	6,445	11%
FMC	4,285	7.3%
ADAMA	3,617	6.3%
UPL	2,741	4.7%
Sumitomo Chemical	2,538	4.3%
Nufarm	2,332	4.0%
Huapont Life Sciences	935	1.7%

Figure 2.5-1 The pesticide industry collaborates in three major international associations representing R&D companies, producers of generic pesticides and biopesticide companies

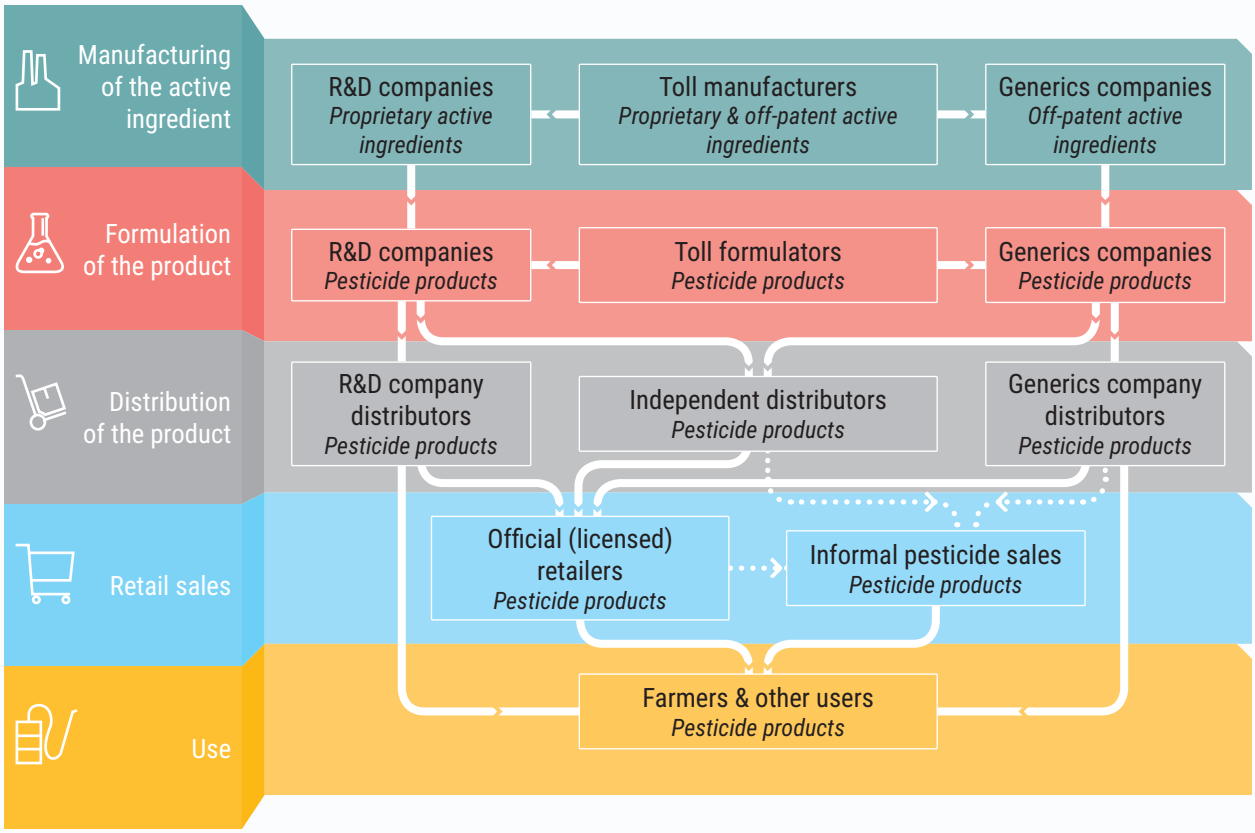


In addition, the integration of pesticide, seed and biotech activities in the same companies has allowed the plant science industry to better develop and control combined agricultural inputs. This includes seeds and seed coatings, seeds and genetically modified (GM) traits, and GM traits and pesticides.

Proponents of mergers have argued that companies need to operate on a larger scale in order to invest in and support research, and that these mergers – by creating more balanced portfolios of seed and chemical businesses – would incite greater combined seed and chemical innovations. Opponents have pointed out that, with less competition, it is in the interests of combined firms to raise product prices. The resulting companies may also be less likely to invest in research and innovation once the degree of competition is less (MacDonald 2019). An important risk associated with mergers is further loss of crop diversity (an already ongoing process) and the associated increased susceptibility of crops to insect pests and diseases (FAO 2019b).

At global and regional levels, pesticide companies are collaborating in a number of industry associations (Figure 2.5-1). The mainly R&D companies collaborate in CropLife International, which consists of a number of regional associations and six major companies. A large number of generic pesticide companies are united under AgroCare, which is currently made up of four regional associations. Companies active in biological pest control are represented by BioProtection Global, which covers many types of bioprotectants including natural pest enemies and biopesticides.

Figure 2.5-2 Simplified pesticides supply chain, where pesticide products may originate from different sources.



2.5.2 Pesticide supply chains

A variety of pesticide supply chains exist. These supply chains often operate in parallel in a given country or region (Figure 2.5-2).

Pesticide technical materials (active ingredients) will be manufactured by R&D companies themselves, especially in the case of proprietary active ingredients still under patent. In some cases such materials are produced under contract by independent toll manufacturers, which are frequently specialized in particular chemistries. Off-patent technical materials can be manufactured by generic companies, R&D companies or toll manufacturers.

Commercial pesticide products are formulated by plants directly under the control of R&D or generic companies, or by specialized toll formulators. While technical materials for a given pesticide tend to be produced at a limited number of manufacturing locations, formulation plants may be more decentralized in specific parts of the world. In some countries relatively small formulators may also produce for the local market.

Pesticide distribution companies (national or international) ensure the importation and distribution of commercial pesticide products in a country or region. Some of these companies may be under the management of the pesticide manufacturers, but most are independent, distributing pesticide products under licence from one or more manufacturers or formulators. Distributors may sell pesticide products directly to (larger) pesticide users or governmental and non-governmental organizations, as well as supplying pesticide retailers.

Pesticides are sold to users by a variety of retail outlets, ranging from shops or dealers specialized in agricultural inputs, to supermarkets, household, garden and do-it-yourself retailers, general retail outlets, markets or travelling pesticide sellers. In many countries retail outlets need to be licensed to sell pesticides, while pesticide sales

in supermarkets and other more general retailers tend to be limited to low-risk household and garden products. However, this is not always the case and more hazardous pesticides may be sold in general retail outlets. Recently, direct sales from manufacturers to local dealers or to end users appear to have become increasingly common in an attempt to better control the supply chain.

In some (mainly high income) countries restricted use pesticides (i.e., those which pose high risks) are sold only to professional pest control operators licensed to apply them. These operators generally carry liability protection for possible crop damage or other adverse effects. Such liability procedures are much less common in many low and middle income countries where, if they exist, they are rarely used effectively.

Pesticide sales in markets or by street sellers and travelling pesticide sellers are common, especially in low and lower-middle income countries, often constituting an informal, unregulated supply chain by means of which pesticides may be illegally repackaged into small quantities, not properly labelled, and with the quality unclear. Nevertheless, the informal supply chain is frequently a very important source of pesticides for farmers as well as for households in urban areas. This is because specialized retailers may not be present, especially in rural areas.

Sales of biopesticides can follow the above model, especially if products are relatively broad spectrum and can be stored easily. However, given the target specificity of many biopesticides, as well as specific storage requirements, they are often sold directly by the manufacturer or distributor to end users, which tend to be larger agricultural producers.

2.5.3 Illegal pesticides

The growth of global pesticide markets (Chapter 2.4) has been accompanied by an important increase in the trade of illegal or illicit pesticides. While illegal pesticides were hardly an issue in the late 1990s, they now form a significant fraction of pest control products sold (Food Chain Evaluation Consortium [FCEC] 2015).

Illegal pesticides can take many forms: they may never have been authorized for use in a given country; their legal use may have been cancelled or banned; they may be legal for use on one crop but not on another; or they may be counterfeit, fake, or illegitimately or illegally relabelled or (re) packaged (Box 2.5-1).

These illegal pesticides can damage crops, contaminate water and soil, and harm human health. They may erode public confidence in and perceptions of food safety, as well as confidence

Box 2.5-1 Illegal pesticides can take different forms. OECD (2019), UNICRI (2016).

Illegal pesticide	Any pesticide which, for whatever reason, is not legal in the country of destination. This includes the sub categories of counterfeits, fakes, obsolete and unauthorised pesticides.
Unauthorised pesticide	A pesticide that is not authorised for use by the regulatory authorities in the country in which it is being placed on the market.
Counterfeit pesticide	An illegal copy of a legitimate, branded pesticide which may be difficult to distinguish from the legal product due to the high quality branding and packaging.
Fake pesticide	An illegal copy of a legitimate, branded pesticide which may make some effort to imitate the original product but which can be identified with relative ease due to the poor quality of the product and packaging
Illegally (re-)packaged pesticide	A pesticide sold in an illegally (re-)filled pesticide container or a non-approved type of packaging, such as food or beverage containers.

Table 2.5-2 Recent reports on the scale of the trade in illegal pesticides.

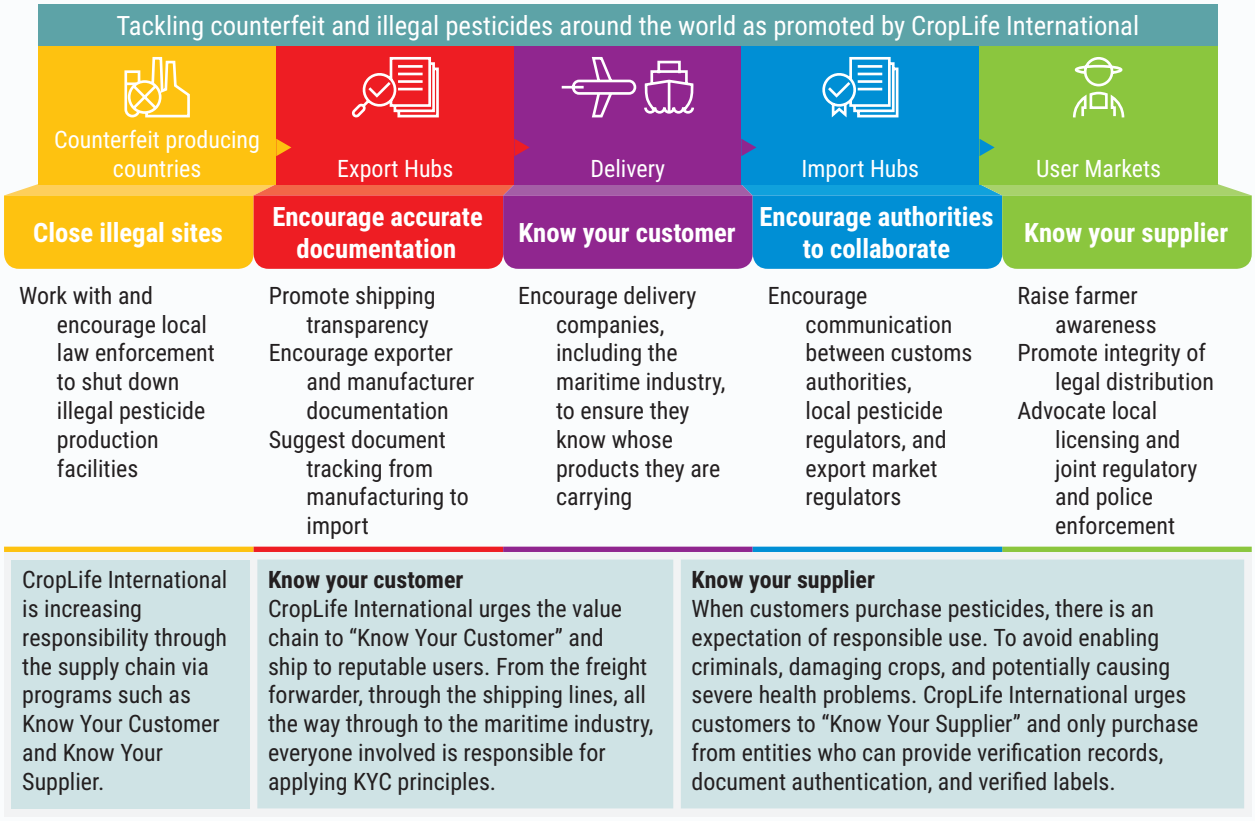
Region / country	Findings	Source
Global	Estimates of global trade in illegal and counterfeit pesticides range from 5 to 15 per cent of the total pesticide market.	UNICRI (2016)
Brazil	In 2015/2016 illegal pesticides represented about 24 per cent of the total crop protection product market.	Instituto de Desenvolvimento Econômico e Social de Fronteiras (IDESF) (2019)
Mali	45 per cent of glyphosate products from national retailers sampled were unregistered or counterfeit.	Haggblade *et al*. (2019)
Africa	Most national pesticide industry associations reported that counterfeit or illegal pesticides represented about 15-20 per cent of total markets.	Guyer and Davreux (2012)
EU	Illegal plant protection products (PPP) represented approximately 10 per cent of the EU's PPP market in 2014, compared with about 7.5 per cent in 2008.	FCEC (2015)
EU	It is estimated that the legitimate industry loses approximately EUR 1.3 billion in revenue annually due to the presence of counterfeit pesticides in the EU marketplace, corresponding to 13.8 per cent of sales in this sector.	European Union Intellectual Property Office (EUIPO) (2017)
	If knock-on effects on other industries and on government revenue are added (and both direct and indirect effects are considered), counterfeiting in this sector causes approximately EUR 2.8 billion in lost sales to the EU economy, leading in turn to the loss of about 11,700 jobs and of EUR 238 million in government revenues.	
India	Non-genuine and illegal pesticides represented approximately 25 per cent by value and 30 per cent by volume of domestic pesticide industry in 2013.	Federation of Indian Chambers of Commerce and Industry (FICCI) (2015)
	Non-genuine and illegal pesticides included products not registered in India, pesticides with low/incorrect active ingredients, products containing substances banned in India, counterfeits, and products laced with chemicals allegedly sold as biopesticides.	
Ukraine	Share of illegal pesticides represents 19 to 25 per cent of the national pesticides market	UNEP and GRID-Arendal (2020)

in reputable agricultural producers and producing countries. They may also cause reputational damage to established pesticide manufacturers and distributors or undermine the authority of national regulators. (UNICRI 2016; United Nations Environment Programme [UNEP] and GRID-Arendal 2020; European Crop Protection Association n.d.).

Owing to its nature, the importance of the trade in illegal pesticides is difficult to quantify (UNEP and GRID-Arendal 2020). Various recent estimates indicate that, depending on the region or country, illegal pesticides represent between 10 and 25 per cent of national pesticide markets, with even higher estimates for selected countries (Table 2.5-2). Based on the global UNICRI estimate of 10-15 per cent illegal pesticides, and a total market for legal pesticides of USD 65 billion in 2018 (Chapter 2.4.3), the value of trade in illegal pesticides would amount to USD 7-11 billion annually.

National enforcement authorities are increasingly clamping down on trade in illegal pesticides. Recent examples are the Silver Axe operations which seized over 550 tons of illegal pesticides throughout Europe in 2019 (European Union Agency for Law Enforcement Cooperation [Europol] 2019); anti-counterfeiting operations in Indonesia (Tribun Jateng 2020);

Figure 2.5-3 Countering the illegal trade in pesticides should address all stages of the pesticide distribution chain to be effective. CropLife International (CLI) (n.d.).



and the confiscation of illegal pesticides worth USD 1.5 million in Brazil in December 2019 (Betancur 2020). Competent authorities' level of awareness of illegal trade in pesticides, together with the resources made available for addressing it, appear to be key factors determining the success of initiatives in this area (FCEC 2015).

The legitimate pesticide industry has been very active in creating awareness of illegal trade in pesticides, with activities along the entire pesticide distribution chain. These activities include training programmes for enforcement agencies, "know your customer" approaches for pesticide manufacturers and distributors, and "know your supplier" programmes for pesticide users (Figure 2.5-3). A study recently commissioned by CropLife International (CLI) resulted in concrete recommendations for improved supply chain traceability of agrochemicals (Accenture 2019).

UNEP and GRID-Arendal (2020) have recommended – at national and regional level – improving the monitoring and understanding of the pesticide supply chain (e.g., national reporting of chemical movements from source to end use and disposal); more cross-border cooperation among national authorities regulating and controlling the trade in pesticides; and the development of joint regional action plans to fight illegal trade in chemicals and waste. At the global level they urge stronger coordination among United Nations agencies and others involved in preventing illegal trade in chemicals.

At the international level, the Organisation for Economic Co-operation and Development (OECD) has established the Network on Illegal Trade of Pesticides (ONIP), in which governments and the private sector collaborate, and a Rapid Alert System (RAS) for suspected illegal international trade in pesticides. In 2019 the OECD Council issued a Recommendation on Countering the Illegal Trade of Pesticides, accompanied by a best practice guidance document (OECD 2019).

2.6 Pesticide application technology

2.6.1 Application techniques

Application equipment and methods affect both environmental and human exposure to a pesticide. Using the right equipment in the right way goes a long way towards minimizing exposure and the associated risks. On the other hand, using equipment which is of bad quality, faulty or inappropriate – or applying pesticides under improper environmental conditions – tend to greatly increase exposure and therefore environmental and human health risks.

Most pesticides are applied in the form of sprays, using equipment such as boom sprayers, orchard sprayers, foggers, spray aircraft, backpack sprayers, and other manually carried sprayers. In solid form pesticides may be applied as granular formulations (e.g., to the soil), baits (e.g., for rodent control) or dusts. Fumigation is carried out using pesticides in gaseous form. For seed treatments, crop seeds are coated with a pesticide before planting to create a protective zone of active ingredient in the soil against soil-borne pathogens and insects, while systemic seed treatments also provide additional protection against early-season foliar diseases and insects (Nuyttens *et al*. 2013).

In many instances the plant foliage is the intended target for sprayed herbicides, fungicides and insecticides. Sometimes the soil is targeted (e.g., for pre-emergence herbicides) or insects may be targeted directly (e.g., for locust swarm control). During spray applications only a fraction of the applied pesticide reaches the intended target, generally the leaf canopy. As most pesticides are applied using hydraulic nozzles that produce a spray containing droplets that vary in size, the smallest droplets can be carried by air currents/wind and drift outside the target area while the largest may be deposited on the soil. Additional losses occur when pesticides deposited on the target are washed off by rain or volatize (if the pesticide is relatively volatile).

In the case of pesticides intentionally applied to soil, a large fraction may run off during rainstorms or drain into the soil. When treated seeds are drilled into the soil, the pesticide can be abraded and lost as dust drift (Nuyttens *et al*. 2013; Jensen and Olesen 2014). Recently seed coatings have been improved and drilling equipment adapted to reduce the risk of dust drift. The share of the applied dose that does not reach the intended target is a loss to the farmer and a potential source of environmental pollution.

Pesticide losses during and after application are extremely variable, depending on crop cover, application method, type of nozzle, pesticide formulation and environmental conditions. Jensen and Olesen (2014) reviewed pesticide spray mass balances for vehicle-mounted sprayers. They found that on average 66 per cent of the applied pesticide reached the foliage when applied by boom sprayers, 46 per cent when applied by orchard sprayers and 55 per cent when applied by tunnel sprayers, all crops and growth stages combined. The rest of the sprayed product was deposited on soil, drifted outside plots or was not accounted for.

2.6.2 Reducing environmental and occupational exposure

Continuous progress is being made in the development of more precise pesticide application technology and engineering controls to reduce environmental and occupational exposure. However, when compared with the development of new pesticide products, application technology has received relatively little attention from the private sector or from governments (International Pesticide Application Research Consortium [IPARC] 2020; Matthews 2020).

Occupational exposure has been minimized though technologies such as enclosed and ventilated tractor cabins, low-level induction bowls to facilitate placing the pesticide product in the tank, triple-rinse pesticide containers, closed pesticide transfer systems, and modular-mix-on-demand (MMOD) systems whereby the pesticide is mixed on demand from concentrate to avoid pre-mixing or disposal of

unused chemicals (Matthews, Bateman and Miller 2014).

Environmental exposure, especially through pesticide drift, has been reduced by using specific application technologies. Air induction nozzles have been introduced to minimize the proportion of small droplets to reduce drift, while rotary atomizers provide more uniform droplet size. Other technologies that have been introduced include spray boom shields or shrouds, self-levelling spray booms to maintain the right distance above the crop canopy and minimize drift, and orchard sprayers using sensor systems to detect gaps in the tree canopy and reduce spray delivery when there is no crop to intercept it (Matthews, Bateman and Miller 2014; IPARC 2020).

A reduction in environmental exposure to applied pesticides can also be realized through mitigation measures not directly related to the equipment. In many countries buffer zones are required in order to minimize drift to surface waters, neighbouring fields or human dwellings. They can be strips of unsprayed crops adjacent to field boundaries or vegetated areas which partly intercept spray drift (Matthews, Bateman and Miller 2014). Buffer zones tend not to be very popular with farmers, particularly when relatively wide zones are involved, as cropland is lost and such zones are a potential source of pests, diseases and weeds. Buffer zones are virtually impossible to implement in regions with many small adjacent crop fields, for example paddy rice or horticultural systems, where they would take up an unrealistically large fraction of individual fields.

2.6.3 Improving the precision of pesticide applications

Since much pesticide is lost during application, increasing the precision and efficacy of pesticide application equipment and techniques has received particular attention.

Global positioning system (GPS) equipment has been used in spray aircraft for some time. More recently, it has been built into tractors. Combining GPS with in-cab controls (e.g., to adapt flow rates) and a geographic information system (GIS) that contains cartographic information about a crop or pest provides detailed information to the driver and enables individual applications to be recorded (Matthews, Bateman and Miller 2014). On-off switching sprayers and canopy-optimized distribution sprayers, using 3D sensing systems able to detect the shape and volume of the sprayed canopy, allow more precise application and minimize losses and drift (Tona, Calcante and Oberti 2018).

Unmanned aerial vehicles (UAVs), also known as drones or remotely piloted aircraft (RPA), are increasingly used in some parts of the world (e.g., Brazil, China, Japan and North America). Originally, UAVs were primarily employed to monitor the presence of weeds, pests and diseases in crops through a variety of sensors. More recently, drone-based systems have also been developed for aerial pesticide applications (Iost Filho *et al*. 2020).

UAVs are useful, in particular, for spraying relatively small fields or complex terrains that are not easily accessible by personnel or large machinery. They are also replacing back-pack spraying in some countries (He 2018; Carvalho *et al*. 2020). Spraying with UAVs has the advantage that the airflow which keeps the drone airborne can blow the spray into the crop canopy. Furthermore, UAVs operate at a much slower speed than traditional spray aircraft, so that they can be used closer to the crop, reducing the risk of drift, especially when rotary nozzles are used (Matthews, G.A., personal communication).

Further studies are needed to provide data on droplet deposition, spray coverage and drift when spraying with UAVs. The next step for agricultural UAVs is to use them in an integrated system for pest control comprising complementary components: one for remote sensing, detection and mapping of weed, pest or disease infestations and another with precision spraying capability (Hunter *et al*. 2020).

2.6.4 Pesticide formulation

Pesticide active ingredients are generally not applied in their pure form, but are formulated into a commercial product. The main objective of formulation is to ensure that the active ingredient

remains stable and effective from manufacture through application to its final target (Bullock 2020a). This includes the following aspects:

- product stability during manufacture, packaging, storage and application, as well as after it has been applied;
- compatibility with the application equipment and process, as well as with other products and diluents;
- effective delivery to the target, e.g., by reducing volatilization, improving wetting and adhesion, aiding penetration and uptake of the active ingredient, and improving rain fastness.

In addition, formulation technology can reduce (and sometimes even eliminate) exposure of the pesticide user and bystanders and reduce exposure of non-target organisms and the environment, e.g., by reducing spray drift (Bullock 2020a).

In the past, most formulations were based on simple solutions in water (SL), emulsifiable concentrates (EC) in petroleum-based solvents, or dusts (DP) and wettable powders (WP). The presence of solvents in EC formulations, and fine dusts in DP and WP formulations, may cause occupational health risks during use and adverse effects on the environment (Knowles 2008). Therefore, newer formulations have been developed which are safer for the user, have less impact on the environment, and can be applied at minimum effective dose rates.

Examples of more recent formulation types which generally present lower environmental or health risks are (Knowles 2008; Bullock 2020a):

- water dispersible granules (or dry flowables) – as a replacement for WP – because they are non-dusty, free-flowing granules which disperse quickly when added to water in the spray tank;
- water-soluble packs or sachets which can be added directly to a spray tank, thus minimizing exposure of the user and pack disposal problems;
- microencapsulated controlled release formulations, which may reduce mammalian and fish toxicity, lower application rates, and reduce leaching to groundwater and surface water;
- oil-in-water emulsions – as a replacement for EC – as a way to reduce or eliminate volatile organic solvents and reduce handling risks.

Use of nanopesticides is a novel approach to formulation which has been the topic of recent research and patent activity. Nanopesticides are generally classified into two types (Kookana *et al*. 2014; Li *et al*. 2019):

- very small particles of a (generally inorganic) pesticide active ingredient, such as nanometals (e.g., silver, copper) or nanoclays, which are biologically active against a disease or pest;
- engineered (often organic) nanocarrier particles (e.g., polymers, solid lipids) which contain a pesticide active ingredient – the nanocarrier may be designed, for instance, to protect the active ingredient or enhance its delivery to the pest or disease.

Nanopesticides are an emerging technology that potentially offers a range of benefits, including increased efficacy and/or a reduced amount of the active ingredient to be applied (e.g., through improved solubility/dispersibility, controlled release, targeted delivery, enhanced bioavailability, increased leaf adhesion, or improved stability in the environment) (Kah *et al*. 2018). A reduction in the amount of pesticide active ingredient applied may reduce environmental and human health risks.

Since nanopesticides will have both benefits and risks for pest management, the environment and human health, careful and comprehensive assessments of these products need to be made before they are marketed (Kah 2015; Li *et al*. 2019). Important data gaps continue to exist with regard to the interactions and behaviours of nanomaterials in the human body, methods to determine such interactions and behaviours, and the relevance of these types of data for risk assessment (FAO and WHO 2013). If nanopesticides become "emerging

contaminants" rather than "emerging solutions", potential useful tools for pest control may be lost (Kah 2015).

While newer formulations have increasingly come on the market, they have by no means replaced the more conventional formulation types such as EC, WP or granular (GR), which, in 2016-2018, represented two-thirds of the total number of products on the market and more than 50 per cent of total market value (Bullock 2020b). So far, commercial adoption of recent innovations in pesticide formulation has been relatively modest. Nevertheless, because the market is expected to continue its drive towards safer, environmentally acceptable and economically efficient solutions, the demand for novel formulations can be expected to grow (Bullock 2020a; Bullock 2020b).

2.6.5 Regional differences in pesticide application innovations

While improvements in pesticide application and formulation technology over the last few decades have led to more effective and safer pesticide application, innovations have not been evenly applied across the world.

There is a striking difference in the availability and use of modern pesticide application technology between countries with large-scale industrialized agriculture and those where small-scale subsistence farming predominates. In the former, tractor-mounted sprayers and aircraft tend to be used, often with modern technology. In most low and lower-middle income countries, knapsack sprayers and other types of relatively simple hand-carried pesticide application equipment prevail, for which many of the innovations discussed above do not apply (Matthews, Bateman and Miller 2014; Horne 2019).

Hand-held sprayers have a relatively high risk of operator exposure to pesticides when the sprayer is filled and when the pesticide is applied (Matthews 2020). Since engineering controls to reduce exposure have primarily been developed for modern tractor-mounted sprayers, users of knapsack sprayers are particularly dependent on the use of appropriate personal protective equipment (PPE) to reduce the risk of exposure. However, such PPE is often not accessible for smallholder farmers (Chapter 4.4.5). In addition, training in the correct use of hand-held sprayers is of great importance even if the equipment is relatively simple.

The Food and Agriculture Organization of the United Nations (FAO) has published guidelines on procedures for the registration, certification and testing of new pesticide application equipment (FAO 2020a), and the International Organization for Standardization (ISO) has published standards for testing and equipment (Matthews, Bateman and Miller 2014). However, quality control and certification of pesticide application equipment is legally required and/or systematically conducted in only a few countries.

Innovations in pesticide formulation have mainly been marketed in high and upper-middle income countries, where more expensive specialized formulations are being used. Conventional formulation types, often posing higher occupational risks, dominate the markets in low and lower-middle income countries, probably since many of them are cheaper generic products (Bullock 2020a).

2.7 Drivers of pesticide use

Global pesticide use, and pesticide use intensity, have increased considerably during the last decades (Chapter 2.4). Many drivers influence the use of pesticides in agriculture, as well as public health and domestic uses. Some drivers tend to increase the use of pesticides while others may lower it (Box 2.7-1). To identify future policies to reduce the risks of pesticides, knowledge of the main drivers influencing pesticide use is essential. Therefore, this chapter discusses important current drivers of pesticide use and how they can

Box 2.7-1 Key current drivers of pesticide use. The predominant ways in which these drivers tend to influence pesticide use are presented. However, it is recognized that such influences may be more subtle, as is discussed in the referenced chapters.

Driver	Tends to increase use	Tends to lower use	Chapter
Agronomic			
Agricultural production growth	All pesticides		2.7.1
Crop losses	All pesticides		2.7.2
Agricultural intensification	All pesticides		2.7.3
Pesticide resistance	All pesticides		4.4.3
Genetically modified crops	Mainly herbicides	Mainly insecticides	2.7.4
Integrated pest and vector management	Biopesticides	Synthetic pesticides	2.7.5
Organic production	Biopesticides	Synthetic pesticides	2.7.6
Regulatory			
Pesticide legislation and policy	Low risk pesticides	High risk pesticides	2.7.7; Chapter 3
Health and environmental policy and legislation	Low risk pesticides	High risk pesticides	2.7.7; Chapter 3
Economic			
Pesticide marketing	All pesticides		2.7.8
Pesticide prices	If low: all pesticides		2.7.9
Commodity prices	If high: all pesticides	If low: all pesticides	2.7.10
Fiscal policies	If tax breaks or subsidies: all implicated pesticides	If taxes: all implicated pesticides	2.7.11
Voluntary sustainability standards	Biopesticides	Synthetic pesticides	2.7.12
Public health			
Food safety		Synthetic pesticides	2.7.13
Public concerns about health		Synthetic pesticides	2.7.14
Vector-borne diseases	Public health pesticides		2.7.15
Environmental			
Climate change	All pesticides		2.7.16
Invasive species and pest outbreaks	All pesticides		2.7.17
Public environmental concerns		Synthetic pesticides	2.7.18
Information			
Information sources	Depends on type of source	Depends on type of source	2.7.19
Knowledge, awareness and attitudes		Limited impact	2.7.20
Training		Limited impact	2.7.21

affect the extent and manner in which pesticides are applied.

The drivers of pesticide use are described here as if they were generally applicable everywhere in the world and in all crops or pesticide use situations. However, the extent to which individual drivers influence pesticide use is greatly influenced by local agronomic, regulatory, economic and social conditions.

In some cases drivers are mentioned only briefly in this chapter, but are reviewed in more detail elsewhere in the report.

Agronomic drivers

2.7.1 Growth in agricultural production

Global agricultural production is projected to increase by around 20 per cent between 2018 and 2027, with considerable variation across regions (Figure 2.7-1). Strong growth is expected in sub-Saharan Africa, South and East Asia, and the Middle East and North Africa. By contrast, production growth in industrialized countries is expected to be much lower, especially in Western Europe, where agricultural and fish production are projected to increase by only around 3 per cent during this period (Organisation for Economic Co-operation and Development and Food and Agriculture Organization of the United Nations [OECD/FAO] 2018).

Owing to population growth and higher per capita income, food consumption will continue to expand with regard to most commodities. Low and middle income countries will be the source of most demand growth in the next ten years, with sub-Saharan Africa and India accounting for a large share of additional food demand for cereals (OECD/FAO 2018).

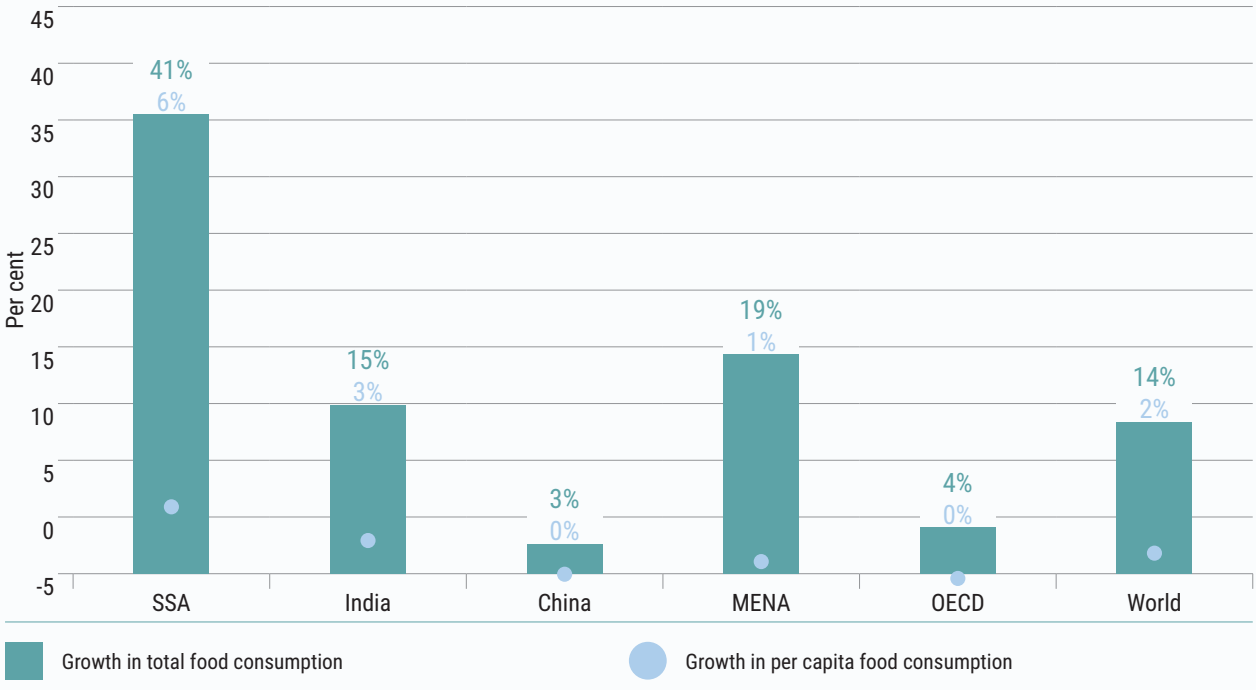
Growth in production will be achieved primarily by (preferably sustainable) intensification and efficiency gains, and partially by enlargement of the production base through herd expansion and conversion of pasture to cropland (OECD/FAO 2018).

In the past, growth in agricultural production has been accompanied by increased use of pesticides. Without changes in environmental and agricultural development policies, as well as pest management practices, it is expected that pesticide use will continue to grow (McIntyre *et al*. 2009).

2.7.2 Crop and post-harvest losses due to weeds, pests and diseases

The main rationale for using pesticides in agriculture is to minimize economic losses. Pests, diseases and weeds may lead to crop injuries which result in crop damage or losses that cause economic losses. However, these

Figure 2.7-1 Growth rates of total and per capita food consumption, 2018-2027. OECD and FAO (2011).



Note: SSA = Sub-Saharan Africa; MENA = Middle East and North Africa; OECD = Organisation for Economic Co-operation and Development countries

Table 2.7-1 Estimates of global yield losses in major crops caused by pests, diseases and weeds.

	Average actual yield loss as % of attainable yield (global range)				
Crop	Weeds	Animal pests*	Diseases	Total	Source
Wheat	**7.7** (3 – 13)	**7.9** (5 – 10)	**12.6** (7 – 16)	**28.2** (14 – 40)	1
	not assessed	**2.0** (0 – 4)	**19.5** (10 – 24)	--	2
Maize	**10.5** (5 – 19)	**9.6** (6 – 19)	**11.2** (6 – 20)	**31.2** (18 – 58)	1
	not assessed	**4.7** (4 – 11)	**17.7** (14 – 37)	--	2
Rice	**10.2** (6 – 16)	**15.1** (7 – 18)	**12.2** (8 – 19)	**37.4** (22 – 51)	1
	not assessed	**9.1** (6 – 12)	**21.0** (20 – 30)		2
Potato	**8.3** (4 – 14)	**10.9** (7 – 13)	**21.1** (12 – 33)	**40.3** (24 – 59)	1
	not assessed	**2.8** (1 – 4)	**14.5** (6 – 19)		2
Soybean	**7.5** (5 – 16)	**8.8** (3 – 16)	**10.1** (3 – 18)	**26.3** (11 – 49)	1
	not assessed	**5.7** (2 – 10)	**15.6** (9 – 24)		2
Cotton	**8.6** (3 – 13)	**12.3** (5 – 22)	**7.9** (5 – 15)	**28.8** (12- 48)	1

* primarily insects, mites and nematodes.
1: Oerke 2006;
2: Savary *et al*. 2019 (only assessed losses by pests and diseases; loss percentages were disaggregated on the basis of Supplementary Table 3)

relationships are not linear or automatic: pest, disease or weed infestations do not always lead to measurable crop injuries, nor do injuries necessarily lead to crop losses, while crop losses do not always lead to economic losses (Savary *et al*. 2012) (Box 2.7-2). This makes assessments of economic losses due to crop pests, diseases and weeds complicated.

It would be logical to expect that estimates of crop losses and subsequent economic losses due to pests, diseases and weeds are based on thorough science, as these estimates form the basis for pest management decisions including use of pesticides. However, perhaps surprisingly, many projections of the costs and benefits of pesticide interventions lack corroborating data on actual field losses in the geographical areas concerned (Savary *et al*. 2012; Institut national de la recherche agronomique 2017; Savary *et al*. 2019) (see also Chapter 5).

Over the years many estimates have been made of crop losses due to Pests, diseases and weeds, and associated economic losses. Estimates of crop losses show high variability, depending on crop variety, geographical area, agronomic and environmental factors, the estimation methods used, and baselines selected for crop damage and yield. The most recent reviews, while using different methods, show similar overall crop losses of 20-40 per cent of attainable yields (Oerke 2006; Savary *et al*. 2019) (Table 2.7-1).

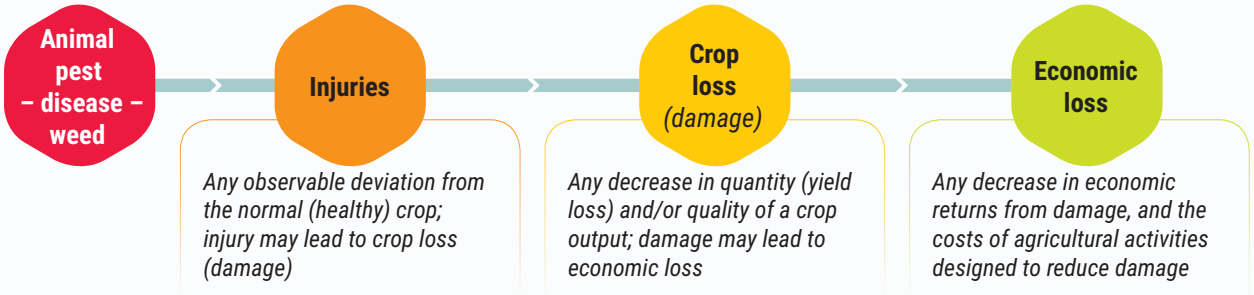
Very large regional differences exist, however, in crop losses across the world. High losses – relative to per capita production – have been found in sub-Saharan Africa and on the Indian subcontinent and for certain crops in China. Relatively lower losses are seen in some regions of high crop production, such as North America and southern South America (Savary *et al*. 2019). Crop losses appear to be more important in food insecure regions than in regions with production surpluses. Such regional variations are partly explained by environmental (climatic, social and economic) differences, but also by differences in the efficiency of crop health management practices, with regard to which scope for improvement may be indicated (Savary *et al*. 2019).

2.7.3 Agricultural intensification

Growth in agricultural production will, to a large extent, need to be achieved through the intensification of production processes (i.e., increasing production per unit land, preferably sustainably).

Box 2.7-2 Concepts of crop yield and crop loss. Savary *et al.* (2006); Savary *et al.* (2012); Esker, Savary and McRoberts (2013); Boote (2017)

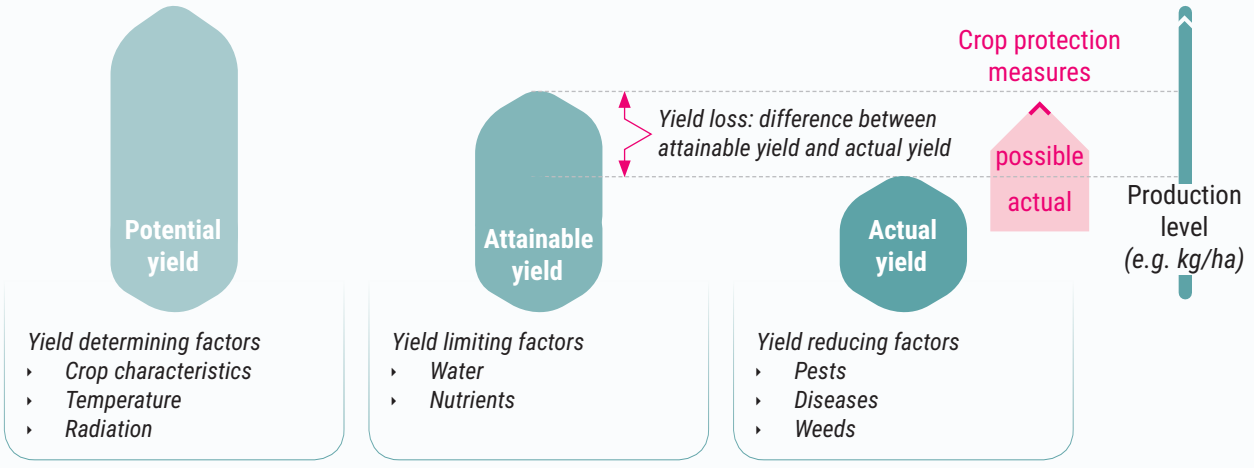
Pests, diseases and weeds may cause injuries to the plant, such as lesions on fruits or leaves, reduction of turgor, or effects on photosynthesis. Such injuries may lead to crop damage or crop loss, either decreased yield or adverse effects on the quality of the crop. The production of aflatoxins by fungal diseases is an example of an effect on crop quality. Whether or not injuries lead to crop loss depends on crop management, environmental conditions and crop protection measures and is often locally specific. Crop loss can result in economic loss for the farmer. Economic loss depends on the magnitude of the crop loss, but also on the costs of pest management measures and pesticides and the price elasticity of the crop.



The yield concepts below are generally used in crop loss assessments. The potential (or theoretical) yield of a crop is determined by the genetic makeup of the plant, as well as by temperature and radiation. Potential yield is achieved if there are no limitations of nutrients and water, or any injury by pests, diseases or weeds. The attainable yield is the potential yield, less any effect of water or nutrient shortages (or excesses) in the local production situation. The actual yield is the yield actually harvested; it incorporates the yield reducing effects on the attainable yield by pests, diseases, weeds (and possibly pollutants), as well as actual crop protection measures.

Yield loss by pests, diseases and weeds is defined as the difference between attainable yield and actual yield.

Crop protection measures aim to increase actual yields as much as possible towards attainable yields. Generally, crop protection measures will not fully achieve attainable yields, as this this is not economically affordable.



Intensification of crop production to meet increasing demand for food and fodder began in earnest in the 1950s in North America and Europe and expanded in the 1960s with the Green Revolution in Asia and Latin America. The production model focused initially on the introduction of improved, genetically uniform crops grown in large homogeneous areas with high levels of complementary inputs such as irrigation, fertilizers and pesticides. Green Revolution technologies are estimated to have fed more than a billion extra persons. The Green Revolution is credited, especially in Asia, with alleviating rural poverty, saving large areas of fragile land from conversion to extensive farming, and helping to reduce hunger (FAO 2011; Fresco 2016).

However, these large gains in agricultural production were often accompanied by negative effects on the environment, human health, and agriculture's natural resource base. For instance, pest and disease pressure increased due to monocultures of high-yielding but pest-sensitive crop varieties, reduction of fallow and crop rotation, and high use of fertilizers. This led to increased pesticide use, resulting in the development of pesticide resistance, the resurgence of existing pests, and upsurges of secondary pests – leading in turn to further repeated use of pesticides. A "pesticide treadmill" was the result (van den Bosch 1989).

Many of the adverse effects of this type of intensification have been mitigated, at least partly. Nevertheless, current agricultural intensification processes may still lead to large increases in the use of agricultural inputs and associated environmental and health impacts. For example, Riwthong *et al*. (2015) found that intensification of smallholder agriculture in northern Thailand resulted in higher productivity (expressed as gross margins at the farm gate) by up to a factor 13, but also increased pesticide use intensity by a factor 16.

It is widely recognized that meeting future demand for food will require different, more sustainable approaches. As stated by Godfray and Garnett (2014): Sustainability is a "must have" not a "nice to have". Further production optimization and intensification has been described in different ways, including sustainable crop production intensification, ecological intensification and agroecology, among others (FAO 2011; Bommarco, Kleijn and Potts 2013; Pretty 2018; Garibaldi *et al*. 2019; High Level Panel of Experts on Food Security and Nutrition 2019)..

2.7.4 Genetically modified crops

The global area of genetically modified (GM) crops (also referred to as biotech crops) increased more than 100-fold, from 1.7 million ha at the time of their commercial introduction in 1996 to 192 million hectares in 2018. In that year GM crops were grown in 26 countries; 54 per cent of the global GM crop area was in developing and 46 per cent in industrialized countries (International Service for the Acquisition of Agri-biotech Applications [ISAAA] 2019).

In 2018 about half the global acreage of GM crops consisted of soybean, followed by maize (21 per cent), cotton (13 per cent) and canola (5 per cent). Worldwide, 78 per cent of soybean, 76 per cent of cotton, 30 per cent of maize and 29 per cent of canola were GM crops (ISAAA 2019).

Overall, almost 80 per cent of all GM crop approvals involve traits which affect weed, pest and disease control and therefore potentially influence pesticide use (Table 2.7-2). Herbicide tolerance (HT) in soybeans, canola, maize, alfalfa and cotton is the dominant trait, representing 42 per cent of approvals for cultivation and 46 per cent of the global GM crop area in 2018

Table 2.7-2 Global share of approvals for cultivation of genetically modified crops according to the main commercial GM traits. Approved events may be single traits or combinations of traits. ISAAA (2020).

Commercial trait	Number of approvals for cultivation	Percentage of total
Herbicide tolerance	226	42%
Insect resistance	178	33%
Disease resistance	20	3.70%
Abiotic stress tolerance	10	1.90%
Altered growth/yield	3	0.50%
Modified product quality	76	14%
Pollination control system	26	5%

Table 2.7-3 Impact of growing genetically modified crops on volume of pesticide use.

Coverage	Assessment method	Study size	Effect of GM crop on pesticide use volume	Reference
Global	Comparison of GM crop pesticide use with typical application rates in non-GM crops	31 countries	Various HT crops: aggregate impact = 0-19 per cent reduction in a.i. per ha	Brookes and Barfoot (2018), partly reviewed in NAS (2016)
		12 countries	IR cotton and maize: aggregate impact = 30-60 per cent reduction in a.i. per ha	
Global	Systematic review (studies until 2015)	13 studies	In all cases, use of IR crops reduced application of insecticides	NAS 2016
		6 studies	Use of HT crops sometimes initially correlated with decreases in total amount of herbicide applied per hectare, but decreases were generally not sustained	
Global. HT soybean, maize and cotton; IR maize and cotton	Meta-analysis: differences in pesticide use between GM and non-GM crops. (studies 1996-2014)	31 data sets	IR crops: 42 per cent reduction in insecticide use	Klümper and Qaim (2014), reviewed in NAS (2016)
		7 data sets	HT crops: 2.4 per cent increase in herbicide use	
United States and global	Glyphosate use statistics from multiple sources from 1974 to 2014 in the United States and from 1994 to 2014 globally. ISAAA data for HT crops	Global, with special focus on the United States, Brazil and Argentina	Globally, glyphosate use has risen almost 15-fold since genetically engineered glyphosate tolerant (GT) crops were introduced in 1996. Genetically engineered HT crops account for about 56 per cent of global glyphosate use.	Benbrook (2016)
United States Maize, soybean, cotton, rice, wheat	United States Department of Agriculture (USDA) herbicide use data for individual a.i.'s: 1990-2015	National	Average number of herbicide treatments increased in all crops except soybean. Increases were similar in mainly HT crops (cotton, maize) compared to mainly non-HT crops (rice, wheat). In the majority of crops acute and/or chronic hazard quotients decreased over time, indicating potentially lower human health risks.	Kniss (2017)
United States Soybean and maize	Comparative study. Data 1998-2011	5,424 maize farmers	Adopters of IR maize used 11 per cent less insecticide than non-adopters. IR maize adopters used increasingly less insecticides over time than non-adopters.	Perry *et al*. (2016)
		5,029 soybean farmers	Adopters of GT soybeans used 28 per cent more herbicide than non-adopters. Adopters of GT maize used 1.2 per cent less herbicide than non-adopters. GT adopters used increasingly more herbicides over time relative to non-adopters.	
United States Soybean, corn and cotton	Pesticide use statistics 1992-2009 (no explicit distinction between GM and non-GM crops)	Surveys of more than 20,000 farm operations	Substantial decrease in the mass of insecticides applied since the introduction of Bt corn and cotton. Rates of herbicide application to soybean, corn and cotton initially decreased after GT crop introductions. By the early 2000s herbicide rates started to increase in all crops. By 2009 herbicide rates in soybean were substantially higher than pre-GM.	Coupe and Capel (2016)
Bangladesh Eggplant	Pesticide use evaluation	Survey of 1,200 farm households	IR eggplant: 51 per cent reduction in the number of pesticide applications and 39 per cent reduction in the quantity of pesticides applied.	Ahmed *et al*. 2019

a.i. = active ingredient; GT = glyphosate tolerant; HT = herbicide tolerant; IR = insect resistant

Figure 2.7-2 Herbicide use in soybean (kg/ha) in the United States following the introduction in 1996 of GM herbicide tolerant soybean. Glyphosate has progressively replaced other herbicides. Perry *et al.* (2016).

(ISAAA 2019). HT crops are dominated by glyphosate tolerance and, to a lesser extent, tolerance to glufosinate, dicamba, sulfonylurea and oxynil herbicides. Insect resistance is the second most important trait being commercialized; insect resistant (IR) crops are virtually all based on the expression of Bacillus thuringiensis (Bt) toxins.

Insect and disease resistance in a crop can potentially reduce the use of insecticides and fungicides. There is general agreement that the growth of IR crops has significantly reduced the quantity of insecticides applied (Klümper and Qaim 2014; National Academy of Sciences, Engineering and Medicine [NAS] 2016). Reductions in the quantities of insecticides applied range from 11 to 60 per cent compared to non-IR crops, depending on the crop, country, and assessment method used (Table 2.7-3). In some cases the use of IR crop varieties has also been associated with reduced use of insecticides in fields where there are non-IR varieties of the crop, and even with reduced use in other crops, probably due to area-wide pest suppression (NAS 2016). However, in some regions pests have developed resistance to IR crops, leading to a renewed increase of insecticide use (Chapter 4.3.3).

Disease resistant GM crops are grown on a much more limited scale. No reviews were found concerning their impact on fungicide use. However, GM crops with disease resistant traits could potentially reduce the use of fungicides significantly (e.g., late blight resistant potato; Ghislain *et al*. 2018).

The impact of HT crops on the use of herbicides is more ambiguous. An obvious effect is that herbicides used after the introduction of an HT crop converge with those to which the crop is tolerant. Since most HT crops are currently glyphosate tolerant, use of this herbicide has experienced large growth (Figure 2.7-2).

Some studies and reviews show an increase of up to 30 per cent in herbicide use following the introduction of an HT crop, while others indicate declines (Table 2.7-3). Variability in the outcomes of studies appears to be influenced by the assessment methodology used, the HT crop, the time since introduction of the HT crop, and the types of herbicides used before and after introduction of the HT crop.

Several studies appear to indicate an initial decrease in the total rate of herbicide application following introduction of an HT crop, but (significant) increases of herbicide rates afterwards, sometimes reaching higher levels than was the case before its introduction (Coupe and

Capel 2016; NAS 2016; Perry *et al*. 2016). It is likely that this observed pattern of change in herbicide use over time is caused by the emergence of herbicide resistance in the weeds (Chapter 4.3.3). Benbrook (2016) calculated a 15-fold increase in glyphosate use following the commercial introduction of HT crops, considerably higher than in the case of non-HT crops.

2.7.5 Integrated pest and vector management

The concept of integrated pest management (IPM) originated in the late 1950s in the United States, in response to the increasing and indiscriminate use of pesticides. At that time IPM had a relatively narrow focus, combining biological and chemical control and applying economic thresholds before interventions would take place (Ehler 2006). From the 1970s onwards IPM developed into a more holistic ecosystem approach to crop production and protection, combining different management strategies and practices to grow healthy crops and reduce the use and/or risks of pesticides. IPM is a method for analysis of the agro-ecosystem and management of its different elements, in order to control pests and keep them at an acceptable level with regard to environmental, human health and economic requirements.

Many definitions of IPM exist, ranging from what has been referred to as integrated pesticide management (basically promoting judicious pesticide use) (Ehler 2006) to biointensive IPM, which mainly relies on enhancing plant health and conserving beneficial organisms and habitats to limit pest populations (Box 2.7-3). IPM can take different forms that vary in time and space. It is shaped according to site-specific factors such as cropping patterns, cultivation practices, pest pressure, field size, the broader landscape, R&D efforts, availability of training, and farmer attitudes. Furthermore, farmers do not adopt IPM strategies based solely on technical parameters; the social and economic environment in which they operate is also critical (Barzman *et al*. 2017).

Table 2.7-4 **Integrated pest management (IPM) and integrated vector management (IVM) have become central policies for pest and vector management at national and international levels: some examples.**

Entity	Policy instrument	
International organizations		
FAO	*Save and Grow – A Policymaker's Guide to the Sustainable Intensification of Smallholder Crop Production*	FAO (2011)
FAO and WHO	International Code of Conduct on Pesticide Management	FAO/WHO (2014)
WHO	Global Vector Control Response 2017-2030	WHO (2017)
World Bank	Environmental and Social Standards No. 3: Resource Efficiency and Pollution Prevention and Management	World Bank (2018)
European Union and national governments		
European Union	EU Framework Directive on the sustainable use of pesticides, and EU Regulation on the placing of plant protection products on the market	EU (2009a; 2009b)
United States	National roadmap for integrated pest management	United States Department of Agriculture (USDA) (2018)
Morocco	National vector control plan 2019-2025	Direction de l'Epidémiologie et de la Lutte contre les Maladies (DELM) (2019)
Pesticide industry		
CropLife International	Crop protection industry supports FAO on IPM	CropLife International (CLI) (2017)
Voluntary sustainability standards		
Rainforest Alliance	Sustainable agriculture standards	Rainforest Alliance (2020)

Box 2.7-3 Aspects of integrated pest management (IPM). IPM as a continuum scheme adapted from Benbrook *et al.* (1996)

IPM and IVM defined

IPM and IVM have been defined in many ways. FAO and WHO currently use the following definitions (FAO/WHO 2014; FAO 2020):

Integrated pest management (IPM) means the careful consideration of all available pest control techniques and subsequent integration of appropriate measures that discourage the development of pest populations and keep pesticides and other interventions to levels that are economically justified and reduce or minimize risks to human health and the environment. IPM emphasizes the growth of a healthy crop with the least possible disruption to agro-ecosystems and encourages natural pest control mechanisms.

Integrated vector management (IVM) means the rational decision-making process for the optimal use of resources for disease vector control. It aims to improve efficacy, cost-effectiveness, ecological soundness and sustainability of disease vector control interventions for control of vector-borne diseases.

Principles of IPM

The EU has identified eight general principles of IPM which should be implemented by professional pesticide users in all Member States (EU 2009a; as summarized by Creissen *et al.* 2019)

Principle	Components
1. Prevention and suppression	Crop rotation, cultivation techniques, varietal resistance, phytosanitary measures, beneficial organisms ("grow a healthy crop"; FAO 2020)
2. Monitoring	Field monitoring, forecasting, seeking expert advice
3. Informed decision making	Protection measures based on expert advice, action thresholds
4. Non-chemical methods	Preference for biological and physical control methods over chemical ones
5. Pesticide selection	Using pesticides that minimize negative effects on human health and the environment
6. Reduced pesticide use	Reduced doses, reduced application frequency considering the risk for development of pesticide resistance
7. Anti-resistance management	Alternation/mixing pesticides containing multiple modes of action
8. Evaluation	Assessment of the efficacy of measures used to inform future management decisions

IPM as a continuum

Benbrook *et al.* (1996) proposed that IPM systems could be thought of as falling along a continuum. In the shift from chemical-intensive to biointensive IPM, reliance on interventions with pesticides drops and reliance on prevention-based biological practices increases.

→ *Shifting reliance from treatments with pesticides to prevention* →

No IPM	Low level IPM	Medium level IPM	Biointensive IPM
Typical elements of IPM systems (examples)			
▸ Proper calibration of spray equipment ▸ Good application practices ▸ Pest monitoring ▸ Good agronomic practices ▸ Field sanitation	▸ Pest monitoring followed by threshold based pesticide treatments ▸ Delay resistance and secondary pest development ▸ Optimally time pesticide applications ▸ Preventive practices, e.g. resistant varieties, mechanical control	Multi-tactic approaches to: ▸ Limit pest habitats ▸ Enhance beneficial organisms ▸ Grow resistant varieties ▸ Use of cover crops ▸ Use of longer rotations ▸ Apply disease forecasting models	Reliance on preventive measures to limit pest pressure and enhance beneficial: ▸ Enhance plant health and soil quality ▸ Focus on conservation of beneficials and their habitats ▸ Use of biocontrol methods

Table 2.7-5 Effects of integrated pest management (IPM) on pesticide use (global and regional reviews conducted since 2010).

Region	Review coverage	Economic effects	Effect on pesticide use and risks	Reference
Global (mainly low and middle income countries)	Meta-analysis IPM Farmer field schools (FFS) compared to non-FFS farmers 15 studies with medium or low risk of bias	13 per cent average increase in yields 19 per cent average increase in net profits	23 per cent average reduction in pesticide use for IPM and IPPM FFS farmers compared with other farmers Significant increase in adoption of other beneficial practices 39 per cent average reduction in environmental impact quotient (EIQ) score as a result of reduced pesticide use among FFS farmers compared with other farmers No valid estimates possible of impacts on farmer health outcomes	Waddington *et al*. 2014
Global (mainly low and middle income countries)	Systematic review IPM and other FFS compared to non-FFS farmers 7 "more rigorous" studies, published after Waddington *et al*. 2014	9 per cent average increase in yields 28 per cent increase in profits (one study)	35 per cent reduction in insecticide expenditure (one study) 32 per cent increase in adoption of recommended practices	Rejesus 2019
Africa and Asia	Quantitative review 85 IPM projects in 24 countries implemented from 1990 to 2014	Average yield increase of 41 per cent	Average pesticide reduction of 31 per cent Under IPM, a total of 35 of 115 (30 per cent) crop combinations resulted in a transition to zero pesticide use.	Pretty and Bharucha (2015); Pretty (2018)
Western United States	Narrative review Pest management studies and survey published since 2000	IPM programmes have contributed to increased agricultural productivity	IPM programmes have reduced some environmental and human health risks due to decreases in pesticide use. Risks from potential carcinogens and toxic air contaminants may have increased. Pesticide use has declined 40-90 per cent in hazelnut production in the State of Oregon, almond and fresh-market grape production in the State of California, and pear production in California, Oregon and the State of Washington. Significant reductions in insecticide use against pink bollworm and 70 per cent reduction of foliar insecticides against whitefly observed in cotton (includes the effect of Bt cotton)	Farrar, Baur and Elliott (2015; 2016a; 2016b)
United States	Systematic review for various key arable crops (excluding fruits and vegetables) in the period 1996-2005	Not assessed	Pesticide cost per acre was used a proxy for pesticide use. On average throughout the United States, IPM adoption and pest management training led to slightly greater pesticide spending by farmers.	Maupin and Norton (2010)

IPM has become the official policy for pest management, as part of sustainable intensification of crop production, of international organizations and many national governments, as well as of the crop protection industry and standard-setting bodies. Similarly, integrated vector management (IVM) is currently the central paradigm for disease vector control (Table 2.7-4). IPM is thus an important part of sustainable intensification of crop production. Through enhancing ecosystem function and making the agricultural ecosystem healthier, more ecosystem services are provided: in this case pest control (FAO 2020b). In EU Member States applying the principles of IPM has become an obligation for farmers (Box 2.7-3).

Measuring the level of IPM implementation has been difficult, as crop production and protection are site-specific, dynamic, and influenced by economic, agronomic and social parameters (Barzman *et al*. 2015). Despite considerable investment in IPM-related research and broad political support for IPM (at least on paper) over several decades, the adoption of IPM has generally been considered to be relatively low (IAASTD 2009) both in higher income countries (Ehler 2006 for the United States; Hokkanen 2015 and European Commission [EC] 2020a for Europe; Zalucki, Adamson and Furlong 2009 for Australia), as well as in lower income economies (Parsa *et al*. 2014; Bottrell and Schoenly 2018; Alwang *et al*. 2019).

Nevertheless, IPM has been highly successful and cost-effective in many crops and countries, where over time pest management systems have been implemented that are more sustainable from both an agronomic and economic point of view (e.g., pears in California [Weddle, Welter and Thomson 2009], cotton in Arizona [Naranjo and Ellsworth 2009], various crops in the western United States [Farrar, Baur and Elliott 2015], tomatoes in New Zealand [Cameron *et al*. 2009], rice in Indonesia in the 1990s [Thorburn 2015] and cotton in Mali [Settle *et al*. 2014], among many others).

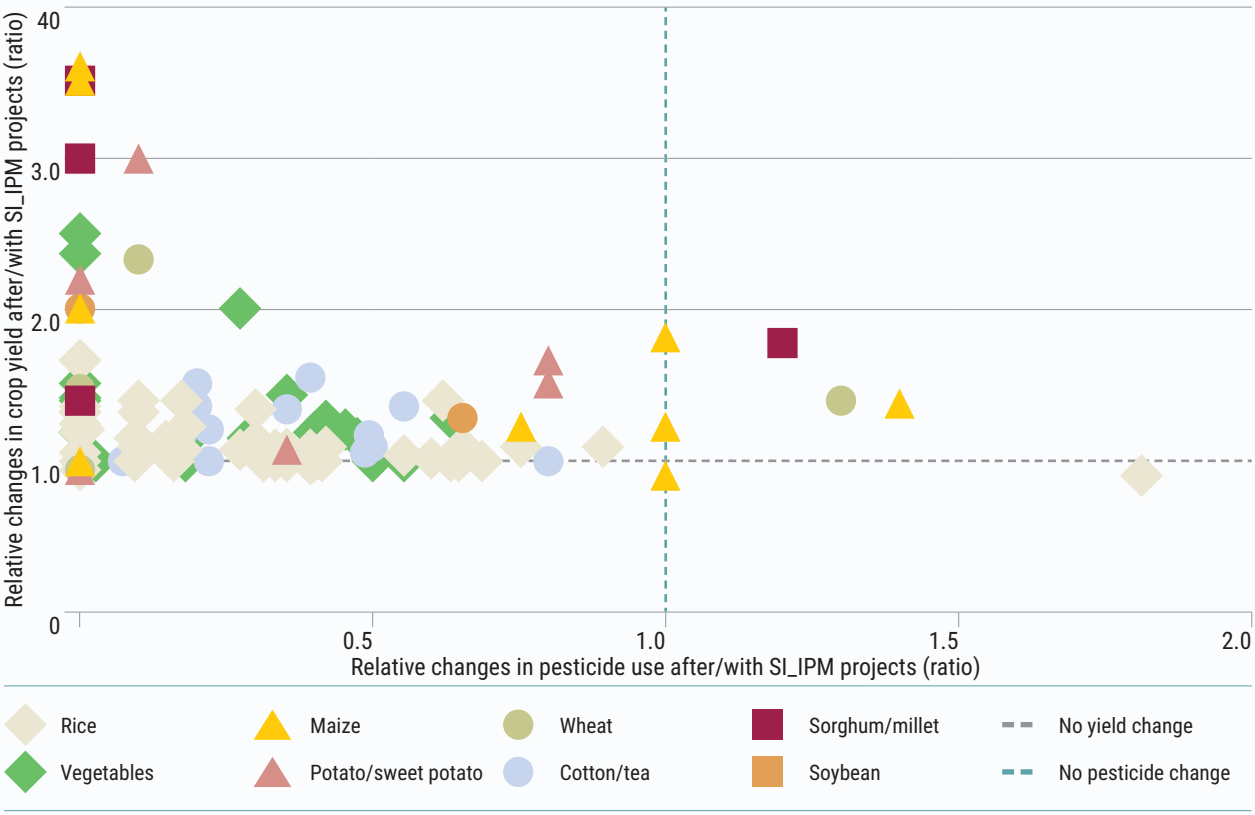
To some extent the degree of adoption of IPM by farmers depends on their understanding of what constitutes IPM. Most farmers will apply one or more crop production and protection measures which can be part of an IPM approach (e.g., resistant cultivars, crop rotation, regular cleaning of machinery, balanced fertilization, pest monitoring), often without being recognized as such, but they may not apply all the practices which would constitute "ideal" (ecological or biointensive) IPM (Maupin and Norton 2010; Barzman *et al*. 2015). For example, Creissen *et al*. (2019) surveyed arable farmers in the United Kingdom and found that all of them had adopted IPM to some extent (an average 65 per cent adoption rate) but only 6 per cent had adopted more than 85 per cent of the measures that were theoretically possible. Although simpler practices may be adopted more rapidly than more complex ones, that does not imply that partial adoption of IPM is a poor investment (Norton *et al*. 2019).

It is important to emphasize, however, that IPM implementation does not mean choosing a number of measures "randomly" based on personal preferences, cost or ease of implementation. As proposed by Benbrook *et al*. (1996), IPM systems can be thought of as falling along a continuum. In the shift from chemical-intensive to biointensive IPM, reliance on interventions with pesticides should drop and reliance on prevention-based biological practices should increase (Box 2.7-3).

While it may be pragmatic to accept incomplete adoption of IPM, especially if it still leads to reduced pesticide risks, this does not solve the problem of ensuring long-term sustainability, let alone long-term sustainable intensification. More fundamental changes in agricultural production are then needed.

An important question to be addressed in this report is whether IPM adoption (irrespective of the exact definition) will lead to a reduction in the use of pesticides, their risks to the environment and human health, or farmers' dependency on pesticides. Most recent reviews indicate that farmers use less pesticides if they apply IPM (Table 2.7-5, Figure 2.7-3). Furthermore, pesticide use reductions generally do not result in reduced crop yields; on the contrary, in most cases (limited) increases in yields have been observed. The only exception is a review by Maupin and Norton (2010) for the United States, where on average a slight increase in pesticide expenditures was observed in

Figure 2.7-3 IPM projects and programmes in Asia and Africa result in increased crop yields and reduced pesticide use. Shown are 85 projects in 24 countries. Pretty (2018)



the case of farmers who adopted either partial or complete IPM measures. Overall, it seems justified to conclude that IPM practices, if effectively adopted and implemented, generally reduce the use of pesticides in cropping systems.

One argument for IPM adoption has been that it leads to a reduction in environmental and human health risks as a result of reduced application of pesticides, expanded use of reduced risk products, improved pesticide application practices, or better use of precautionary measures such as PPE.

Waddington *et al*. (2014) found a considerable reduction in environmental and health risks following IPM adoption in a review of three studies, as measured through an aggregate environmental impact indicator (Table 2.7-5). In individual studies it has often been shown that implementation of IPM reduces human health effects observed in farmers, e.g., in India (Mancini *et al*. 2009), Cambodia and Viet Nam (FAO 2013), Bolivia (Jørs *et al*. 2014) and Costa Rica (Fuhrimann *et al*. 2020). However, this is not always the case, e.g., in Uganda (Clausen *et al*. 2017) or Bangladesh (Gautam *et al*. 2017) where no clear impacts on risk reduction could be determined. This variability in results is at least partly explained by the complexity of measuring pesticide exposure and effects (Fuhrimann *et al*. 2020). So far, no systematic review appears to have been conducted assessing the empirical relationship between IPM adoption and the reduction of environmental and health effects.

Despite successful local application of IPM, its impact on pesticide use has not been visible at global or continental scale due to its limited large-scale adoption. Peshin and Zhang (2014) analysed IPM and pesticide use in the United States, Europe and Asia and concluded that pesticides were and are the primary pest management tools. Low-volume pesticides and insect resistant transgenic crops both decreased and stabilized pesticide use in the 1990s and early 2000s. Since then, pesticide sales have regained an upward trajectory and their use in agriculture has increased. In Australia, Zalucki, Adamson and

Furlong (2009) found that during the previous 30 years insecticide input costs per hectare had increased faster than the price index, indicating that insecticide inputs had actually increased.

Overall, global pesticide use per unit cropland has steadily increased since the early 1990s, while use per unit crop output has remained stable despite the increased crop protection potency of more modern pesticides (Figure 2.4-18).

It can be concluded that IPM, however defined, reduces the use and risk of pesticides locally. However, its implementation is limited in both lower and high income countries. The overall impact of IPM has therefore not resulted in reduced global dependence on pesticides in crop protection.

2.7.6 Organic production

There are many definitions of organic agriculture, but all converge to characterize it as a system that relies on ecosystem management rather than external agricultural inputs. Organic agriculture begins to consider potential environmental and social impacts by eliminating the use of inputs such as synthetic fertilizers and pesticides, veterinary drugs, genetically modified seeds and breeds, preservatives, additives and irradiation. Use of these inputs is replaced by site-specific management practices that maintain and increase long-term soil fertility and prevent pest and diseases (FAO 2020c).

The Codex Alimentarius Commission has defined organic agriculture as "a holistic production management system which promotes and enhances agro-ecosystem health, including biodiversity, biological cycles, and soil biological activity. It emphasizes the use of management practices in preference to the use of off-farm inputs, taking into account that regional conditions require locally adapted systems. This is accomplished by using, where possible, agronomic, biological, and mechanical methods,

Figure 2.7-4 Distribution of main organic land use types and crop categories in 2018. Globally, 71.5 million ha were under organic production in 2018. Schlatter *et al.* (2020).

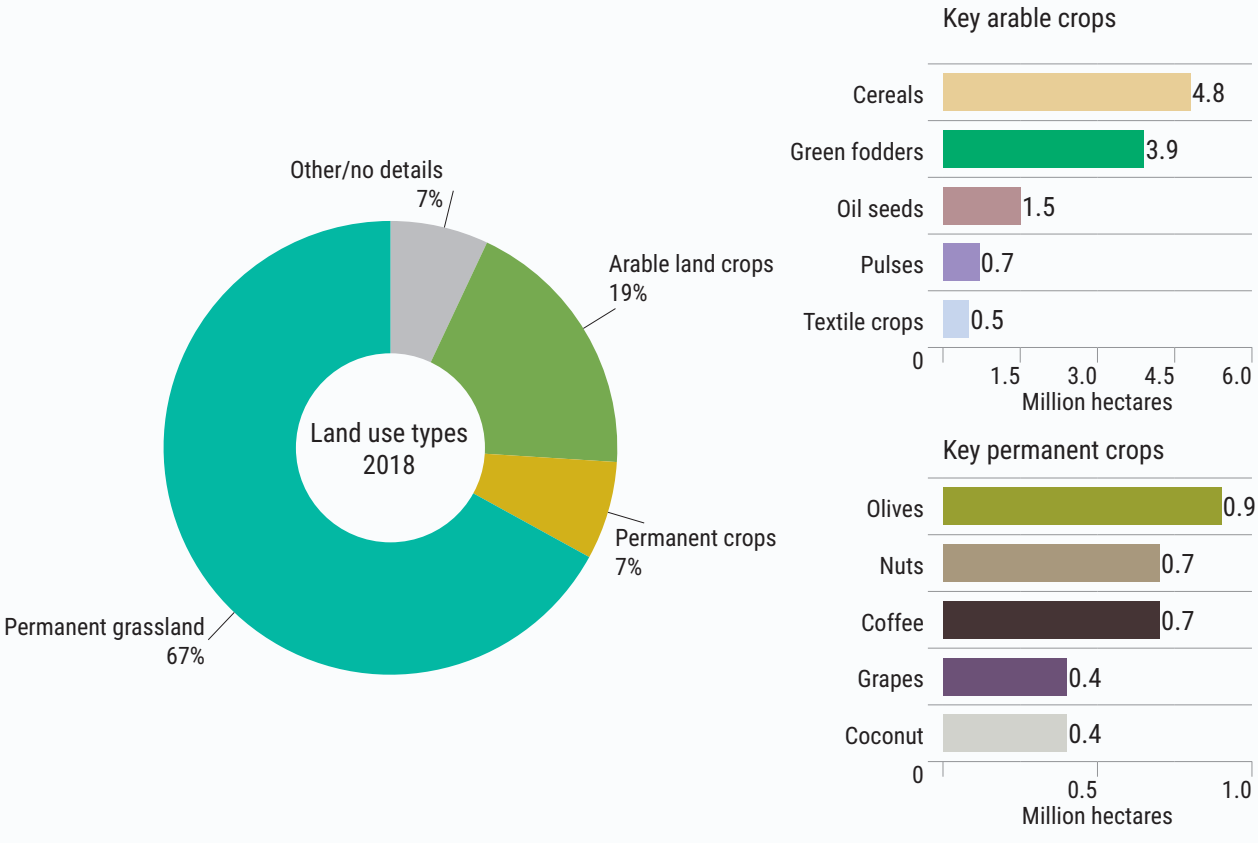


Table 2.7-6 Examples of pesticides allowed in organic agriculture. IFOAM – Organics International (2014)

Substances of plant or animal origin	Substances of mineral origin	Microorganisms
Chitin nematicides (natural origin)	Copper salts	Fungal preparations (e.g. spinosad)
Natural acids	Diatomaceous earth	Bacterial preparations (e.g. *Bacillus thuringiensis*)
Neem	Light mineral oils (paraffin)	**Others**
Pyrethrum	Sulfur	Iron phosphates (as molluscicide)
		Soft soap

as opposed to using synthetic materials, to fulfil any specific function within the system" (Codex Alimentarius Commission 2013).

In 2018, 71.5 million ha of certified organic agricultural land was recorded (two-thirds grazing land). Arable land comprised 13.3 million ha and permanent crops 4.7 million ha (Figure 2.7-4). The area under organic agriculture is steadily increasing globally, having doubled since 2010. Organic agriculture represents about 1.5 per cent of the world's total agricultural land, while organic arable and permanent crops are grown on 1.2 per cent of total global cropland (Schlatter *et al*. 2020; FAOSTAT 2020).

In organic agriculture the use of almost all synthetic pesticides is prohibited, but some biological pest control agents and inorganic compounds are allowed (Table 2.7-6).

Given the limited area of cropland currently under organic agriculture, its contribution to a reduction in pesticide use at the global scale is still small. However, certain principles of organic agriculture have been adopted in more mainstream forms of agriculture.

Regulatory drivers

2.7.7 Pesticide legislation and policies

Pesticide legislation[1] and policies, as well as legislation and policy that address public health, the environment and trade more generally, can greatly influence pest and pesticide management, including production and use. This is true not only nationally, but also in the case of regional and international instruments.

Overall, legislation concerned with pesticide management – but also public health and environmental legislation – aims to reduce the risks and sometimes the use of pesticides. The promotion of international trade in agricultural commodities is increasingly accompanied by policies that encourage the sound management of pesticides and reduction of pesticide residues in commodities.

Agricultural policies, on the other hand, have the potential either to increase or limit pesticide use. Pesticide dependency may grow, especially if policies exclusively emphasize production growth in specific crops; or it may decrease if policies focus on supporting production approaches which are agronomically and environmentally sustainable in the long term and/or promote more integrative farming (e.g., rice plus fish plus ducks plus vegetables).

[1] Throughout this report references to legislation are understood to refer to legal and other measures adopted by a country, including both primary and secondary legislation.

Legislation and policies that directly affect pesticide production, management and use are discussed in detail in Chapter 3.

Economic drivers

2.7.8 Pesticide marketing

In low income countries, pesticide distributors and retailers are farmers' main source of information. Agricultural extension services and private advisory services have only limited coverage. This appears to be a consistent pattern across low and middle income regions of the world (Anang and Amikuzuno 2015; Haj-Younes, Huici and Jørs 2015; Okonya and Kroschel 2015; Schreinemachers *et al*. 2017), but it is also encountered in some high income economies (Oleksa Vanzant 2014-2015). As a result, there is a high likelihood that farmers and other pesticide users will receive information in which pesticides are proposed as the predominant pest management option. Schreinemachers *et al*. (2017) found that farmers in Cambodia, Laos and Viet Nam who sought advice from pesticide shopkeepers used 251 per cent more pesticide than the average.

While public and private sector initiatives to train retailers in more comprehensive pest management advice may lead to increased use of lower risk pesticides by farmers (Lekei, Ngowi and London 2014), they cannot be expected to result in overall reduced pesticide sales.

Furthermore, widespread pesticide advertising in some parts of the world exposes potential pesticide users to constant information and incentives encouraging them to purchase and use pesticides. Article 11 in the International Code of Conduct on Pesticide Management addresses pesticide advertising in all media (FAO and WHO 2014). It has been endorsed by major pesticide industry associations. However, a report by the Special United Nations Rapporteur on the right to food has suggested that "aggressive, unethical marketing tactics" are still used by pesticide and agroindustry (United Nations [UN] 2017).

Continuous pesticide advertising using radio and television in the mid-2000s was identified as a key reason why rice farmers in Viet Nam discontinued IPM practices, even though rice yields were maintained with a reduced number of insecticide applications (Escalada *et al*. 2009). No independent reviews are available of the impact of current pesticide advertising on pest management practices in different parts of the world.

2.7.9 Pesticide prices

The price of a pesticide at the sales outlet depends on many factors. They include the cost of the manufacturing and formulation process, the size of the market in a given country of region, exchange rate factors, the level of national taxation, the purchasing power of farmers and other pesticide users, the expected benefits of the pesticide for a farmer, and competition between manufacturers.

Competition based on whether an active ingredient is under patent or off-patent has markedly influenced pesticide pricing in the last few decades. Production and sales of off-patent, non-proprietary pesticides, also referred to as generic pesticides, has greatly increased over the last 20 years or so. Off-patent, non-proprietary pesticides currently represent around 70 per cent of the total market for crop protection products (AgbioInvestor 2019). This situation has increased competition and reduced sales prices.

For example, glyphosate, the most used herbicide in the world (UNEP 2020a), was introduced on the market in 1974 and the last global patent protection of the Roundup (the first product containing glyphosate) expired in 2000. Subsequently, large-scale generic production got under way, especially in China. Prices of glyphosate products in the United States dropped by almost 60 per cent between 2000 and 2014; similarly, glyphosate prices in Mali in 2015 were 35 per cent lower than in 2008 (Benbrook 2016; Diarra and Haggblade 2017).

Lower pesticide prices increase access to such products by farmers and other users. The presence of cheap generic products on the market also complicates the introduction of newer,

low-risk chemical products and biopesticides, as well as non-pesticide alternative pest management options.

2.7.10 Commodity prices

Pesticides tend to be used more in the case of high-value agricultural commodities than in that of low-value subsistence crops. Furthermore, the structure of the value chain can influence commodity prices, as the prices paid to a farmer for her/his crop may be higher when value chains are more direct. Investments in pest management are considered worthwhile if potential revenues are higher, which often translates into use of pesticides. When commodity prices are high, farmers tend to spend more on pesticides (Waterfield and Zilberman 2012; AgbioInvestor 2019).

2.7.11 Fiscal policies

Fiscal instruments that can affect pesticide use have been adopted by countries, such as direct subsidies, taxes, charges and fees, implicit subsidies, and various tax exemptions and reductions.

Direct subsidies

Agricultural input subsidies are used to meet policy objectives ranging from short-term responses (to enhance food security and income and avoid poverty traps, maintain affordable prices for major crops, and avoid high costs of agricultural inputs) to more long-term structural objectives such as addressing market imperfections and supporting the adoption of new technologies. In general, input subsidies lower costs to producers, which increases the risk of their over- or misuse, with potentially harmful consequences for the environment and the health of farmers and consumers (UNEP 2020b).

In the 1980s and 1990s the majority of developing countries provided financial incentives to farmers to use pesticides; in particular, they directly and indirectly subsidized pesticide imports, domestic manufacture, and local sales and use with a combination of mechanisms (Farah 1994).

Direct subsidization of pesticides has virtually been abandoned in OECD countries (UNEP 2020b). It is also becoming less common in low and middle income countries (unlike subsidization of fertilizers; see Chapter 7.4). On the other hand, subsidies increasingly appear to be provided for biopesticides or other low risk pest control tools as a way to promote IPM and biocontrol (Box 2.7-4).

Implicit subsidies

Much more common than direct subsidies are fiscal measures that implicitly subsidize pesticide use. Agricultural inputs are subject to general ad-valorem taxes as applied to other commodities, such as an import tax, value added tax (VAT) or other general taxes. Many high income countries and an increasing number of lower income countries currently exempt or reduce (i.e., set at zero rate) general and import taxes on pesticides. Without specific conditions, such tax exemptions act as an implicit or hidden subsidy (UNEP 2020b). Given their dominant share of the pesticide market, the use of conventional chemical pesticides tends to profit most from tax reductions/exemptions. However, some countries also offer fiscal incentives to encourage, for instance, organic farming practices, thus facilitating pest management with lower environmental and health risks.

Pesticide taxes

Taxes on pesticides are generally imposed to create incentives for producers and consumers to shift towards less polluting products or substances, stimulate innovation and raise additional fiscal revenues. Countries use revenues obtained from pesticide taxes in different ways. These revenues may simply accrue to the state budget, but in some countries they are used to compensate farmers, cover the costs of pesticide inspection and enforcement, provide training, support projects promoting more sustainable pest control, or fund research on innovative practices (UNEP 2020b).

Despite this potential, beyond general ad valorem taxes only a few countries (mainly in Europe) have so far levied taxes on pesticides with the clear intention to reduce pollution (Box. 2.7-4).

Box 2.7-4 Examples of fiscal instruments that may influence pesticide use.

Direct pesticide subsidies

The number of countries with direct subsidies on pesticides has declined over time (UNEP 2020). Some examples of current pesticide subsidies are:

- **Botswana:** Integrated Support Programme for Arable Agriculture Development (ISPAAD) (since 2013). Subsidy on herbicides for emerging and commercial farmers [30-35 per cent of costs of pesticide] and horticultural enterprises [40-60 per cent of costs] (ISPAAD 2013).
- **State of Hawaii, United States:** Coffee Berry Borer (CBB) Pesticide Subsidy Program (since 2014). To assist Hawaii coffee farmers with the cost of biopesticides containing the fungus, Beauveria bassiana. [up to 50 per cent of cost of the pesticide]. (State of Hawaii Department of Agriculture 2019).
- **China (Beijing Municipality):** "Green Pest Control" subsidy programme (since 2009). Products are subsidized at different rates prioritizing natural enemies, pollinating insects, biopesticides and plant protection tools, and finally by least toxic/residual synthetic pesticides [50-90 per cent of the cost of the product] (Wei *et al*. 2019).

Indirect pesticide subsidies

A considerable number of both high and low income countries currently apply exemptions or zero rates on VAT and general taxes on pesticides (UNEP 2020). These include:

All pesticides

- **European countries:** standard VAT rate on goods ranges from 17 – 25 per cent. Italy applies a reduced VAT rate at 4 per cent on pesticides; Cyprus, Poland, Portugal, Romania, Slovenia and Spain all apply a reduced VAT rates; and Switzerland a reduced VAT rate on pesticides at 2.5 per cent.
- **Republic of Korea:** Pesticides are VAT exempted.
- **United States:** Complex set of exemptions from sales taxes for pesticides.
- **Thailand, Kenya, India:** General and import taxes on pesticides exempted or at zero rate.

Pesticides for organic agriculture

- **Norway:** Pesticide product applicable under organic farming practices are fully tax exempted.
- **Denmark:** Organic farms are entitled to receive benefits from tax revenues.
- **France:** Introduced a combined tax system with preferential treatment of organic farming practices, whereby a reduced tax rate on "organic pesticides" is applied.

Pesticide taxes

Few countries currently impose specific taxes on pesticides (UNEP 2020):

- **Denmark:** Since 1996. A new tax system was introduced in 2013, based on the "environmental load index" and sales volumes: the higher the potential risk of a pesticide, the higher the tax, amounting up to 100 per cent of the sales price for high risk pesticides (Hansen 2017).
- **France:** Since 2000. Pesticide tax was updated in 2008, based on the toxicity of the pesticide and its sales volume, ranging from 0.9 – 5.1 EUR/tonne, amounting to up to 5-6 per cent of the sales price for the most toxic pesticides (OECD 2017)
- **Norway:** Since 1988. Tax rate is based on environmental and human health risks of the pesticide, and recommended use per hectare; in 2015 it ranged from 1 – 21 EUR/ha for farming, and higher for domestic use (Böcker & Finger 2016).
- **Sweden:** Since 1984. Flat rate of (in 2015) 3.6 EUR/kg of active ingredient (Böcker & Finger 2016).
- **Mexico:** Since 2014. Tax rate of 6 – 9 per cent based on acute toxicity, classified according to the GHS (Servicio de Administración Tributaria 2020).
- **Mozambique:** Import fee varying according the pesticide acute toxicity class, ranging from 0.15 – 0.2 per cent of the FOB value of the pesticides (Government of Mozambique 2007).
- **Kenya:** Import fee of 0.8 per cent of FOB value of the pesticides; part of the revenues is used for training and inspections (Pest Control Products Board 2020)

The adoption of pesticide taxes has been constrained primarily by concerns about the negative impact on competitiveness in global markets.

In terms of the tax rate applied, countries increasingly classify pesticides according to their public health and environmental risks or hazards and assign a specific tax rate accordingly. For instance, Denmark, France and Norway have adopted different tax rates per active substance, reflecting these substances' toxicity to the environment and human health and taking into the account characteristics such as degradability in soils, bioaccumulation and leaching potential. There is no international consensus on how to categorize such relative risks (UNEP 2020b) (Box 2.7-4).

The impact of pesticide subsidies and taxes on pesticide use and impacts

The available literature remains largely divided on the economic benefit-cost ratio of pesticide subsidy programmes. The drawbacks include high fiscal costs, mismanagement of funds, appropriation of subsidies by local elites, and ineffectiveness in reaching poor smallholder farmers. At the same time, agricultural input subsidies generally lower the per unit variable cost of pesticides and create incentives to increase the intensity of input use, with potentially harmful consequences for the environment and the health of farmers and consumers (UNEP 2020b).

Possibly due to their limited current use, few if any recent studies appear to have been conducted on the economic – as well as environmental and health – impacts of direct and indirect pesticide subsidies (unlike the impacts of fertilizer subsidies, which have been studied more widely). Neither does the effect on farmer revenues and agricultural production of abandoning pesticide subsidies seem to have been much studied; no examples of significant declines in production or higher food prices were found. Overall, the costs and benefits of pesticide subsidies are unclear.

It is also challenging to assess the impacts of pesticide taxes, given their relatively recent introduction in only a handful of countries. These impacts depend on tax design, including the structure of the incentives created, the tax rates adopted, demand price elasticity, and precision in targeting, among other factors. Moreover, as countries often introduce a complex package of policies to address the use and management of pesticides, it is difficult to disentangle the contributions of specific instruments from the impact of the wider policy package (Lee, den Uyl and Runhaar 2019; UNEP 2020b).

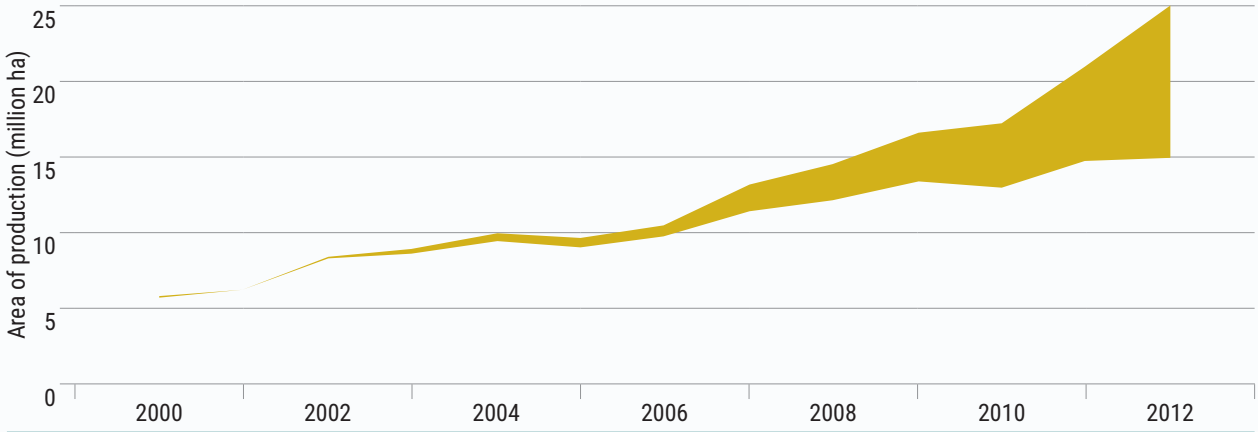
Several studies suggest that the overall impact of these taxes as currently designed has been rather limited, and that there is a broad perception that existing tax rates have been too low to foster significant changes in the use of pesticides or their environmental or health impacts (Böcker and Finger 2016; OECD 2017; UNEP 2020b). However, reductions in the risks of pesticide use have been associated with the introduction or strengthening of pesticide taxes in Denmark and Norway (Hansen 2017; UNEP 2020b).

The effectiveness of pesticide taxes in reducing the use of harmful products depends on various factors, including price elasticities which vary depending on the product type, time frame and type of farming. The elasticity of demand for pesticides is generally low (i.e., changes in pesticide purchasing by farmers are not very responsive to price changes). Experiences in European countries further highlight the importance of revenue redistribution mechanisms in ensuring acceptance of the tax (UNEP 2020b). In reviewing policy instruments for pesticide use reduction in Europe, Lee, den Uyl and Runhaer (2019) found that imposing pesticide taxes in isolation was not particularly effective. Dedicated taxes did result in pesticide use reductions in combination with one or two additional policy measures, such as training, advisory services or regulations.

2.7.12 Voluntary sustainability standards

Voluntary Sustainability Standards (VSS) (sometimes referred to as private standards) are standards that specify requirements for a product or a process that producers, traders or retailers need to meet in relation to sustainability indicators. These requirements can include respect

Figure 2.7-5 The global area of cropland under VSS certification has been increasing rapidly. Tayleur *et al*. (2017).



Note: Data are for 12 major certification standards; since multiple certifications can apply to the same area of land, the minimum and maximum range is shown.

for basic human rights, workers' health and safety, the environmental impacts of production, and land use planning (FAO 2017; United Nations Forum on Sustainability Standards 2020). VSS have been developed by the private sector, governments, non-governmental organizations, and through multi-stakeholder initiatives, often in response to pressures from consumers.

Contrary to mandatory governmental measures, a producer's decision to participate in and comply with VSS is voluntary, although demand-led pressure from buyers can make the producer's choice to adopt a VSS less discretionary. Compliance with these standards by producers and other market actors is normally monitored through third-party certification.

Rewards for adoption of VSS by producers range widely. They include price premiums for the production of certified products, greater market access, and training and support for production and marketing (Smith *et al*. 2019).

The total area under certification for VSS has been increasing rapidly (from approximately 5.7 million hectares in 2000 to 15-25 million hectares in 2012, or about 11.0 per cent annually) (Figure 2.7-5). Across all crops overall coverage has remained low, at just 1 per cent of global cropland (Potts *et al*. 2017; Tayleur *et al*. 2017). However, in the case of certain crops a relatively large percentage of production volume was certified in 2014: for example, cocoa (30 per cent), oil palm (20 per cent), coffee (19 per cent), tea (18 per cent) and bananas (12 per cent). On the other hand, for major crops such as wheat, rice, maize, soybean and cotton certification levels were at a maximum of 1 per cent of production volume (Potts *et al*. 2017).

Many standards include IPM, the prohibition of certain pesticides, and pesticide use monitoring as part of their requirements. They can therefore influence pesticide use and resulting risks (Table 2.7-7). Nevertheless, it has been argued that those who adopt voluntary standards are faced with the dilemma that pesticide inputs are often directly related to (increased) yields. Consequently, there appears to be some reluctance to explicitly require that pesticides are used only as a last resort. Potts *et al*. (2017) found that the degree of obligation to meet this requirement was only 48 per cent across the 15 VSS reviewed.

Most VSS have systems in place to monitor their success towards achieving sustainability goals. However, few published reports of these standards' economic, environmental or social effectiveness meet rigorous scientific requirements (DeFries *et al*. 2017; Oya, Schaefer and Skalidou 2018). Out of 24 cases

Table 2.7-7 Selected pest and pesticide management requirements of major voluntary sustainability standards.
Fairtrade (2014); IFOAM – Organics International (2014); DeFries *et al*. (2017); Potts *et al*. (2017); Tayleur *et al*. (2017); Better Cotton Initiative (2018); GlobalG.A.P. (2018); Fairtrade (2019); Rainforest Alliance (2020).

Standard	Major crops	Requirements			
		IPM as required approach (pest/ disease monitoring, pesticides used as last resort)	Pesticide restrictions (not all registered pesticides allowed for use)	Occupational health (e.g. PPE, training, washing facilities)	Pesticide use monitoring (record keeping for pesticide applications)
IFOAM – Organics International (organic)*	Bananas, cereals, coffee, cocoa, oilseeds	Organic production	Yes (all prohibited except those on the list of allowed crop protectants)	Yes	Yes
Rainforest Alliance/UTZ	Bananas, coffee, cocoa, tea, palm oils	Mostly	Yes (lists of prohibited and risk mitigation use pesticides]	Yes	Yes
Fairtrade	Bananas, coffee, cocoa, tea	Mostly	Yes (hazardous materials list: Red List = prohibited, Orange List = restricted, Yellow List = flagged)	Yes	Yes
Better Cotton Initiative (BCI)	Cotton	Partly	Yes (prohibition of Stockholm Convention, Rotterdam Convention and Montreal Protocol listed pesticides; phase out of pesticides with other Highly Hazardous Pesticide [HHP] criteria)	Yes	Yes
GlobalGAP	Bananas	Yes	No [national registration]	Yes	Yes

* In many countries organic production standards have been included in legislation, in which case they are no longer voluntary.

of implementation of sustainability standards reviewed by DeFries *et al*. (2017), 36 per cent of environmental impact indicators showed a positive effect while 64 per cent showed no significant effect. This study did not assess pest and pesticide management parameters.

It can be expected that even partial implementation of pest and pesticide management requirements will lead to a reduction in adverse effects on the environment and human health. So far, however, no systematic evaluations of the effects of VSS on pesticide use, or on pesticides' adverse effects, have been made.

Private food safety standards are a particular type of VSS. These standards are generally established by private firms (e.g., large retailers or brand owners) or standard-setting coalitions (e.g., GlobalG.A.P or SQF 1000) operated by retailer associations (Hensen and Humphrey 2009; Ecratum 2020).

Among the main drivers of private food safety standards are demonstration of due diligence by food chain operators to ensure food safety; global sourcing and the need for improved supply chain management; heightened consumer interest in food safety; and the possibility for individual food firms to distinguish themselves on the market

(although there is general agreement that food safety should not be used by the food industry as a competitive tool) (Clarke 2010).

In general, collective private food standards refer to prevailing official pesticide residue regulation and do not set additional requirements. However, private pesticide residue provisions may also be stricter than corresponding Codex Alimentarius provisions and national regulations. In particular, private retail labels may impose more stringent maximum residue limits or impose limitations on the total number of residues present on the food, neither of which is generally based on scientific considerations (Clarke 2010). It is therefore not clear cut whether private food standards significantly reduce the risks of pesticide residues to consumers.

There is an ongoing debate about the impact of private standards on small producers that might be excluded from access to the market. It is easier for bigger producers to make the investments to meet certain requirements, putting small producers at a disadvantage. This is likely to lead to "winners" and "losers" in a world where increasingly strict food safety requirements – driven by both the public and private sectors – need to be complied with. In practice, it has been found difficult to separate out the specific impacts that private standards could have on developing countries' food commodity exports from a host of other factors (Hensen and Humphrey 2009).

Public health drivers

2.7.13 Food safety

Limiting the levels of pesticide residues in food and drinking water has been an important driver for regulating and restricting the use of pesticides. Maximum residue limits (MRLs) (or tolerances) have been established for many pesticide-commodity combinations, with the aim of enforcing good agricultural practices (Chapter 4.4.6). The establishment of MRLs in major importing countries, such as the EU Member States, the United States and Australia, has also had a wide influence on pesticide use in countries exporting agricultural commodities. This also holds true for countries where national MRLs may not have been established (yet) or are insufficiently enforced.

On the other hand, pesticides may also be used to increase food safety. This is the case, for instance, if they are used to control the production of mycotoxins such as aflatoxins and ochratoxin A in food. These mycotoxins, which may be formed by fungi (moulds) that grow on numerous foodstuffs such as cereals, dried fruits, nuts and spices, can cause severe acute and chronic health effects. Fungicides and insecticides may be applied to control fungi that produce mycotoxins, although the extent to which this results in major increases in pesticide use is not known.

Overall, food safety issues most likely lead to a reduction in pesticide use.

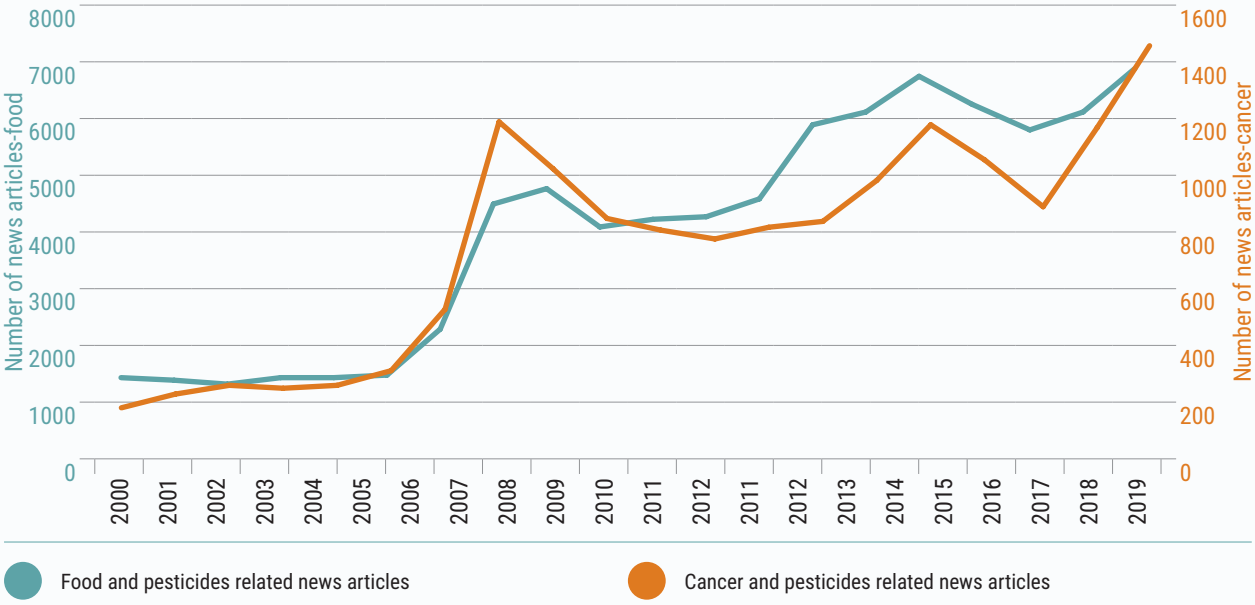
2.7.14 Public concerns about human health

Public concerns about the effects of pesticides on human health and food safety have been voiced almost as long as synthetic pesticides have been used (Dunlap and Beuss 1992).

The presence of pesticide residues in fruit, vegetables or cereals was found to be the overall highest concern in EU Member States in 2010: 31 per cent of citizens were very worried about this issue and 41 per cent were fairly worried, an increase compared to 2005 (EC 2010). In India, 62 per cent of respondents in a wide survey felt that pesticide residues were associated with high or medium human health risks (Khrishna and Qaim 2008). More recently, pesticides, chemicals and toxins were seen as the greatest threat to food safety by German consumers (Koch *et al*. 2017). In a national survey on perceptions of environmental public health risks in the United States, 31 per cent of adults indicated they were concerned about the health risks posed by pesticides (Shin *et al*. 2019).

Pesticides and food have also figured prominently in newspaper reports. Since the early 2000s the number of articles published globally on this topic in English has increased almost five-fold, from 1,400 to 6,200 per year (Figure 2.7-6).

Figure 2.7-6 Public health concerns in the news. Shown are the number of newspaper articles about food and pesticides as well as cancer and pesticides, published annually over the last 20 years.



Data obtained from the NexisUni database (LexisNexis 2020) using the search terms *pesticid* AND food and pesticide* AND cancer*, for newspaper articles published globally in English.

Health concerns that have received much public attention include pesticide drift to residential areas and chronic health effects such as cancer or neurotoxic reactions, both in high income countries (Horan 2015; BNNVARA 2019; Faux 2020) and low and middle income countries (Route to Food 2019). As an example of greater public interest in pesticides and public health, the number of newspaper articles about pesticides and cancer almost quadrupled over the last 20 years (Figure 2.7-6).

Increased public concerns about pesticides and health or food safety have clearly been an important incentive for governments around the world to tighten pesticide legislation and improve monitoring controls.

2.7.15 Vector-borne diseases

Vector-borne diseases, which account for an estimated 17 per cent of the global burden of communicable diseases, claim more than 700,000 lives every year. The burden is highest in tropical and subtropical areas, with malaria the most important disease. More than 80 per cent of the world population is also at risk of other vector-borne diseases such as dengue, chikungunya leishmaniasis, Chagas disease, schistosomiasis and lymphatic filariasis. Others such as Lyme disease and tick-borne encephalitis, are spreading rapidly in temperate regions (WHO 2017).

While the overall volume of insecticides used for human disease control seems to have decreased, this is largely due to the replacement of high volume organochlorines and organophosphate insecticides by low volume pyrethroids and neonicotinoids (Chapter 2.4.3).

Given the recent alarming resurgence of certain vector-borne diseases such as dengue and Zika, and the serious threat they pose to public health and economic development, the World Health Assembly adopted a Global Vector Control Response 2017-2030 (WHO 2017). It aims to reposition vector control as a key approach to reduce mortality and prevent, control and eliminate the burden of vector-borne diseases. One of the pillars of action of this strategy is to scale up and integrate vector control tools and approaches. It can therefore be expected that the use of insecticides for vector control,

including insecticide-treated nets, will increase. The resurgence of some vector-borne diseases is exacerbated by the widespread increase in the frequency and intensity of insecticide resistance observed in several disease vectors, especially mosquitos, the world over (Chapter 4.3.3).

The volume of pesticides used for vector control equals less than 0.5 per cent of the volume of pesticides used in agriculture (Chapter 2.4.3). Thus even a considerable increase in their application will have a limited impact on the quantity of pesticides used globally. Nevertheless, since much vector control is conducted in or close to human habitations, the possible human health risks of vector control pesticides merit specific attention.

Environmental drivers

2.7.16 Climate change

The impact of climate change on agricultural The impact of climate change on agricultural production and food security has been an increasing focus of research. Climate change affects agriculture in many ways, which vary from one region to another. Increasing temperatures and changes in precipitation lead to shifts in the location and incidence of pest outbreaks, as well as the types of pests concerned. Assessing the consequences of climate change with regard to crop losses due to pests is complex, as these losses result from interactions among effects on plant vigouur (due to abiotic stressors), crop yields, pest population changes, and pests' natural enemies (Gregory *et al*. 2009).

Deutsch *et al*. (2018) modelled crop production losses for rice, maize and wheat due to insect pests and found that global yield losses of these grains would increase by 10-25 per cent per degree of global mean surface warming. Losses would be most acute in in temperate regions, where most of these cereals are produced.

Based on published observations of more than 600 insect pests and pathogens, Bebber *et al*. (2013) assessed global shifts in their latitudinal ranges since 1960. They found a significant trend of increasing numbers of pest and pathogen observations at higher latitudes, globally and in both the northern and southern hemispheres. The mean shift in detection since 1960 (26.6 km per decade) was more rapid than that reported for many wild species (17.6 km per decade), but was nearly identical to that expected as a result of temperature changes (27.3 km per decade).

The responses of forest insect pests to climate change were recently reviewed by Jactel, Koricheva,and Castagneyrol (2019), who noted that the complex interplay between abiotic stressors, host trees, insect herbivores and their natural enemies makes it very difficult to predict the overall consequences of climate change for forest health. However, most of the responses of forest insect herbivores to climate change were expected to be positive, with shorter generation time, higher fecundity and greater survival, leading to increased range expansion and outbreaks. The observed positive latitudinal trends in many taxa support the hypothesis of global warming-driven pest movement.

Wilke *et al*. (2019) argue that due to global warming arthropod vectors of human disease (e.g., mosquitos, ticks and sandflies) will expand their ranges, as they will be able to colonize new areas including urban environments and be present for extended periods during the year. This, in turn, will likely drive an increase in the incidence of vector-borne diseases.

Overall, among the main impacts of climate change on agriculture are increased incidence of drought and extreme weather events, more intense pest and disease pressures, and loss of biodiversity. Changes in pest-crop complexes mean there will be a need to better understand the changes that may occur in pest management methods, including management of both synthetic and biological pesticides. Moreover, climate change will increase the potential for movements of pests and diseases, as well as movements of products, from one country to another (FAO 2016).

It is likely that such climate change-driven changes in pest incidence will result in greater pesticide use in agriculture, especially in intensive

agriculture environments, and to protect public health (Delcour, Spanoghe and Uyttendaele 2015; Deutsch *et al*. 2018).

2.7.17 Invasive species and pest outbreaks

Invasive alien species are species that have been intentionally or accidentally introduced into a new ecosystem. They can become problematic for biodiversity, human health, and agriculture as well as other human activities. Invasive species are a driver of biodiversity loss (International Union for Conservation of Nature 2020) and can be a major cause of crop losses (Paini *et al*. 2016). Greater globalization and connectedness increase the threat that invasive species will arrive in countries where they were previously absent.

A recent review of the global threat to agriculture from invasive species indicated that countries which are large agricultural producers (e.g., Brazil, China, India, the United States) will experience the highest potential costs from invasive species. However, the countries with the highest costs from invasive species relative to their GDP were all developing ones, with the top six most vulnerable located in sub-Saharan Africa (Paini *et al*. 2016). Examples of recent serious invasive pests include the Asian tiger mosquito, Panama disease and parthenium weed (Table 2.7-8).

A much publicized case is the spread in Africa of the fall armyworm (Spodoptera frugiperda), a moth indigenous to the Americas which feeds on over 80 different crops including maize, rice, sorghum and sugarcane. The fall armyworm has been reported to cause potential maize yield losses estimated at between USD 2.5 billion and US 6.2 billion per year in just 12 of Africa's maize-producing countries (Day *et al*. 2018). Its invasion of Africa and massive crop losses have led to important government-sponsored control campaigns using large volumes of pesticides (Hruska 2019). In most cases, however, maize yield reductions tend to be much lower, typically ranging from 10 to 30 per cent, and more sustainable pest management approaches are now being developed and implemented (Hruska 2019).

The often sudden appearance of invasive species in agriculture, or in relation to public health, tends to lead to sharp increases in pesticide use in an

Table 2.7-8 **Some examples of recent serious invasive pests.** Centre for Agriculture and Bioscience International (CABI) (2020).

Species		Origin	Current presence	Economic importance
	Fall armyworm (*Spodoptera frugiperda*)	Tropical and subtropical Americas	Africa, Asia	Damage to maize, rice, sorghum, sugarcane and wheat, as well as other vegetable crops and cotton
	Asian tiger mosquito (*Aedes albopictus*)	Southeast Asia, the Pacific, China, Japan and Madagascar	North and South America, Africa, Australia and Europe	Transmission of many human viral diseases, including dengue, West Nile virus and Japanese encephalitis
	Panama disease (*Fusarium oxysporum* f.sp. *cubense*)	Southeast Asia	Asia, Africa, the Americas, Oceania	TR4 isolates of the pathogen threaten production of Cavendish banana cultivars, which produce the bulk of global exports
	Parthenium weed (*Parthenium hysterophorus*)	Americas	Africa, Asia, Europe Oceania	Major weed in rangeland and field crops
	Melon fly (*Bactrocera cucurbitae*)	India (Southeast Asia is its natural range)	Africa, Oceania, Hawaii	Very serious pest affecting cucurbit crops

Box 2.7-5 **The desert locust upsurge, 2019-2020.** FAO (2020d); Cressman, K., FAO (personal communication).

Desert locust upsurges are not a new phenomenon. Locusts are one of the oldest migratory pests in the world. They have wreaked havoc on crops across the globe for centuries. When large swarms infest many countries and spread across several regions or continents, it becomes a plague.

The recent desert locust upsurge (see map)

In **2018** two cyclones brought heavy rains that gave rise to favourable breeding conditions in the Empty Quarter of the southern Arabian Peninsula. As a result, three generations of desert locust breeding occurred, causing an 8,000-fold increase in locust numbers.

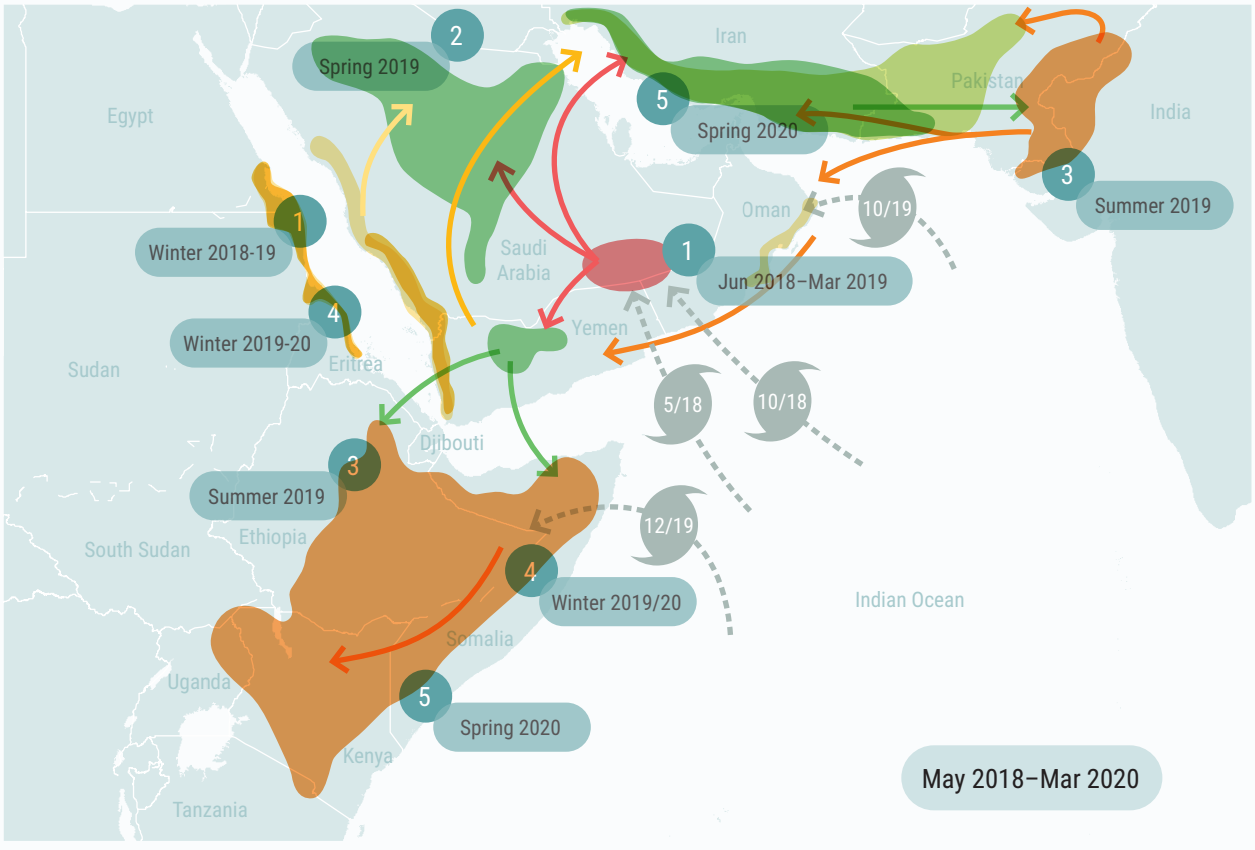
In **2019** locust swarms reached Saudi Arabia, Iran, and Yemen where widespread breeding occurred during the spring. Later that year, swarms invaded the Indo-Pakistan border and also migrated to Somalia and Ethiopia.

© FAO/Sven Torfinn, Walking through a Desert Locust swarm

Despite control operations, massive waves of swarms invaded Kenya in late 2019 and early 2020, and other swarms moved to Eritrea and Djibouti.

During the first half of 2020 two generations of breeding occurred in Ethiopia, Kenya and Somalia as well as in Iran and Pakistan. A few swarms invaded Uganda, the Democratic Republic of the Congo, and South Sudan. In June, swarms started to migrate from the spring to the summer breeding areas.

As a result of the desert locust upsurge, which affected parts of the Horn of Africa and South-West Asia, by mid-2020 an estimated 20 million people were facing severe acute food insecurity.



Given the scale of the locust outbreak, large control operations have been set up by the governments concerned, with support from bi- and multilateral donors and with technical support from FAO.

Between December 2019 and May 2020 approximately 1.4 million hectares were treated with insecticides. Most of these were synthetic chemical insecticides, primarily organophosphates and pyrethroids.

Biocontrol against locusts is conducted using the entomopathogenic fungus *Metarhizium acridum*. Insect growth regulators can also be used and are more target specific than broad-spectrum insecticides.

However, the scale of interventions with such less hazardous products is limited, primarily because they are applied only against the larval stages of the locust, and the operational logistics are more complex.

attempt to control these organisms. Chemical pesticides tend to be the first control measure of choice, as they can be procured and distributed rapidly and can bring about short-term reductions in pest populations. For instance, Pozebon *et al*. (2020) reviewed arthropod invasions in soybean in Brazil and concluded that the introduction of whitefly (Bemisia tabaci complex), two-spotted spider mite (Tetranychus urticae) and cotton bollworm (Helicoverpa armigera) likely led to significant increases in the total amount of insecticide and acaricide used in the country, as well as increased insecticide input per area. They also showed that depending solely on chemical control increased the likelihood of establishment of the invasive pests due to adverse effects on their natural enemies and the development of resistance.

Few invasive pests in agriculture have been controlled through the use of chemical pesticides alone. More sustainable and long-term control often involves the introduction of resistant crop varieties and biological control agents, or the development of integrated pest or vector control approaches. The spread of invasive species can therefore also be an opportunity for research on and development of non-chemical pest management techniques. This will occur especially after the emergency of the invasion has passed and/or the species has established itself to stay.

Endemic pests may also lead to pest outbreaks, even if such organisms have been present in a region for a long time. One of the best known examples is the desert locust (Schistocerca gregaria), which is endemic to semi-arid and arid deserts of northern Africa, the Near East and Southwest Asia where it generally does not pose any risks to crops. During outbreaks the desert locust becomes gregarious, increases in population size and invades much larger areas, where it can cause great damage to crops. The desert locust outbreak in 2019 spread to the Horn of Africa and Southwest Asia (Box 2.7-5). Although the quantities of insecticide used to control such locust upsurges are only a small fraction of global insecticide use, they may constitute a significant increase in pesticide consumption by the countries directly concerned. Furthermore, pest outbreaks such as that of the desert locust may require large numbers of pesticide applicators with no or limited training and experience to be recruited, increasing risks to human health as well as the environment.

2.7.18 Public environmental concerns

As in the case of public health, environmental concerns have also partly shaped the regulation and risk assessments of pesticides and, indirectly, the types and quantities of pesticides used.

The importance of the United States biologist Rachel Carson's path-breaking Silent Spring (Carson 1962), which described the environmental effects of DDT and other organochlorine pesticides, is well known. Public concerns about the environmental effects of many other groups of pesticides followed its publication. More than half a century later, recent scientific and public debates have concentrated on neonicotinoid insecticides and their effects on pollinators and aquatic organisms.

Surveys of the environmental concerns of European citizens show that agricultural pollution (defined as "the use of pesticides, fertilizers etc.") is consistently cited among the most important environmental issues, and that concerns about this issue have increased in the last decade (EC 2011; EC 2014; EC 2020b) (Figure 2.7-7).

There has been a clear and steady increase in the number of published newspaper articles about the environment and pesticides during the last two decades (Figure 2.7-8). A specific example is the steep rise in the number of articles about the environment and honeybees since about 2005, when colony collapse disorder began to be reported and pollinator population reductions were linked to pesticide use.

Public environmental concerns have contributed over time to increased scientific and regulatory scrutiny of certain pesticides and subsequent halting or restriction of their use. For example, local authorities in some countries are moving away from the use of herbicides in public spaces and adopting alternative approaches to control unwanted vegetation.

Figure 2.7-7 **Environmental concerns of European citizens. The percentage of respondents who considered "the use of pesticides, fertilizers, etc" an important environmental issue has slightly increased since the mid-2000s.** EC 2011; EC 2014; EC 2020b).

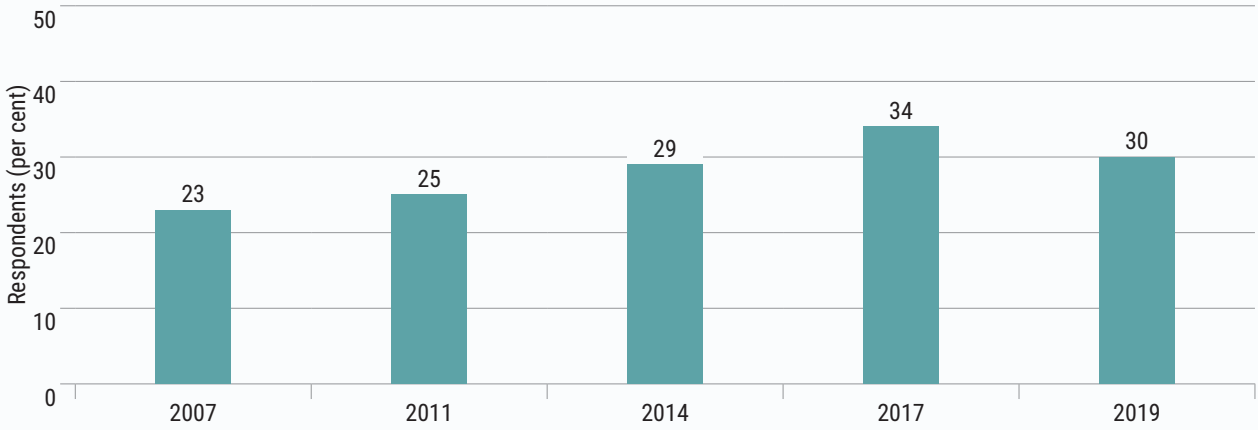
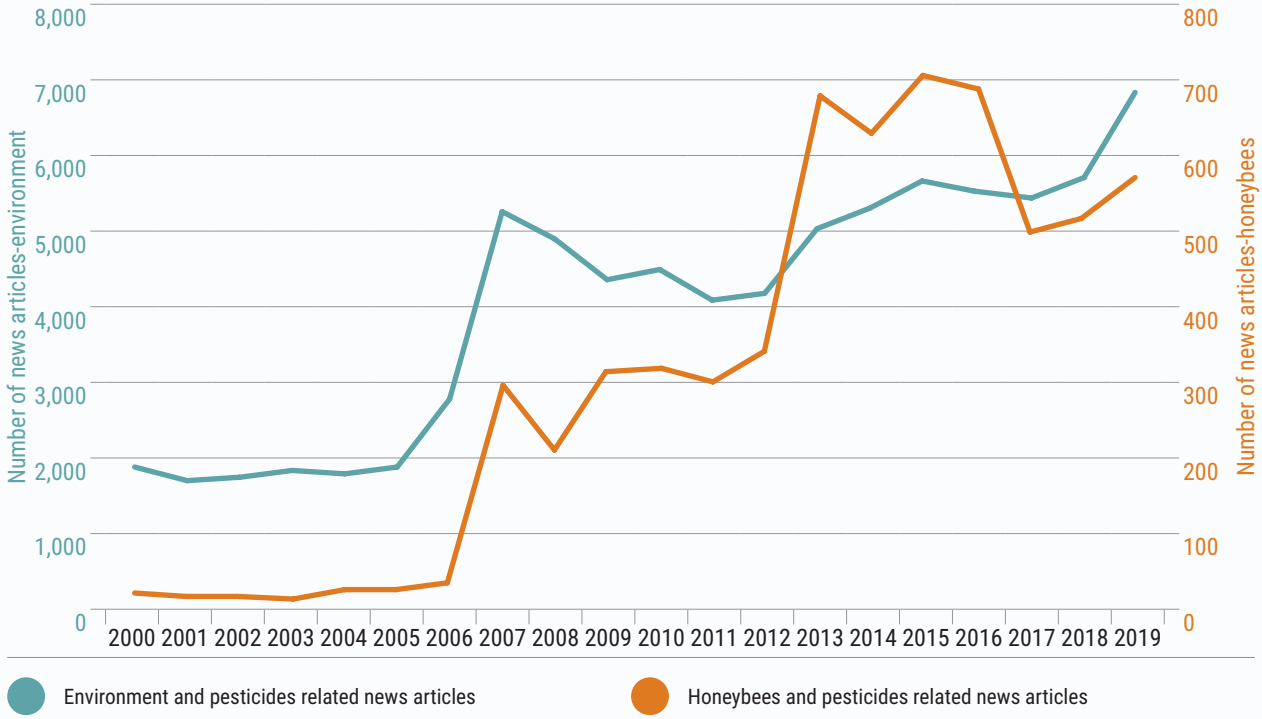


Figure 2.7-8 **Environmental concerns in the news. Shown are the number of newspaper articles about the environment and pesticides, and honeybees and pesticides, published annually in the last 20 years.**



Data obtained from the NexisUni database (LexisNexis 2020) using the search terms *pesticid* AND environment and pesticide* AND (honeybee* or "honey bee*")*, for newspaper articles published globally in English.

Information drivers

The information that pesticide users and others have or lack, for example about whether it is necessary to use pesticides, other pest management options, pesticide risks, and precautionary measures, influences the use of pesticides and of alternative approaches.

2.7.19 Information sources

The information sources available to farmers and other pesticide users may increase or

reduce pesticide use. Information about pest management and pesticides is obtained from various sources, including agricultural extension and advisory services, research institutions, pesticide retailers, pesticide distributors that conduct demonstrations and information sessions, other farmers, and (more recently) public and private digital information providers.

In many countries public extension and advisory services reach only a limited number of farmers, a situation which has worsened in recent decades due to government budget cutbacks and privatization policies. However, farmers increasingly receive technical information from private advisory services or from pesticide distributors and retailers. As most smallholder farmers in low and lower-middle income countries do not have access to private advisory services, information about pest management options in these countries is mainly obtained from pesticide shops and other retail outlets. Recent examples are provided by Haj-Younes, Huici and Jørs (2015) (Bolivia); Jin, Bluemling and Mol (2015) and Sun *et al*. (2019) (China); and Okonya and Kroschel (2015) (Uganda).

It is important for pesticide retailers to give adequate technical advice on the judicious handling and use of the products they sell. Nevertheless, if they are the main information sources influencing farmers' pest management choices, there is a risk that their advice will be biased towards the pesticides they sell rather than alternative pest management options. Independent advisory services may be likely to provide more objective information, including about different (integrated) pest management approaches.

The impact of information sources on the choice of pest management options has not been systematically reviewed, but various studies indicate that information bias does affect farmers' pesticide use. Schreinemachers *et al*. (2017), who studied vegetable farmers in Cambodia, Laos and Viet Nam, observed that those who sought advice from friends and neighbours used 45 per cent less pesticides, while those who sought advice from pesticide shopkeepers used 250 per cent more than the average. They also calculated that the risk of pesticide overuse was lower if farmers had received advice from extension agents and significantly higher if the advice came from pesticide retailers (Schreinemachers *et al*. 2020). In Ghana, Anang and Amikuzuno (2015) found that a higher number of extension contacts tended to reduce the use of pesticides in rice farming. In Hebei Province, China, 87 per cent of smallholder farmers used double or more the recommended amount of pesticides on cotton; farmers who received information mainly from pesticide retailers applied pesticides at 20-70 per cent higher rates than those who were part of a cooperative with extension support and also applied pesticides twice as often (Jin *et al*. 2015). Similar results were found in an analysis of pest management recommendations in Beijing Municipality, China. Although chemical pest management was emphasized by both business-linked and independent extension workers, business-linked advisors recommended the use of pesticides 18 per cent more often than advisors who were not business-linked (Wan *et al*. 2019).

2.7.20 Knowledge, awareness and attitudes

Improving pesticide users' knowledge about proper handling and application of pesticide products, and creating awareness of their risks to the environment and human health, are often considered essential for risk reduction. However, while training farmers and other pesticide users generally succeeds in increasing knowledge about good pesticide application and awareness of risks, knowing about precautionary measures and risks does not necessarily lead to effective changes in behaviour and use of good practices, as demonstrated in reviews by Remoundou *et al*. (2014) and Ricci *et al*. (2016). Schreinemachers *et al*. (2017) reported that in Cambodia, Laos and Viet Nam pesticide use was not reduced by previous training in pest management, greater awareness of adverse health effects, or good agricultural practices.

It is often found that despite farmers having considerable knowledge about (and a positive attitude towards) the use of personal protective equipment (PPE), they may use this equipment to a limited extent if at all (Pasiani *et al*. 2012; Remoundou *et al*. 2014; Yuantari *et al*. 2015;

Gesesew *et al*. 2016). The reasons include lack of availability, high cost, discomfort, habits of personal behaviour and limited social acceptability, among others).

Farmers' knowledge about pests and their natural enemies, and a basic understanding of pest ecology in a crop, have been shown to influence pesticide use. Wyckhuys *et al*. (2019) reviewed the "ecological literacy" of farmers (i.e., their understanding of insect pests and their natural enemies). They found that globally, across studies, farmers' dependency on synthetic insecticides showed a downward (although statistically non-significant) trend with increasing ecological literacy and increasing levels of technical knowledge. In a separate study on vegetable farmers in Cambodia, Laos and Viet Nam, Schreinemachers *et al*. (2017) found that greater knowledge of beneficial and harmful arthropods was associated with lower levels of pesticide use.

Gender differences in knowledge about pest management and pesticide use are also often observed. Women and men from the same community may perceive the identity and importance of pests in different ways; men and women may also apply different control methods; or women may not be invited to participate in training about the judicious pesticide use (Kawarazuka *et al*. 2020). For instance, Ochago (2018) found that men and women coffee farmers in Uganda were equally aware of the adverse effects of pesticides on human health, but more men than women were knowledgeable about pest control methods including IPM.

However, addressing gaps in women farmers' knowledge is not necessarily sufficient to improve pest management, as these may also be limited by gender-related socio-economic constraints. For instance, a study conducted in Ethiopia showed that women farmers were unable to spray their potato fields, primarily due to less access to financial resources than men; this in spite of having been included in training exercises (Damtew *et al*. 2020).

A focus on quick-fix solutions to pest and disease problems may lead to increased pesticide use. Farmers and other pesticide users throughout the world are inclined to choose rapidly acting pesticides, with clearly visible results, as a preferred pest management approach (Alwang, Norton and Larochelle 2019). This is especially the case if pest control is reactive, attempting to reduce crop damage, rather than preventive, focusing on minimizing the build-up of pest or disease populations. For instance, 66 per cent of vegetable farmers in Cambodia, Laos and Viet Nam agreed with the statement that good pesticides are those that kill all insects immediately (Schreinemachers *et al*. 2017).

In conclusion, information and knowledge are rarely the only drivers of pesticide use. Psychological, social, economic and environmental factors play important roles. Providing the right information and building pest management capacity are essential, but are insufficient to change behaviour.

2.7.21 Training

Different types of training are of relevance to pesticide use. Training may focus on reducing the occupational and environmental risks of handling and applying pesticides (sometimes referred to as "safe use training"). This type of training generally does not question the need to use pesticides, but attempts to ensure that they are handled and applied judiciously. At the other end of the spectrum, training is conducted on biointensive integrated pest management or agroecology, which explicitly aim to reduce reliance on pesticides and make maximum use of natural pest management processes.

In some parts of the world pesticide users and retailers are required to successfully complete training as a step towards certification. In many low and middle income countries training is not mandatory before pesticides can be sold or used. Therefore, the large majority of farmers and other pesticide users in those countries will not have had training on judicious pesticide use, let alone broader training on sustainable pest management.

In many cases training on the occupational health and safety of pesticide use has been shown to improve knowledge about pesticide risks, increase pesticide risk perception, and improve attitudes to

good pesticide handling and application (Sam *et al*. 2008; LeProvost *et al*. 2014; Ricci *et al*. 2016; Schreinemachers *et al*. 2016; Vaidya *et al*. 2017). Training on pesticide application has sometimes also led to reductions in exposure or health effects. For instance, Levesque, Arif and Shen (2012) found that farm workers in North Carolina (United States) who had received short pesticide safety training were almost five times more likely to wear PPE than those who had not; nevertheless, more than a quarter of the farm workers studied would still not use PPE. Damalas and Koutroubas (2017) found that pesticide safety training not only improved the knowledge and attitudes of cotton farmers in Greece, but was also accompanied by better safety behaviours. Fuhriman *et al*. (2020) observed a 33 per cent reduction in the exposure scores of farm workers in Costa Rica who had received training compared to those who had not.

Reviews of training on (pesticide) occupational health and safety aimed at improving behaviours and reducing risks have generally concluded that the effectiveness of such interventions is low if they take place in isolation (Lehtola *et al*. 2008; Ricci *et al*. 2016). This may be due in part to the often incidental nature of the training and its short duration. Increased knowledge and better risk perception are generally not enough to change behaviours. The gap between knowledge and practice needs to be bridged using a more interactive and participatory training model (Yuantari *et al*. 2015).

A highly participatory model which has been applied to build capacity in integrated pest management is farmer field schools (FFS). This approach emphasizes hands-on learning to improve the skills and knowledge of groups of farmers in order to help them adapt practices to their specific context (FAO 2020e). While FFS have generally succeeded in changing the behaviours of farmers and reducing pesticide use, they are relatively resource and time intensive compared to "classroom" training, demonstrations and media messages. Upscaling of FFS programmes to cover the majority of farmers in a given country or region has taken place to only a limited extent (Waddington *et al*. 2014).

References

Accenture (2019). *Supply Chain Traceability in Agrochemicals*. Study commissioned by CropLife International. https://croplife.org/wp-content/uploads/2019/11/Accenture_CropLife_SC_Traceability_Final.pdf.

AgbioInvestor (2019). *Trends in Crop Protection and Seed Market Development*. Presentation by Dr Matthew Phillips at CropLife America Convention, Antigua, Guatemala, 9 April 2019. Midlothian: AgbioInvestor – Agricultural Market Intelligence. https://agbioinvestor.com.

Agrow (2016). *Annual Review 2016*. https://agribusinessintelligence.informa.com/resources/product-content/agrow-annual-review-2016.

Agrow (2018). *Agrow Global Biopesticide Regulations 2018*. http://agribusiness.intelligence.informa.com/agrow-global-biopesticide-regulations-report.

Agrow (2019). Global crop protection market rose 6 per cent in 2018, May 2019. https://agrow.agribusinessintelligence.informa.com/AG031247/Global-crop-protection-market-rose-6-in-2018. Accessed 5 November 2020.

Agrow (2020). *Annual Review 2019*. London: IHS Markit, Agrow Agribusiness.

Ahmed, A.U., Hoddinott, J.F., Islam, K.M.S., Khan, A.S.M.M.R., Abedin, N. and Hossain, N.Z. (2019). *Impacts of Bt Brinjal (Eggplant) Technology in Bangladesh*. Dhaka: International Food Policy Research Institute-Bangladesh Policy Research and Strategy Support Program. https://www.ifpri.org/publication/impacts-bt-brinjal-eggplant-technology-bangladesh.

Alliance for Malaria Prevention (2020). Net Mapping Project. https://allianceformalariaprevention.com/net-mapping-project/. Accessed 5 November 2020.

Alwang. J., Norton, G. and Larochelle, C. (2019). Obstacles to widespread diffusion of IPM in developing countries: Lessons from the field. *Journal of Integrated Pest Management* 10(1), 1-8. https://doi.org/10.1093/jipm/pmz008.

Anang, B.T. and Amikuzuno, J. (2015). Factors influencing pesticide use in smallholder rice production in northern Ghana. *Agriculture, Forestry and Fisheries* 4(2), 77-82. https://doi.org/10.11648/j.aff.20150402.19.

Barzman, M., Bàrberi, P., Birch, A.N.E., Boonekamp, P., Dachbrodt-Saaydeh, S., Graf, B. *et al*. (2015). Eight principles of integrated pest management. *Agronomy for Sustainable Development* 35, 1199-1215. https://doi.org/10.1007/s13593-015-0327-9.

Bebber, D.P., Ramotowski, M.A.T. and Gurr, S.J. (2013). Crop pests and pathogens move polewards in a warming world. *Nature Climate Change* 3(11), 985-988. https://doi.org/10.1038/nclimate1990.

Benbrook, C.M. (2016). Trends in glyphosate herbicide use in the United States and globally. *Environmental Sciences Europe* 28, 3. https://doi.org/10.1186/s12302-016-0070-0.

Benbrook, C.M., Growth, E, Halloran, J.M., Hansen, M. and Marquardt, S. (1996). *Pest Management at the Crossroads*. Yonkers, NY: Consumers Union.

Betancur, M. (2020). Brazil fighting pesticide smuggling war on two fronts, 6 January. *InSight Crime*. https://www.insightcrime.org/news/brief/brazil-pesticide-war-two-fronts/. Accessed 7 November 2020.

Better Cotton Initiative (2018). Better Cotton Principles and Criteria V2.1. https://bettercotton.org/better-cotton-standard-system/production-principles-and-criteria/. Accessed 7 November 2020.

BNNVARA (2019). Local residents of bulb cultivation areas more exposed to pesticides. Zembla (investigative television programme, in Dutch). https://www.bnnvara.nl/zembla/artikelen/omwonenden-bollenvelden-hoger-blootgesteld-aan-bestrijdingsmiddelen. Accessed 7 November 2020.

Böcker, T. and Finger, R. (2016). European pesticide tax schemes in comparison: An analysis of experiences and developments. *Sustainability* 8, 378. https://doi.org/10.3390/su8040378.

Bommarco, R., Kleijn, D. and Potts, S.G. (2013). Ecological intensification: Harnessing ecosystem services for food security. *Trends in Ecology and Evolution* 28(4), 230-238. https://doi.org/10.1016/j.tree.2012.10.012.

Boote, K.J. (2017). *Impacts of Disease and Pest Crop Losses on Crop Yields and Agrosystem Performances: International Conference on Crop Losses Global Crop Losses Caused by Diseases, Pests and Weeds, Paris, 16-18 October 2017*. https://iapps2010.me/2018/06/29/conference-on-global-crop-losses/.

Bottrell, D.G. and Schoenly, K.G. (2018). Integrated pest management for resource-limited farmers: Challenges for achieving ecological, social and economic sustainability. *The Journal of Agricultural Science* 156(3), 408-426. https://doi.org/10.1017/S0021859618000473.

Brookes, G. and Barfoot, P. (2018). Environmental impacts of genetically modified (GM) crop use 1996-2016: Impacts on pesticide use and carbon emissions. *GM Crops and Food* 9(3), 109-139. https://doi.org/10.1080/21645698.2018.1476792.

Bullock, J. (2020a). Challenging the formulator: Biocontrol and conventional crop protection. *Outlooks on Pest Management* 31(3), 132-136. https://doi.org/10.1564/v31_jun_10.

Bullock, J. (2020b). What's new in formulation: A review of approaches for biopesticides and conventional agrochemicals, May 2020. *Online presentation*. *IFormulate*. https://iformulate.biz/iformulate-revisited-whats-new-in-agrochemical-formulation/. Accessed 7 November 2020.

Cameron, P.J., Walker, G.P., Hodson, A.J., Kale, A.J. and Herman, T.J.B. (2009). Trends in IPM and insecticide use in processing tomatoes in New Zealand. *Crop Protection* 28(5), 421-427. https://doi.org/doi:10.1016/j.cropro.2009.01.002.

Carson, R. (1962). *Silent Spring*. Boston, MA: Houghton Mifflin Company. http://www.rachelcarson.org/SilentSpring.aspx.

Carvalho, F.K., Chechetto, R.G., Mota, A.A.B. and Antuniassi, U.R. (2020). Challenges of aircraft and drone spray applications. *Outlooks on Pest Management* 31(2), 83-88 https://doi.org/10.1564/v31_apr_07.

Centre for Agriculture and Science International (2020). Invasive species compendium. https://www.cabi.org/ISC. Accessed 25 April 2021.

Chittenden, F.H. (1899). *Some Insects Injurious to Garden and Orchard Crops*. United States Department of Agriculture, Division of Entomology. https://ufdc.ufl.edu/AA00018858/00001/1j.

Clarke, R. (2010). *Private Food Safety Standards: Their Role in Food Safety Regulation and their Impact*. Rome: Food and Agriculture Organization of the United Nations. http://www.fao.org/3/ap236e/ap236e.pdf.

Clausen, A.S., Jørs, E., Atuhaire, A. and Thomsen, J.F. (2017). Effect of integrated pest management training on Ugandan small-scale farmers. *Environmental Health Insights* 11, 1-10. https://doi.org/10.1177/1178630217703391.

Codex Alimentarius Commission (2013). *Guidelines for the Production, Processing, Labelling and Marketing of Organically Produced Foods*. Rome: Joint FAO/WHO Food Standards Programme. https://www.fao.org/3/at715e/at715e.pdf.

Coupe, R.H. and Capel, P.D. (2016). Trends in pesticide use on soybean, corn and cotton since the introduction of major genetically modified crops in the United States. *Pest Management Science* 72, 1013-1022. https://doi.org/10.1002/ps.4082.

CPI Inflation Calculator (2020). Value of $100 from 1995 to 2014. https://www.in2013dollars.com. Accessed 7 November 2020.

Creissen, H.E., Jones, P.J., Tranter, R.B., Girling, R.D., Jess, S., Burnett, F.J. *et al*. (2019). Measuring the unmeasurable? A method to quantify adoption of integrated pest management practices in temperate arable farming systems. *Pest Management Science* 75(12), 3144-3152. https://doi.org/10.1002/ps.5428.

CropLife International (2017). Crop protection industry supports FAO on IPM. https://croplife.org/case-study/crop-protection-industry-supports-fao-on-ipm/. Accessed 7 November 2020.

CropLife International (n.d.). Anti-counterfeiting. https://croplife.org/crop-protection/anti-counterfeiting. Accessed 7 November 2020.

Damalas, C.A. and Koutroubas, S.D. (2017). Farmers' training on pesticide use is associated with elevated safety behavior. *Toxics* 5(3), 19. https://doi.org/10.3390/toxics5030019. Accessed 7 November 2020.

Damtew, E., van Mierlo, B., Lie, R., Struik, P., Leeuwis, C., Lemaga, B. and Smart, C. (2020). Governing a collective bad: social learning in the management of crop diseases. *Systemic Practice and Action Research* 33, 111–134. https://doi.org/10.1007/s11213-019-09518-4.

Day, R., Burgoyne, J., Coles, W., Hirschfeld, M., Reid, D., Winks, R. and Rangi, D. (2018). *Invasive Species: The Hidden Threat to Sustainable Development*. Wallingford, UK: CABI. https://www.invasive-species.org/wp-content/uploads/sites/2/2019/02/Invasive-Species-The-hidden-threat-to-sustainable-development.pdf.

DeFries, R., Fanzo, J., Mondal, P., Remans, R. and Wood, S.A. (2017). Is voluntary certification of tropical agricultural commodities achieving sustainability goals for small-scale producers? A review of the evidence. *Environmental Research Letters* 12, 033001. https://doi.org/10.1088/1748-9326/aa625e.

Delcour, I., Spanoghe, P. and Uyttendaele, M. (2015). Literature review: Impact of climate change on pesticide use. *Food Research International* 68, 7-15. https://doi.org/10.1016/j.foodres.2014.09.030.

Deutsch, C.A., Tewksbury, J.J., Tigchelaar, M., Battisti, D.S., Merrill, S.C., Huey, R.B. and Naylor, R.L. (2018). Increase in crop losses to insect pests in a warming climate. *Science* 361(6405), 916-919. https://doi.org/10.1126/science.aat3466.

Diarra, A. and Haggblade, S. (2017). *National Implementation of Regional Pesticide Policies in West Africa: Achievements, Challenges and Priorities*. *Synthesis Report*. Michigan: Food Security Policy. https://www.canr.msu.edu/resources/national-implementation-of-regional-pesticide-policies-in-west-africa-achievements-challenges-and-priorities-synthesis-report.

Direction de l'Epidémiologie et de la Lutte contre les Maladies (2019). Plan national 2019-2025 de Lutte contre les vecteurs. Rabat, Morocco: Ministère de la Santé. Accessed 7 November 2020.

du Jardin, P. (2015). Plant biostimulants: Definition, concept, main categories and regulation. *Scientia Horticulturae* 196, 3-14. https://doi.org/10.1016/j.scienta.2015.09.021.

DunhamTrimmer (2019). *DunhamTrimmer's Global Biocontrol Market: Overview, Trends, Drivers and Insights*. *Presentation*. *2019*. Lakewood Ranch, FL: DunhamTrimmer LLC. http://dunhamtrimmer.com/wp-content/uploads/2019/05/TOC-DT_Global_Biocontrol_Overview_Links.pdf.

Dunlap, R.E. and Beus, C.E. (1992). Understanding public concerns about pesticides: An empirical examination. *The Journal of Consumer Affairs* 26 (2), 418-461. https://doi.org/10.1111/j.1745-6606.1992.tb00035.x.

Ecratum (2020). Safe Quality Food Standard (SQF) Certification - What you need to know. https://blog.ecratum.com/safe-quality-food-standard-sqf-certification-what-you-need-to-know.

Ehler, L.E. (2006). Integrated pest management (IPM): Definition, historical development and implementation, and the other IPM. *Pest Management Science* 62, 787-789. http://doi.org/10.1002/ps.1247.

Escalada, M.M., Heong, K.L., Huan, N.H. and Chien, H.V. (2009). Changes in rice farmers' pest management beliefs and practices in Vietnam: An analytical review of survey data from 1992 to 2007. In *Planthoppers: New Threats to the Sustainability of Intensive Rice Production Systems in Asia*. Heong, K.L. and and Hardy, B. (eds.). Los Baños, the Philippines: International Rice Research Institute. 447-456. https://ideas.repec.org/b/ags/irricg/281811.html.

Esker, P.D., Savary, S. and McRoberts, N. (2013). Crop loss analysis and global food supply: Focusing now on required harvests. *CAB Reviews Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources* 7(052), 1-14. https://www.researchgate.net/publication/235255555_Crop_loss_analysis_and_global_food_supply_focusing_now_on_required_harvests.

European Commission (2010). *Food-related Risks*. Special Eurobarometer 354. https://ec.europa.eu/commfrontoffice/publicopinion/index.cfm/Survey/index#p=1&search=food.

European Commission (2011). *Attitudes of European Citizens Towards the Environment*. *Summary*. Special Eurobarometer No. 365. https://ec.europa.eu/commfrontoffice/publicopinion/archives/ebs/ebs_365_sum_en.pdf.

European Commission (2014). *Attitudes of European Citizens Towards the Environment*. Special Eurobarometer No. 416. Brussels: Directorate General for Communication. https://ec.europa.eu/commfrontoffice/publicopinion/archives/ebs/ebs_416_en.pdf.

European Commission (2020a). *Report from the Commission to the European Parliament and the Council on the Experience Gained by Member States on the Implementation of National Targets Established in their National Action Plans and on Progress in the Implementation of Directive 2009/128/EC on the Sustainable Use of Pesticides*. https://ec.europa.eu/food/sites/food/files/plant/docs/pesticides_sud_report-act_2020_en.pdf.

European Commission (2020b). *Attitudes of European Citizens Towards the Environment*. Special Eurobarometer No. 501. Brussels: Directorate General for Communication. https://data.europa.eu/euodp/en/data/dataset/S2257_92_4_501_ENG.

European Commission (n.d.). *Pesticides in the European Union – Authorisation and Use*. Factsheet. Brussels. https://ec.europa.eu/food/sites/food/files/plant/docs/pesticides_approval-factsheet.pdf.

European Crop Protection Association (n.d.). Counterfeit & Illegal pesticides. https://www.ecpa.eu/stewardship/counterfeit-illegal-pesticides#section11433. Accessed 7 November 2020.

European Union (2009a). Directive 2009/128/EC of the European Parliament and of the Council of 21 October 2009 establishing a framework for Community action to achieve the sustainable use of pesticides. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32009L0128. Accessed 7 November 2020.

European Union (2009b). Regulation (EC) No 1107/2009 of the European Parliament and of the Council of 21 October 2009 concerning the placing of plant protection products on the market and repealing Council Directives 79/117/EEC and 91/414/EEC. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A02009R1107-20210327. Accessed 7 November 2020.

European Union Agency for Law Enforcement Cooperation (2019). Operation Silver Axe strikes for the fourth time seizing over 550 tonnes of illegal pesticides. Europol press release, 18 June. https://www.europol.europa.eu/newsroom/news/operation-silver-axe-strikes-for-fourth-time-seizing-over-550-tonnes-of-illegal-pesticides. Accessed 7 November 2020.

European Union Intellectual Property Office (2017). *The Economic Cost of IPR Infringement in the Pesticides Sector. Quantification of Infringement in Manufacture of Pesticides and Other Agrochemical Products*. https://euipo.europa.eu/tunnel-web/secure/webdav/guest/document_library/observatory/resources/research-and-studies/ip_infringement/study10/pesticides_sector_en.pdf. Accessed 7 November 2020.

Eurostat (2018). Statistics explained. Agri-environmental indicator – consumption of pesticides. Data from August 2018. European Union. https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Agri-environmental_indicator_-_consumption_of_pesticides. Accessed 4 October 2018.

Fairtrade (2014). *Fairtrade Standard for Hired Labour*. Current version: 15.01.2014_v1.6. https://files.fairtrade.net/standards/HL_EN.pdf.

Fairtrade (2019). *Fairtrade Standard for Small-scale Producer Organizations*. Current version: 03.04.2019_v2.3. https://files.fairtrade.net/standards/SPO_EN.pdf.

FAOSTAT (2019). Pesticides use (http://www.fao.org/faostat/en/#data/RP); Land use (http://www.fao.org/faostat/en/#data/RL) – Area under Crops ; Value of agricultural production (http://www.fao.org/faostat/en/#data/QV) – Crops. Food and Agriculture Organization of the United Nations. Accessed April-June 2019.

FAOSTAT (2020). Food and agriculture data. Food and Agriculture Organization of the United Nations. http://www.fao.org/faostat/en/#home. Accessed 8 November 2020.

Farah, J. (1994). *Pesticide Policies in Developing Countries – Do They Encourage Excessive Use?* World Bank Discussion Papers No. 238. Washington, D.C. https://doi.org/10.1596/0-8213-2830-1. Accessed 8 November 2020.

Farrar, J.J., Baur, M.E. and Elliott, S.F. (2015). *Adoption and Impacts of Integrated Pest Management in Agriculture in the Western United States*. Davis, CA: Western IPM Center. http://westernipm.org/index.cfm/about-the-center/publications/special-reports/adoption-and-impact-of-ipm-in-western-agriculture/.

Farrar, J.J., Baur, M.E. and Elliott, S.F. (2016a). Adoption of IPM practices in grape, tree fruit, and nut production in the Western United States. *Journal of Integrated Pest Management* 7(1), 8. https://doi.org/10.1093/jipm/pmw007.

Farrar, J.J., Baur, M.E. and Elliott, S.F. (2016b). Measuring IPM impacts in California and Arizona. *Journal of Integrated Pest Management* 7(1), 13. https://doi.org/10.1093/jipm/pmw012.

Faux, R. (2020). I want to grow healthy food for you, here is what's stopping me, 30 April. *Pesticide Action Network North America*. http://www.panna.org/blog/i-want-grow-healthy-food-you-here-whats-stopping-me.

Federation of Indian Chambers of Commerce and Industry (2015). *Study on Sub-standard, Spurious/Counterfeit Pesticides in India*. *2015 Report*. Federation of Indian Chambers of Commerce and Industry (FICCI) in association with Tata Strategic Management Group, New Delhi. https://croplife.org/wp-content/uploads/2015/10/Study-on-sub-standard-spurious-counterfeit-pesticides-in-India.pdf.

Fernandez-Cornejo, J., Nehring, R., Osteen, C., Wechsler, S., Marti, A. and Vialou, A. (2014). *Pesticide Use in U.S. Agriculture: 21 Selected Crops, 1960-2008*. Economic Information Bulletin 124. United States Department of Agriculture. https://doi.org/10.2139/ssrn.2502986.

Food and Agriculture Organization of the United Nations (2011). *Save and Grow: A Policy Maker's Guide to the Sustainable Intensification of Smallholder Crop Production*. http://www.fao.org/ag/save-and-grow/en/index.html.

Food and Agriculture Organization of the United Nations (2013). *Empowering Farmers to Reduce Pesticide Risks*. Bangkok: FAO Regional IPM/Pesticide Risk Reduction Programme in Asia, FAO Regional Office for Asia and the Pacific. https://www.researchgate.net/publication/259080275_Empowering_Farmers_to_Reduce_Pesticide_Risks.

Food and Agriculture Organization of the United Nations (2016). *The State of Food and Agriculture 2016: Climate Change, Agriculture and Food Security*. http://www.fao.org/publications/sofa/2016/en/.

Food and Agriculture Organization of the United Nations (2017). *Voluntary Sustainability Standards in Agriculture, Fisheries and Forestry Trade*. Trade Policy Brief No. 30. Rome. http://www.fao.org/3/I8843EN/i8843en.pdf.

Food and Agriculture Organization of the United Nations (2019a). Price data collected by FAO. Statistics Division. http://www.fao.org/economic/the-statistics-division-ess/methodology/methodology-systems/price-statistics-and-index-numbers-of-agricultural-production-and-prices/3-price-data-collected-by-fao/en/. Accessed 16 October 2019.

Food and Agriculture Organization of the United Nations (2019b). *The State of the World's Biodiversity for Food and Agriculture*. Bélanger, J. and Pilling, D. (eds.). Rome: FAO Commission on Genetic Resources for Food and Agriculture. http://www.fao.org/3/CA3129EN/CA3129EN.pdf.

Food and Agriculture Organization of the United Nations (2020a). Pesticide application technology. http://www.fao.org/agriculture/crops/thematic-sitemap/theme/spi/agricultural-mechanization/agricultural-mechanization-technologies-and-equipment/pesticide-application-technology/en/. Accessed 24 April 2021.

Food and Agriculture Organization of the United Nations (2020b). Integrated Pest Management. http://www.fao.org/agriculture/crops/thematic-sitemap/theme/pests/ipm/en/. Accessed 7 November 2020.

Food and Agriculture Organization of the United Nations (2020c). Organic agriculture. http://www.fao.org/organicag/oa-faq/oa-faq1/en/. Accessed 7 November 2020.

Food and Agriculture Organization of the United Nations (2020d). Locust watch – desert locust. http://www.fao.org/ag/locusts/en/info/info/index.html. Accessed 7 November 2020.

Food and Agriculture Organization of the United Nations (2020e). Global farmer field school platform. http://www.fao.org/farmer-field-schools/home/en/. Accessed 7 November 2020.

Food and Agriculture Organization of the United Nations and World Health Organization (2013). *State of the Art on the Initiatives and Activities Relevant to Risk Assessment and Risk Management of Nanotechnologies in the Food and Agriculture Sectors*. http://www.fao.org/3/i3281e/i3281e.pdf.

Food and Agriculture Organization of the United Nations and World Health Organization (2014). *The International Code of Conduct on Pesticide Management*. http://www.fao.org/fileadmin/templates/agphome/documents/Pests_Pesticides/Code/Code_ENG_2017updated.pdf.

Food and Agriculture Organization of the United Nations and World Health Organization (2016). *Manual on Development and Use of FAO and WHO Specifications for Pesticides*. First edition, third revision. Prepared by the FAO/WHO Joint Meeting on Pesticide Specifications (JMPS). FAO Plant Production and Protection Paper 228. Geneva and Rome. https://apps.who.int/iris/bitstream/handle/10665/246192/WHO-HTM-NTD-WHOPES-2016.4-eng.pdf;jsessionid=1A373AF1B461FED50B2F0D971A3A6AF5?sequence=1.

Food and Agriculture Organization of the United Nations and World Health Organization (2017). *International Code of Conduct on Pesticide Management*. *Guidelines for the Registration of Microbial, Botanical and Semiochemical Pest Control Agents for Plant Protection and Public Health Uses*. Rome. http://www.fao.org/3/a-i8091e.pdf.

Food Chain Evaluation Consortium (2015). *Ad-hoc Study on the Trade of Illegal and Counterfeit Pesticides in the EU. Executive Summary*. Prepared for the European Commission, Directorate General for Health and Food Safety (SANCO). https://ec.europa.eu/food/sites/food/files/plant/docs/pesticides_ppp_illegal-ppps-study.pdf.

Fresco, L.O. (2016). *Hamburgers in Paradise: The Stories Behind the Food We Eat*. Princeton NJ: Princeton University Press. https://press.princeton.edu/books/hardcover/9780691163871/hamburgers-in-paradise.

Fuhrimann, S., Staudacher, P., Lindh, C., van Wendel de Joode., Mora, A.M., Winkler, M.S. and Kromhout, H. (2020). Variability and predictors of weekly pesticide exposure in applicators from organic, sustainable and conventional smallholder farms in Costa Rica. *Occupational and Environmental Medicine* 77, 40-47. https://doi.org/10.1136/oemed-2019-105884.

Garibaldi, L.A., Pérez-Méndez, N., Garratt, M.P.D., Gemmill-Herren, B., Miguez, F.E. and Dicks, L.V. (2019). Policies for ecological intensification of crop production. *Trends in Ecology and Evolution* 34(4), 282-286. https://doi.org/10.1016/j.tree.2019.01.003.

Gautam, S., Schreinemachers, P., Uddin, M.N. and Srinivasan, R. (2017). Impact of training vegetable farmers in Bangladesh in integrated pest management (IPM). *Crop Protection* 10, 161e169. https://doi.org/10.1016/j.cropro.2017.08.022.

German Federal Office of Consumer Protection and Food Safety (2019). Co-formulants and formulation chemistry. https://www.bvl.bund.de/EN/Tasks/04_Plant_protection_products/01_ppp_tasks/08_ProductChemistry/01_ppp_coformulants_formulationChemistry/ppp_coformulants_formulationChemistry_node.html. Accessed 2 January 2021.

Gesesew, H.A., Woldemichael, K., Massa, D. and Mwanri, L. (2016). Farmers knowledge, attitudes, practices and health problems associated with pesticide use in rural irrigation villages, Southwest Ethiopia. *PLoS ONE* 11(9), e0162527. https://doi.org/10.1371/journal.pone.0162527.

Ghislain, M., Byarugaba, A.A., Magembe, E., Njoroge, A.W., Rivera, C., Roman, M.L. *et al*. (2018). Stacking three late blight resistance genes from wild species directly into African highland potato varieties confers complete field resistance to local blight races. *Plant Biotechnology Journal* 17, 1119-1129. https://doi.org/10.1111/pbi.13042.

Glare, T., Caradus, J., Gelernter, W., Jackson, T., Keyhani, N., Köhl, J. *et al*. (2012). Have biopesticides come of age? *Trends in Biotechnology* 30(5), 250-258. https://doi.org/10.1016/j.tibtech.2012.01.003.

GlobalG.A.P. (2018). Integrated farm assurance, all farm base, crops base, fruits and vegetables, checklist. version 5.1. https://www.globalgap.org/uk_en/for-producers/globalg.a.p./integrated-farm-assurance-ifa/crops/. Accessed 8 November 2020.

Godfray, H.C.J. and Garnett, T. (2014). Food security and sustainable intensification. *Philosophical Transactions of the Royal Society B* 369, 20120273. http://dx.doi.org/10.1098/rstb.2012.0273.

Government of Mozambique (2007). *Pesticides Management Regulation*. Council of Ministers, Decree 6/2009 of 31 March. https://www.masa.gov.mz/wp-content/uploads/2018/06/PESTICIDES-MANAGEMENT-REGUTATION.pdf.

Gregory, P.J., Johnson, S.N., Newton, A.C. and Ingram, J.S.I. (2009). Integrating pests and pathogens into the climate change/food security debate. *Journal of Experimental Botany* 60(10), 2827-2838. https://doi.org/10.1093/jxb/erp080.

Guyer, R. and Davreux, C. (2012). *The problem of Counterfeit and Illegal Pesticides in Africa Middle East*. Presentation. CropLife Africa Middle East and CropLife Canada. https://tpsalliance.org/pdf/topics/The%20problem%20of%20Illegal%20and%20counterfeit%20Pesticides%20in%20Africa%20Middle%20East.pdf.

Haggblade, A., Diarra, A., Jiang, W., Assima, A., Keita, N., Traore, A. and Traore, M. (2019). Fraudulent pesticides in West Africa: A quality assessment of glyphosate products in Mali. *International Journal of Pest Management*. https://doi.org/10.1080/09670874.2019.1668076.

Haj-Younes, J., Huici, O. and Jørs, E. (2015). Sale, storage and use of legal, illegal and obsolete pesticides in Bolivia. *Cogent Food and Agriculture* 1, 1008860. http://dx.doi.org/10.1080/23311932.2015.1008860.

Hansen, J.H.H. (2017). *Experiences with the New Danish Tax on Pesticides*. Presentation given at the Institute for European Environmental Policy (IEEP) Conference on Capacity Building for Environmental Tax Reform, 5 October 2017, Brussels, Belgium. https://www.slideshare.net/IEEP_eu/case-study-the-danish-pesticide-tax?from_action=save.

He, X. (2018). Rapid development of unmanned aerial vehicles (UAV) for plant protection and application technology in China. *Outlooks on Pest Management* 29(4), 162-167. http://doi.org/10.1564/v29_aug_04.

Henson, S. and Humphrey, J. (2009*). The Impacts of Private Food Safety Standards on the Food Chain and on Public Standard-setting Processes*. Paper prepared for FAO/WHO. http://www.fao.org/3/a-i1132e.pdf.

High Level Panel of Experts on Food Security and Nutrition of the Committee on World Food Security (2019). *Agroecological and Other Innovative Approaches for Sustainable Agriculture and Food Systems that Enhance Food Security and Nutrition*. http://www.fao.org/3/ca5602en/ca5602en.pdf.

Hokkanen, H.M.T. (2015). Integrated pest management at the crossroads: Science, politics, or business (as usual)? *Arthropod-Plant Interactions* 9, 543-545. https://doi.org/10.1007/s11829-015-9403-y.

Horan, L. (2015). Catching the drift in Iowa, 30 April. *Pesticide Action Network North America*. http://www.panna.org/blog/catching-drift-iowa.

Horne, M. (2019). Knapsack sprayers: Operators, standards and selection. *Outlooks on Pest Management* 30(4), 171-175. https://doi.org/10.1564/v30_aug_07.

Hruska, A.J. (2019). Fall armyworm (*Spodoptera frugiperda*) management by smallholders. *CAB Reviews* 14. No. 043. https://doi.org/10.1079/PAVSNNR201914043.

Hunter III, J.E., Gannon, T.W., Richardson, R.J., Yelverton, F.H. and Leon, R.G. (2020). Integration of remote-weed mapping and an autonomous spraying unmanned aerial vehicle for site-specific weed management. *Pest Management Science* 76, 1386-1392. https://doi.org/10.1002/ps.5651.

IFOAM – Organics International (2014). *IFOAM Norms for Organic Production and Processing. Version 2014*. https://www.ifoam.bio/sites/default/files/2020-04/ifoam_norms_version_july_2014.pdf.

Institut national de la recherche agronomique (France) (2017). *International Conference on Global Crop Losses*. https://inra-dam-front-resources-cdn.brainsonic.com/ressources/afile/424160-b660f-resource-smach-crop-losses-conference-final-report.pdf.

Instituto de Desenvolvimento Econômico e Social de Fronteiras (Brazil) (2019). O contrabando de defensivos agrícolas no Brasil, 24 June. http://www.idesf.org.br/2019/06/24/o-contrabando-de-defensivos-agricolas-no-brasil/. Accessed 8 November 2020.

Integrated Support Programme for Arable Agriculture Development (2013). *Guidelines for Integrated Support Programme for Arable Agriculture Development (ISPAAD)*. Revised version 2013. Ministry of Agriculture, Republic of Botswana. https://www.gov.bw/sites/default/files/2019-12/ISPAAD%20Guidelines%20Revised%20May%202013_0.pdf.

International Biocontrol Manufacturers Association (2019). *Definition: Bioprotection as the Global Term for All Biocontrol Technologies*. https://www.ibma-global.org/upload/attach-image/ibma-definition-leaflet-web.pdf.

International Pesticide Application Research Consortium (2020). Dropdata website. http://www.dropdata.org/. Accessed 8 November 2020.

International Service for the Acquisition of Agri-biotech Applications (2019). *Global Status of Commercialized Biotech/GM Crops in 2018: Biotech Crops Continue to Help Meet the Challenges of Increased Population and Climate Change*. https://www.isaaa.org/resources/publications/briefs/54/download/isaaa-brief-54-2018.pdf.

International Service for the Acquisition of Agri-biotech Applications (2020). GM approval database. https://www.isaaa.org/gmapprovaldatabase/. Accessed 8 November 2020.

International Union for Conservation of Nature (2020). Invasive species. https://www.iucn.org/theme/species/our-work/invasive-species. Accessed 8 November 2020.

Iost Filho, F.H., Heldens, W.B., Kong, Z. and de Lange, E.S. (2020). Drones: Innovative technology for use in precision pest management. *Journal of Economic Entomology* 113(1), 1-25. https://doi.org/10.1093/jee/toz268.

Jactel, H., Koricheva, J. and Castagneyrol, B. (2019). Responses of forest insect pests to climate change: Not so simple. *Current Opinion in Insect Science* 35, 103-108. https://doi.org/10.1016/j.cois.2019.07.010. Accessed 8 November 2020.

Jensen, P.K. and Olesen, M.H. (2014). Spray mass balance in pesticide application: A review. *Crop Protection* 61, 23e31. http://dx.doi.org/10.1016/j.cropro.2014.03.006.

Jin, S., Bluemling, B. and Mol, A.P.J. (2015). Information, trust and pesticide overuse: Interactions between retailers and cotton farmers in China. *NJAS – Wageningen Journal of Life Sciences* 72-73. 23-32. http://dx.doi.org/10.1016/j.njas.2014.10.003.

Jørs, E., Lander, F., Huici, O., Morant, R.C., Gulis, G. and Konradsen, F. (2014). Do Bolivian small holder farmers improve and retain knowledge to reduce occupational pesticide poisonings after training on Integrated Pest Management? *Environmental Health* 13, 75. http://www.ehjournal.net/content/13/1/75.

Kah, M. (2015). Nanopesticides and nanofertilizers: Emerging contaminants or opportunities for risk mitigation? *Frontiers in Chemistry* 3, 64. https://doi.org/10.3389/fchem.2015.00064.

Kah, M., Kookana, R.S., Gogos, A. and Bucheli, T.D. (2018). A critical evaluation of nanopesticides and nanofertilizers against their conventional analogues. *Nature Nanotechnology* 13, 677-684. https://doi.org/10.1038/s41565-018-0131-1.

Kawarazuka, N., Damtew, E., Mayanja, S., Okonya, J.S., Rietveld, A., Slavchevska, V. and Teeken, B. (2020) A gender perspective on pest and disease management from the cases of roots, tubers and bananas in Asia and Sub-Saharan Africa. *Frontiers in Agronomy* 2: 7. https://doi.org/10.3389/fagro.2020.00007.

Klümper, W. and Qaim, M. (2014). A meta-analysis of the impacts of genetically modified crops. *PLoS ONE* 9(11), e111629. https://doi.org/10.1371/journal.pone.0111629.

Kniss, A.R. (2017). Long-term trends in the intensity and relative toxicity of herbicide use. *Nature Communications* 8, 14865. https://doi.org/10.1038/ncomms14865.

Knowles, A. (2008). Recent developments of safer formulations of agrochemicals. *Environment Systems and Decisions* (formerly *The Environmentalist*) 28, 35-44. https://doi.org/10.1007/s10669-007-9045-4.

Koch, S., Epp, A., Lohmann, M. and Böl, G.-F. (2017). Pesticide residues in food: Attitudes, beliefs, and misconceptions among conventional and organic consumers. *Journal of Food Protection* 80(12), 2083-2089. https://doi.org/10.4315/0362-028X.JFP-17-104.

Kookana, R.S., Boxall, A.B.A., Reeves, P.T., Ashauer. R., Beulke, S., Chaudhry, Q. *et al*. (2014). Nanopesticides: Guiding principles for regulatory evaluation of environmental risks. *Journal of Agricultural and Food Chemistry* 62(19), 4227-4240. https://doi.org/10.1021/jf500232f.

Krishna, V.V. and Qaim, M. (2008). Consumer attitudes toward GM food and pesticide residues in India. *Review of Agricultural Economics* 30 (2), 233- 251. https://doi.org/10.1111/j.1467-9353.2008.00402.x.

Lamberth, C., Jeanmart, S., Luksch, T. and Plant, A. (2013). Current challenges and trends in the discovery of agrochemicals. *Science* 341(6147), 742-746. https://doi.org/10.1126/science.1237227.

Lee, R., den Uyl, R. and Runhaar, H. (2019). Assessment of policy instruments for pesticide use reduction in Europe; Learning from a systematic literature review. *Crop Protection* 126, 104929. https://doi.org/10.1016/j.cropro.2019.104929.

Lehtola, M.M., Rautiainen, R.H., Day, L.M., Schonstein, E., Suutarinen, J., Salminen, S. and Verbeek, J.H. (2008). Effectiveness of interventions in preventing injuries in agriculture – a systematic review and meta-analysis. *Scandinavian Journal of Work, Environment and Health* 34(5), 327-336. https://doi.org/10.5271/sjweh.1279.

Lekei, E.E., Ngowi, A.V. and London, L. (2014). Pesticide retailers' knowledge and handling practices in selected towns of Tanzania. *Environmental Health* 13, 79. http://www.ehjournal.net/content/13/1/79.

LePrevost, C.E., Storm, J.F., Asuaje, C.R., Arellano, C. and Cope, W.G. (2014). Assessing the effectiveness of the *Pesticides and Farmworker Health Toolkit*: A curriculum for enhancing farmworkers' understanding of pesticide safety concepts. *Journal of Agromedicine* 19(2), 96-102. https://doi.org/10.1080/1059924X.2014.886538.

Levesque, D.L., Arif, A.A. and Shen, J. (2012). Effectiveness of pesticide safety training and knowledge about pesticide exposure among Hispanic farmworkers. *Journal of Occupational and Environmental Medicine* 54(12), 1550-1556. https://doi.org/10.1097/JOM.0b013e3182677d96.

LexisNexis (2020) Nexis Uni. https://www.lexisnexis.com/en-us/products/nexis-uni.page. Accessed 8 November 2020.

Li, L., Xu, Z., Kah, M., Lin, D. and Filser, J. (2019). Nanopesticides: A comprehensive assessment of environmental risk is needed before widespread agricultural application. *Environmental Science and Technology* 53, 7923-7924. https://doi.org/10.1021/acs.est.9b03146.

MacDonald, J.M. (2019). Mergers in seeds and agricultural chemicals: What happened? *Amber Waves,* 15 February. Economic Research Service, United States Department of Agriculture. https://www.ers.usda.gov/amber-waves/2019/february/mergers-in-seeds-and-agricultural-chemicals-what-happened/. Accessed 8 November 2020.

Mancini, F., Jiggins, J.L.S. and O'Malley, M. (2009). Reducing the incidence of acute pesticide poisoning by educating farmers on integrated pest management in South India. *International Journal of Occupational and Environmental Health* 15, 143-151. https://doi.org/10.1179/oeh.2009.15.2.143.

Matthews, G.A. (2018). *A History of Pesticides*. Wallingford, UK: CABI. https://www.cabi.org/bookshop/book/9781786394873/.

Matthews, G.A. (2020). Some views on current concerns about pesticides. *Outlooks on Pest Management* 31(5) http://www.researchinformation.co.uk/pest.php. Accessed 8 November 2020.

Matthews, G.A., Bateman, R. and Miller, P. (2014). *Pesticide Application Methods, 4th edition*. Oxford: Wiley-Blackwell. https://www.wiley.com/en-us/Pesticide+Application+Methods%2C+4th+Edition-p-9781118351307. Accessed 8 November 2020.

Maupin, J. and Norton, G. (2010). *Pesticide Use and IPM Adoption: Does IPM Reduce Pesticide Use in the United States*. https://core.ac.uk/download/pdf/6550526.pdf.

McIntyre, B.D., Herren, H.R., Wakhungu, J. and Watson, R.T. (eds.) (2009). Pest management. In *Agriculture at a Crossroads: Global Report*. International Assessment of Agricultural Knowledge, Science and Technology for Development. Washington, D.C.: Island Press. Chapter 2.3.2, pp. 98-106. https://www.unenvironment.org/resources/report/agriculture-crossroads-global-report.

Naranjo, S.E. and Ellsworth, P.C. (2009). Fifty years of the integrated control concept: Moving the model and implementation forward in Arizona. *Pest Management Science* 65, 1267-1286. https://doi.org/10.1002/ps.1861.

National Academies of Sciences, Engineering, and Medicine (2016). *Genetically Engineered Crops: Experiences and Prospects*. Washington, D.C.: The National Academies Press. https://doi.org/10.17226/23395.

Norton, G.W., Alwang, J., Kassie, M. and Muniappan, R. (2019). Economic impacts of integrated pest management practices in developing countries. In *The Economics of Integrated Pest Management of Insects*. Onstad, D.W. and Crain, P.R. (eds.). Wallingford, UK: CABI. Chapter 8, pp. 140-154. https://www.cabi.org/bookshop/book/9781786393678/.

Nuyttens, D., Devarrewaere, W., Verboven, P. and Foquéa, D. (2013). Pesticide-laden dust emission and drift from treated seeds during seed drilling: A review. *Pest Management Science* 69, 564-575. http://www.ask-force.org/web/Bees/Nuyttens-Pesticide-Laden-Dust-Emission-Seeds-3013.pdf.

Ochago, R. (2018) Gender and pest management: constraints to integrated pest management uptake among smallholder coffee farmers in Uganda. *Cogent Food & Agriculture* 4(1). https://doi.org/10.1080/23311932.2018.1540093.

Oerke, E.C. (2006). Crop losses to pests. *Journal of Agricultural Science* 144(1), 31-43. https://doi.org/10.1017/S0021859605005708.

Okonya, J.S. and Kroschel, J. (2015). A cross-sectional study of pesticide use and knowledge of smallholder potato farmers in Uganda. *BioMed Research International* 2015, 759049. http://dx.doi.org/10.1155/2015/759049.

Oleksa Vanzant, J. (2014-2015). A modern tale of the fox guarding the hen house: The inherent conflict of interest that exists when pesticide distributors employ pest control advisors. *San Joaquin Agricultural Law Review* 24, 247-275. https://www.sjcl.edu/images/stories/sjalr/volumes/V24N1C6.pdf.

Oliver, E. (2018). Growing importance of China and India in the global crop protection market. Agribusiness Intelligence, Informa.

Organisation for Economic Co-operation and Development (2017). The evolution of the tax on pesticides and the pesticide savings certificates in France. *In The Political Economy of Biodiversity Policy Reform*. Paris. Chapter 3, pp. 41-58. https://dx.doi.org/10.1787/9789264269545-7-en.

Organisation for Economic Co-operation and Development (2019). *Recommendation of the Council on Countering the Illegal Trade of Pesticides*. OECD/LEGAL/0446. https://legalinstruments.oecd.org/en/instruments/OECD-LEGAL-0446.

Organisation for Economic Co-operation and Development and Food and Agriculture Organization of the United Nations (2018). *OECD-FAO Agricultural Outlook 2018-2027*. Paris: OECD Publishing and Rome: FAO. http://www.fao.org/3/i9166en/I9166EN.PDF.

Oya, C., Schaefer, F. and Skalidou, D. (2018). The effectiveness of agricultural certification in developing countries: A systematic review. *World Development* 112, 282-312. https://doi.org/10.1016/j.worlddev.2018.08.001.

Paini, D.R., Sheppard, A.W., Cook, D.C., De Barro, P.J., Worner, S.P. and Thomas, M.B. (2016). Global threat to agriculture from invasive species. *Proceedings of the National Academy of Sciences of the United States of America* 113(27), 7575-7579. https://doi.org/10.1073/pnas.1602205113.

Pandya, I.Y. (2018). Pesticides and their applications in agriculture. *Asian Journal of Applied Science and Technology* 2(2), 894-900. https://www.researchgate.net/publication/325011090_Pesticides_and_Their_Applications_in_Agriculture.

Parsa, S., Morse, S., Bonifacio, A., Chancellor, T.C.B., Condori, B., Crespo-Pérez, V. *et al*. (2014). Obstacles to integrated pest management adoption in developing countries. *Proceedings of the National Academy of Sciences of the United States of America* 111(10), 3889-3894. www.pnas.org/cgi/doi/10.1073/pnas.1312693111.

Pasiani, J.O., Torres, P., Silva, J.R., Diniz, B.Z. and Caldas, E.D. (2012). Knowledge, attitudes, practices and biomonitoring of farmers and residents exposed to pesticides in Brazil. *International Journal of Environmental Research and Public Health* 9, 3051-3068. https://doi.org/10.3390/ijerph9093051.

Perry, E.D., Ciliberto, F., Hennessy, D.A. and Moschini, G. (2016). Genetically engineered crops and pesticide use in U.S. maize and soybeans. *Science Advances* 2, e1600850. https://doi.org/10.1126/sciadv.1600850.

Peshin, R. and Zhang, W.J. (2014). Integrated pest management and pesticide use. In *Integrated Pest Management*. Pimentel, D. and Peshin, R. (eds.). Dordrecht: Springer Science and Business Media. Chapter 1, pp. 1-46. https://doi.org/10.1007/978-94-007-7796-5_1.

Pest Control Products Board (Kenya) (2020). PCPB fees. http://www.pcpb.go.ke/pcpb-fees/. Accessed 8 November 2020.

Phillips McDougall (2017). *The Global Agrochemical Market – Trends by Crop*. Presentation at the 11th China International Forum on Development of Pesticide Industry, Shanghai, 28 February 2017. Phillips McDougall Agribusiness intelligence. http://www.cac-conference.com/Uploads/Editor/2017-03-07/58be2c387de29.pdf.

Phillips McDougall (2018a). *Evolution of the Crop Protection Industry since 1960*. https://croplife.org/wp-content/uploads/2018/11/Phillips-McDougall-Evolution-of-the-Crop-Protection-Industry-since-1960-FINAL.pdf.

Phillips McDougall (2018b). The global crop protection industry in 2017, 6 July. https://agribusiness.ihsmarkit.com/sectors/crop-science/index.html. Accessed 11 November 2020.

Phillips McDougall (2019a). Key performance trends in the global crop protection industry in 2018. Inforgraphic, May 2019. https://agribusiness.ihsmarkit.com/sectors/crop-science/index.html. Accessed 11 November 2020.

Phillips McDougall (2019b). *Evolution of the Leading Agrochemical Companies: A Graphical Representation of the Key Agrochemical Mergers and Acquisitions*. 4th edition. https://cdn.nufarm.com/wp-content/uploads/sites/31/2019/03/22175501/PP_Industry.pdf.

Potts, J., Voora, V., Lynch, M. and Mamadova, A. (2017). *State of Sustainability Initiatives – Standards and Biodiversity*. Winnipeg: International Institute for Sustainable Development. https://www.iisd.org/system/files/publications/standards-biodiversity-ssi-report.pdf.

Pozebon, H., Marques, R.P., Padilha, G., O´Neal, M., Valmorbida, I., Bevilaqua, J.G. *et al*. (2020). Arthropod invasions versus soybean production in Brazil: A review. *Journal of Economic Entomology* 113(4), 1591-1608. https://doi.org/10.1093/jee/toaa108.

Pretty, J. (2018). Intensification for redesigned and sustainable agricultural systems. *Science* 362(6417), eaav0294. https://doi.org/10.1126/science.aav0294.

Pretty, J. and Bharucha, Z.P. (2015). Integrated pest management for sustainable intensification of agriculture in Asia and Africa. *Insects* 6, 152-182. https://doi.org/10.3390/insects6010152.

Rainforest Alliance (2020). 2020 Sustainable Agriculture Standard – i. Applicable for medium large farms, and ii. Applicable to smallholder farms. Draft Standards v2.0. https://www.rainforest-alliance.org/business/innovating-certification/2020-sustainable-agriculture-standard/. Accessed 11 November 2020.

Rejesus, R. (2019). What can we learn from more recent (and more 'rigorous') economic impact assessments of integrated pest management farmer field schools (IPM-FFS)? In *The Economics of Integrated Pest Management of insects*. Onstad, D.W. and Crain, P.R. (eds.). Wallingford, UK: CABI. Chapter 3, pp. 35-48. https://www.cabi.org/bookshop/book/9781786393678/.

Remoundou. K., Brennan, M., Hart, A. and Frewer, L.J. (2014). Pesticide risk perceptions, knowledge, and attitudes of operators, workers, and residents: A review of the literature. *Human and Ecological Risk Assessment: An International Journal* 20(4), 1113-1138. https://doi.org/10.1080/10807039.2013.799405.

Ricci, F., Chiesi, A., Bisio, C., Panari, C. and Pelosi, A. (2016). Effectiveness of occupational health and safety training – A systematic review with meta-analysis. *Journal of Workplace Learning* 28(6), 355-377. https://doi.org/10.1108/JWL-11-2015-0087.

Riwthong, S., Schreinemachers, P., Grovermann, C. and Berger, T. (2015). Land use intensification, commercialization and changes in pest management of smallholder upland agriculture in Thailand. *Environmental Science and Policy* 45, 11-19. http://dx.doi.org/10.1016/j.envsci.2014.09.003.

Robin, D.C. and Marchand, P.A. (2018). Evolution of the biocontrol active substances in the framework of the European Pesticide Regulation (EC) No. 1107/2009. *Pest Management Science* 75(4), 950-958. https://doi.org/10.1002/ps.5199.

Rotterdam Convention (2019). *Analysis of Definitions of the Term "Pesticides" sed by Parties to the Rotterdam Convention*. UNEP/FAO/RC/COP.9/INF/7. Conference of the Parties to the Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade. Ninth meeting, Geneva, 29 April-10 May 2019. http://www.pic.int/TheConvention/ConferenceoftheParties/Meetings/COP9/Overview/tabid/7528/language/en-US/Default.aspx.

Route to Food (2019). *Pesticides in Kenya: Why Our health, Environment and Food Security Are at Stake*. https://routetofood.org/wp-content/uploads/2019/08/Pesticide-use-in-Kenya-COMP.pdf.

Sam, K.G., Andrade, H.H., Pradhan, L., Pradhan, A., Sones, S.J., Rao, P.G.M. and Sudhakar, C. (2008). Effectiveness of an educational program to promote pesticide safety among pesticide handlers of South India. *International Archives of Occupational and Environmental Health* 81, 787-795. https://doi.org/10.1007/s00420-007-0263-3.

Savary, S., Ficke, A., Aubertot, J.-N., and Hollier, C. (2012). Crop losses due to diseases and their implications for global food production losses and food security. *Food Security* 4(4), 519-537. https://doi.org/10.1007/s12571-012-0200-5.

Savary, S., Teng, P.S., Willocquet, L. and Nutter, F.W. (2006). Quantification and modeling of crop losses: A review of purposes. *Annual Review of Phytopathology* 44, 89-112. https://doi.org/10.1146/annurev.phyto.44.070505.143342.

Savary, S., Willocquet, L., Pethybridge, S.J., Esker, P., McRoberts, N. and Nelson, A. (2019). The global burden of pathogens and pests on major food crops. *Nature Ecology and Evolution* 3, 430-439. https://doi.org/10.1038/s41559-018-0793-y.

Schlatter. B., Trávníček, J., Lernoud, J. and Willer, H. (2020). Current statistics on organic agriculture worldwide: Area, operators and market. In *The World of Organic Agriculture. Statistics and Emerging Trends 2020*. Willer, H., Schlatter, B., Trávníček, J., Kemper, L. and Lernoud, J. (eds.). Frick, Switzerland: Research Institute of Organic Agriculture (FiBL) and Bonn: Organics International. 31-131. https://www.organic-world.net/yearbook/yearbook-2020.html.

Schreinemachers, P. and Tipraqsa, P. (2012). Agricultural pesticides and land use intensification in high, middle and low income countries. *Food Policy* 37(6), 616-626. https://doi.org/10.1016/j.foodpol.2012.06.003.

Schreinemachers, P., Chen, H.P., Nguyen, T.T.L., Buntong, B., Bouapaoe, L., Gautam, S. *et al*. (2017). Too much to handle? Pesticide dependence of smallholder vegetable farmers in Southeast Asia. *Science of The Total Environment* 593-594, 470-477. http://dx.doi.org/10.1016/j.scitotenv.2017.03.181.

Schreinemachers, P., Grovermann, C., Praneetvatakul, S., Heng, P., Nguyen, T.T.L., Buntong, B. *et al*. (2020). How much is too much? Quantifying pesticide overuse in vegetable production in Southeast Asia. *Journal of Cleaner Production* 244, 118738. https://doi.org/10.1016/j.jclepro.2019.118738.

Schreinemachers, P., Wua, M., Uddin, M.N., Ahmad, S. and Hanson, P. (2016). Farmer training in off-season vegetables: Effects on income and pesticide use in Bangladesh. *Food Policy* 61, 132-140. http://dx.doi.org/10.1016/j.foodpol.2016.03.002.

Schulz, R., Bub, S., Petschick, L.L., Stehle, S. and Wolfram, J. (2021). Applied pesticide toxicity shifts toward plants and invertebrates, even in GM crops. *Science* 372 (6537), 81-84. https://doi.org/10.1126/science.abe1148.

Servicio de Administración Tributaria (Mexico D.F.) (2020). Impuesto a log plaguicidas. http://omawww.sat.gob.mx/fichas_tematicas/reforma_fiscal/Paginas/plaguicidas_2014.aspx. Accessed 11 November 2020.

Settle. W., Soumare, M., Sarr, M., Garba, M.H. and Poisot, A.-S. (2014). Reducing pesticide risks to farming communities: Cotton farmer field schools in Mali. *Philosophical Transactions of the Royal Society B* 369, 20120277. http://dx.doi.org/10.1098/rstb.2012.0277.

Shin, M., Werner, A.K., Strosnider, H., Hines, L.B., Balluz, L. and Yip, F.Y. (2019). Public perceptions of environmental public health risks in the United States. *International Journal of Environmental Research and Public Health* 16, 1045. https://doi.org/10.3390/ijerph16061045.

Smith, W.K., Nelson, E., Johnson, J.A., Polasky, S., Milder, J.C., Gerber, J.S. *et al*. (2019) Voluntary sustainability standards could significantly reduce detrimental impacts of global agriculture. *Proceedings of the National Academy of Sciences of the United States of America* 116(6), 2130-2137. https://doi.org/10.1073/pnas.1707812116.

State of Hawaii Department of Agriculture (United States) (2019). Coffee Berry Borer (CBB) Pesticide Subsidy Program. http://hdoa.hawaii.gov/pi/main/cbbsubsidy/. Accessed 8 November 2020.

Sun, S., Hu, R., Zhang, C. and Shi, G. (2019). Do farmers misuse pesticides in crop production in China? Evidence from a farm household survey. *Pest Management Science* 75, 2133-2141. https://doi.org/10.1002/ps.5332.

Tayleur, C., Balmford, A., Buchanan, G.M., Butchart, S.H.M., Ducharme, H., Green *et al*. (2017). Global coverage of agricultural sustainability standards, and their role in conserving biodiversity. *Conservation Letters* 10(5), 610-618. https://doi.org/10.1111/conl.12314.

Thorburn, C. (2015). The rise and demise of integrated pest management in rice in Indonesia. *Insects* 6, 381-408. https://doi.org/10.3390/insects6020381.

Tona, E., Calcante, A. and Oberti, R. (2018). The profitability of precision spraying on specialty crops: A technical–economic analysis of protection equipment at increasing technological levels. *Precision Agriculture* 19, 606-629. https://doi.org/10.1007/s11119-017-9543-4.

Tribun Jateng (2020). Breaking News: Brebes regional police raided Rosok warehouse, place to make fake pesticides: Two persons arrested, 2 January. *Article in Indonesian*. https://jateng.tribunnews.com/2020/01/02/breaking-news-polres-brebes-gerebek-gudang-rosok-tempat-bikin-pestisida-palsu-dua-orang-ditangkap. Accessed 11 November 2020.

United Nations (2017). *Report of the Special Rapporteur on the Right to Food*. Human Rights Council, 34th Session, 27 February-24 March 2017. United Nations General Assembly, New York. https://documents-dds-ny.un.org/doc/UNDOC/GEN/G17/017/85/PDF/G1701785.pdf.

United Nations Environment Programme (2017). *Report of the Effectiveness Evaluation of the Stockholm Convention on Persistent Organic Pollutants*. UNEP/POPS/Conf.8/INF/40. 8th Meeting of the Conference of the Parties of the Stockholm Convention on Persistent Organic Pollutants, Geneva, 24 April-5 May 2017. http://www.pops.int/TheConvention/ConferenceoftheParties/Meetings/COP8/tabid/5309/Default.aspx.

United Nations Environment Programme (2019). *Report of the DDT Expert Group on the Assessment of the Production and Use of DDT and its Alternatives for Disease Vector Control*. UNEP/POPS/COP.9/INF/6. 9th Meeting of the Conference of the Parties to the Stockholm Convention on Persistent Organic Pollutants, Geneva, 29 April-10 May 2019. http://www.pops.int/TheConvention/ConferenceoftheParties/Meetings/COP9/tabid/7521/Default.aspx.

United Nations Environment Programme (2020a). *An Assessment Report on Issues of Concern: Chemicals and Waste Issues Posing Risks to Human Health and the Environment*. https://wedocs.unep.org/bitstream/handle/20.500.11822/33807/ARIC.pdf?sequence=1&isAllowed=y.

United Nations Environment Programme (2020b). *Study on the Effects of Taxes and Subsidies on Pesticides and Fertilizers*. Background document to UNEA-5 Review Report on the Environmental and Health Effects of Pesticides and Fertilizers. https://greenfiscalpolicy.org/reports/unep-study-on-the-effects-of-taxes-and-subsidies-on-pesticides-and-fertilizers/.

United Nations Environment Programme (2022). *Synthesis Report on Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them*. https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

United Nations Environment Programme and GRID-Arendal (2020). *The Illegal Trade in Chemicals*. https://wedocs.unep.org/bitstream/handle/20.500.11822/32021/Chemicals.pdf?sequence=1&isAllowed=y.

United Nations Forum on Sustainability Standards (2020). About UNFSS. https://unfss.org/home/about-unfss/. Accessed 28 April 2021.

United Nations Interregional Crime and Justice Research Institute (2016). *Illicit Pesticides, Organized Crime and Supply Chain Integrity*. http://www.unicri.it/sites/default/files/2019-11/Illicit%20pesticides%2C%20organized%20crime%20and%20supply%20chain%20integrity.pdf. Accessed 11 November 2020.

United States Department of Agriculture (2018). *A National Roadmap for Integrated Pest Management. Revised September 21, 2018*. https://www.ars.usda.gov/ARSUserFiles/OPMP/IPM%20Road%20Map%20FINAL.pdf.

Vaidya, A., Gyenwali, D., Tiwari, S., Pande, B.R. and Jørs, E. (2017). Changes in perceptions and practices of farmers and pesticide retailers on safer pesticide use and alternatives: Impacts of a community intervention in Chitwan, Nepal. *Environmental Health Insights* 11, 1-12. https://doi.org/10.1177/1178630217719270.

van den Berg, H., Zaim, M., Yadav, R.S., Soares, A., Ameneshewa, B., Mnzava A. *et al*. (2012). Global trends in the use of insecticides to control vector-borne diseases. *Environmental Health Perspectives* 120(4), 577-582. https://doi.org/10.1289/ehp.1104340.

van den Bosch, R. (1989). *The Pesticide Conspiracy*. University of California Press. https://www.ucpress.edu/book/9780520068230/the-pesticide-conspiracy.

van Lenteren, J.C., Bale, J., Bigler, F., Hokkanen, H.M.T. and Loomans, A.J.M. (2006). Assessing risks of releasing exotic biological control agents of arthropod pests. *Annual Review of Entomology* 51, 609-634. https://doi.org/10.1146/annurev.ento.51.110104.151129.

van Lenteren, J.C., Bolckmans, K., Kohl, J., Ravensberg, W.J. and Urbaneja, A. (2018). Biological control using invertebrates and microorganisms: Plenty of new opportunities. *BioControl* 63(1), 39-59. https://doi.org/10.1007/s10526-017-9801-4.

Vayssière, P. and Mimeur, J. (1926). *Les Insectes Nuisibles au Cotonnier en Afrique Occidentale Française*. Bibliothèque de l'Institut National d'Agronomie Coloniale. Paris: Librairie Émile Larose. https://bibliotheques.mnhn.fr/medias/detailstatic.aspx?INSTANCE=EXPLOITATION&RSC_BASE=HORIZON&RSC_DOCID=227052.

Waddington, H., Snilstveit, B., Hombrados, J., Vojtkova, M., Phillips, D., Davies, P. and White, H. (2014). Farmer Field Schools for improving farming practices and farmer outcomes: A systematic review. *Campbell Systematic Reviews* 10(1), i-335. https://doi.org/10.4073/csr.2014.6.

Wan, M., Gu, R., Zhang, T., Zhang, Y., Ji, H., Wang, B. *et al*. (2019). Conflicts of interests when connecting agricultural advisory services with agri-input businesses. *Agriculture* 9(10), 218. https://www.mdpi.com/2077-0472/9/10/218.

Waterfield, G. and Zilberman, D. (2012). Pest management in food systems: An economic perspective. *Annual Review of Environment and Resources* 37, 223-45. https://doi.org/10.1146/annurev-environ-040911-105628.

Weddle, P.W., Welter, S.C. and Thomson, D. (2009). History of IPM in California pears – 50 years of pesticide use and the transition to biologically intensive IPM. *Pest Management Science* 65(12), 1287-1292. https://doi.org/10.1002/ps.1865.

Wei, X., Zhao, L., Qiao, Y., Wang, B., Wan, M. and Toepfer, S. (2019). *Implementing Agri-policies on Pesticide Reduction through Subsidies and Plant Clinics in China*. CABI Working Paper 13. Wallingford, UK: CABI https://dx.doi.org/10.1079/CABICOMM-62-8118.

Wilke A.B.B., Beier, J.C. and Benelli, G. (2019). Complexity of the relationship between global warming and urbanization – an obscure future for predicting increases in vector-borne infectious diseases. *Current Opinion in Insect Science* 35, 1-9. https://doi.org/10.1016/j.cois.2019.06.002.

Wood, A. (2019). Compendium of pesticide common names: Classified lists of pesticides. http://www.alanwood.net/pesticides/class_pesticides.html. Accessed 11 November 2020.

World Bank (2018). *Environmental and Social Framework Standards for IPF Operations*. *ESS 3: Resource Efficiency and Pollution Prevention and Management*. *Guidance Note for Borrowers*. http://documents1.worldbank.org/curated/en/112401530216856982/ESF-Guidance-Note-3-Resource-Efficiency-and-Pollution-Prevention-and-Management-English.pdf.

World Bank (2020a). The world by income and region: World Bank historical classification by income. Consulted 30 June 2019. http://datatopics.worldbank.org/world-development-indicators/the-world-by-income-and-region.html. Accessed 11 November 2020.

World Bank (2020b). GDP per capita, PPP (current international $). Extracted from World Development Indicators, World Bank Data Bank. Consulted 28 June 2019. https://data.worldbank.org/indicator/NY.GDP.PCAP.PP.CD?view=chart. Accessed 11 November 2020.

World Health Organization (2011). *Global Insecticide Use for Vector-Borne Disease Control – A 10-Year Assessment (2000-2009)*. *Fifth Edition*. Geneva. https://apps.who.int/iris/handle/10665/44670.

World Health Organization (2017). *Global Vector Control Response 2017-2030*. https://apps.who.int/iris/bitstream/handle/10665/259205/9789241512978-eng.pdf?sequence=1.

Wyckhuys, K.A.G., Heong, K.L., Sanchez-Bayo, F., Bianchi, F.J.J.A., Lundgren, J.G. and Bentley, J.W. (2019). Ecological illiteracy can deepen farmers' pesticide dependency. *Environmental Research Letters* 14, 093004. https://doi.org/10.1088/1748-9326/ab34c9.

Yuan, G. (2019). Global top 20 agrochemical companies achieve business growth in 2018, 10 Chinese companies stepping into top 20, 1 November. *AgroPages*. http://news.agropages.com/News/NewsDetail---32698.htm. Accessed 11 November 2020.

Yuantari, M.G.C., Van Gestel, C.A.M., Van Straalen, N.M., Widianarko, B., Sunoko, H.R. and Shobib, M.N. (2015). Knowledge, attitude, and practice of Indonesian farmers regarding the use of personal protective equipment against pesticide exposure. *Environmental Monitoring and Assessment* 187, 142. https://doi.org/10.1007/s10661-015-4371-3.

Zalucki, M.P., Adamson, D. and Furlong, M.J. (2009). The future of IPM: Whither or wither? *Australian Journal of Entomology* 48, 85-96. https://doi.org/10.1111/j.1440-6055.2009.00690.x.

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them

Envisioning A Chemical-Safe World

Chapter 3 of 12

The regulatory and policy environment for pesticide management

Contents

1 Global drivers, actors and policies affecting pesticides and fertilizer use

2 Status and trends of pesticide use

About iv

3 The regulatory and policy environment for pesticide management 1

3.1 Overview 1

3.2 International instruments that address pesticide distribution and use 3

3.2.1 Introduction 3
3.2.2 Legally binding international instruments that directly address pesticides 3
3.2.3 Voluntary international instruments and mechanisms that directly address pesticides 18
3.2.4 Legally binding international instruments indirectly addressing pesticides 31
3.2.5 Voluntary international instruments and mechanisms indirectly addressing pesticides 35
3.2.6 Discussion – International instruments and mechanisms 35

3.3 Regional collaboration on pesticide management 37

3.3.1 Introduction 37
3.3.2 Legally binding instruments 40
3.3.3 Voluntary regional collaboration mechanisms 40
3.3.4 Discussion: Regional agreements and mechanisms 40

3.4 National policies and legislation on pesticides 42

3.4.1 Pesticide legislation 42
3.4.2 Pesticide registration 46
3.4.3 Compliance monitoring and enforcement 47
3.4.4 Other legislation relevant to pesticides 49
3.4.5 Pest and pesticide management policy 49
3.4.6 Discussion – National legislation and policies 51

3.5 Corporate stewardship 52

3.5.1 Introduction 52
3.5.2 The pesticide industry 53

3.5.3 Food and commodities industries 54
3.5.4 Pesticide users 55
3.5.5 Discussion: Corporate stewardship 55

References 56

4 Environmental and health effects of pesticide use

5. The environmental, human health and economic impacts of pesticides

6 Current pesticide risk reduction and risk management

7 Status and trends of fertilizer use

8 The regulatory and policy environment of fertilizer management and use

9 Environmental and health effects of fertilizer use

10 The impact of fertilizer use

11 Current fertilizer risk reduction and risk management

12 Transformative actions to minimize the adverse impacts of pesticide and fertilizers

List of Boxes

Box 3.4-1 Elements of pesticide legislation. 42

List of Figures

Figure 3.2-1 Key international instruments and mechanisms which address pesticide management and use, either directly or indirectly. Conventions are listed only if they have been ratified by more than 50 countries. 4

Figure 3.2-2 The Rotterdam Convention has greatly increased its capacity-building initiatives. Shown are the annual number of workshops organized by or with the Rotterdam Convention on pesticides and industrial chemicals combined between 2010 and 2018. 7

Figure 3.2-3 The large majority of responses received by the Rotterdam Convention regarding importation of the pesticides listed in Annex III were "no consent". Situation on 23 April 2019; total number of import responses was 4,404. 7

Figure 3.2-4 Changes in the quantities of POPs imported for use from before 2004 to 2014. 11

Figure 3.2-5 74 per cent of Parties used POPs pesticides in 2001, compared with 15 per cent in 2017, but the number of Parties using industrial POPs scarcely declined during the same period. 11

Figure 3.2-6 Estimated global consumption of methyl bromide 1999-2018 for quarantine and pre-shipment (QPS) uses and non-QPS uses. 14

Figure 3.2-7 Pesticide-related notifications under the Agreement on the Application of Sanitary and Phytosanitary Measures (SPS Agreement) as a share of all SPS notifications, January 2010-July 2020. 16

Figure 3.3-1 Cooperation on pesticide management and regulation can take many forms, with increasing intensity and complexity 37

Figure 3.3-2 Examples of legally binding regional legislation and agreements for the management of pesticides 39

Figure 3.3-3 Examples of regional mechanisms for the management of pesticides 41

Figure 3.4-1 Elements of the pesticide life cycle and aspects in each step that are often regulated 44

Figure 3.4-2 Percentage of countries reporting that they have dedicated legislation for (primarily) agricultural pesticides, or for public health pesticides, based on surveys conducted during the last 30 years. The apparent reduction in 2010 is likely due to a different target being addressed by the survey: ministries responsible for public health vs. ministries responsible for agriculture. 45

Figure 3.4-3 There is little difference in the percentage of countries that have pesticide legislation in place in low, lower-middle and upper-middle income groups. 45

Figure 3.4-4 Staff capacity for pesticide registration increases with countries' income levels. GDP is per capita purchasing power parity (PPP); 2017 international dollars. 47

Figure 3.4-5 The percentage of countries in which substandard pesticide products are a concern is especially high in the African and Eastern Mediterranean regions. Laboratory capacity for quality control is low in the African and the Western Pacific regions. 48

Figure 3.5-1 Key elements of crop protection stewardship as promoted by CropLife International ("CropLife"). Individual companies focus on the first part of the stewardship life cycle, while industry associations and CropLife International are mainly concerned with the second part. 53

Figure 3.5-2 Countries with container management programmes (including pilots) supported by CropLife International and its members as of early 2020. 54

List of Tables

Table 3.2-1	Pesticides listed in Annex III of the Rotterdam Convention.	5
Table 3.2-2	Pesticides listed under the Stockholm Convention.	9
Table 3.2-3	History of the Code of Conduct.	19
Table 3.2-4	Outline of the Code of Conduct.	19
Table 3.2-5	Surveys conducted by FAO and WHO on implementation of the Code of Conduct.	20
Table 3.2-6	Trends in the implementation of certain key provisions of the Code of Conduct (percentage of responding countries agreeing with these statements)	21
Table 3.2-7	Highly Hazardous Pesticides (HHPs) are defined by FAO and WHO as having one or more of the following characteristics	23
Table 3.2-8	History of relevant international policy for the identification and risk reduction of Highly Hazardous Pesticides (HHPs).	24
Table 3.2-9	Examples of recent restrictions and bans in low and middle income countries of pesticides considered to pose unacceptable risk to the environment and/or human health.	25
Table 3.2-10	Aichi Biodiversity Targets that directly address the management and use of pesticides and selected recommended specific indicators	32
Table 3.2-11	Progress towards Aichi Biodiversity Indicators 3, 7 and 8 made by 2020 (with particular reference to pesticide risk reduction), based on information provided in the Global Biodiversity Outlook 5 (GBO-5) and the reviews conducted in this report.	33
Table 3.4-1	Percentage of countries reporting that they regulate specific elements of the pesticide life cycle.	46
Table 3.4-2	Responsible government authority which issues registrations for several types of pesticides. Shown is the percentage of countries answering each question positively, and the number of responding countries (n). Some countries reported there was more than one responsible authority.	46
Table 3.4-3	Some factors that could lead to non-compliance with pesticide legislation.	48
Table 3.4-4	Examples of national pesticide management policies and strategies, and achievements insofar as these have been established.	50

About

In December 2017, Resolution 4 of the 3rd Session of the United Nations Environment Assembly (UNEA 3) requested "the Executive Director to present a report on the environmental and health impacts of pesticides and fertilizers and ways of minimizing them, given the lack of data in that regard, in collaboration with the World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO) and other relevant organizations by the fifth session of the United Nations Environment Assembly". In response to this request, UNEP published a *Synthesis Report on the Environmental and Health Impacts of Pesticides and Fertilizers and Ways to Minimize Them*[1] in February 2022 (United Nations Environment Programme [UNEP] 2022).

The overall goal of the synthesis report is to provide the information base to enable other advocacy actions to be taken by stakeholders to minimize the adverse impacts of pesticides and fertilizers. Specific objectives of the synthesis report are to:

- Update understanding of current pesticide and fertilizer use practices;
- Present major environmental and health effects of pesticides and fertilizers, during their life cycle, and identify key knowledge gaps;
- Review current management practices, legislation and policies aimed at reducing risks in the context of the global chemicals, environmental and health agenda;
- Identify opportunities to minimize environmental and health impacts, including proven and innovative approaches.

This chapter on "The regulatory and policy environment for pesticide management" is the 3rd in a series of 12 chapters that make up a comprehensive compilation of scientific information. The chapters were developed to both inform and further elaborate on the information provided in the synthesis report. Please note that the disclaimers and copyright from the synthesis report apply

1 The Synthesis report is available at https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

The regulatory and policy environment for pesticide management

3

3.1 Overview

A broad and varied legislative and policy landscape exists for pesticide management. It addresses the manufacturing, trade and use of pesticides at international, regional and national levels. A range of instruments and mechanisms can exert a direct influence on pesticide management, for example through provisions that address individual pesticides, the trade in pesticides or pesticide-treated commodities, and pesticide legislation; or they may affect pesticide distribution and use indirectly by shaping broader environmental, health or economic legislation and policies. The extent to which such instruments contribute to pesticide risk reduction is assessed in this report, although not exhaustively.

Most international instruments, both legally binding and voluntary, have substantial provisions and programmes aimed at promoting information exchange, conducting training and awareness building activities, providing guidance on best practices, and strengthening technical and administrative skills. However, improving knowledge, awareness and attitudes with regard to sound pest and pesticide management does not necessarily reduce pesticide risks in a given country, region or situation. [Chapter 3.2]

Four legally binding instruments (the Montreal Protocol and the Rotterdam, Stockholm and Minamata Conventions) address individual pesticides. When those pesticides were listed under the respective conventions, however, the use of a considerable number of them had already been greatly reduced or had discontinued. At the same time, prospects of new listings of pesticides of concern have been shown to face constraints due to both technical and political limitations. The contribution of these instruments to reducing the environmental and human health risks posed by pesticides has therefore been limited. Exceptions are methyl bromide (under the Montreal Protocol) and endosulfan (under the Rotterdam and Stockholm Conventions), where significant reductions in use have been noted after they were listed. [Chapter 3.2.2]

There are indications that participation in multilateral environmental agreements (MEAs) results in the strengthening of national legislation. This improvement has occurred through strengthening of legislation and pesticide registration, harmonization of standards such as maximum residue limits (MRLs), and better evaluation and hazard classification of pesticides. An international mechanism, the Strategic Approach to International Chemicals Management (SAICM), has set the stage for addressing issues of concern such as the risks posed by Highly Hazardous Pesticides, globally. [Chapter 3.2]

A challenge in the case of many legally binding and voluntary international instruments is that their impact on pesticide risk reduction cannot be well evaluated. Appropriate indicators and procedures to allow such evaluation are not adequate, partly because the key objectives of these instruments may focus on addressing some specific aspects of the broader pesticide or chemical risk reduction. An exception is the Global Monitoring Plan of the Stockholm Convention on Persistent Organic Pollutants (POPs) and the implementation of the International Code of Conduct of Conduct on Pesticide Management [Chapter 3.2.6].

Regional cooperation is potentially a powerful approach for activities carried out over larger geographical regions, such as efficacy and residue testing and the evaluation, classification and authorization of pesticides, especially if resources are limited. Regional collaboration has also proven effective in high income regions. Such collaboration tends to be easier to implement in regions with similar administrative cultures, economic and agronomic conditions, and constraints. [Chapter 3.3].

Most countries in the world have adopted dedicated pesticide legislation providing a national legal basis for managing the registration, trade and use of pesticides. In many cases, however, not all aspects of the pesticide life cycle are effectively covered by national legislation, while procedures may be insufficient for effective pesticide management. Furthermore, legislation on domestic and public health pesticides is still inadequate in many countries (FAO 1996; WHO 2004; FAO 2010; Matthews *et al*. 2011; FAO and WHO 2019a). [Chapter 3.4.1].

The majority of countries have a pesticide registration system in place to evaluate and authorize the use of agricultural pesticides. Nevertheless, registration of public health and domestic use pesticides is inadequate in many countries. Especially in low and lower-middle income countries, human resources for the sound evaluation of pesticides are very limited while assessing efficacy, hazards and risks is becoming increasingly complex. [Chapter 3.4.2].

Government inspection and control of pesticide-related activities are essential for effective implementation of national pesticide legislation. However, capacity for enforcement is generally considered weak in many countries, often due to limited resources and fragmented regulatory entities. [Chapter 3.4.3].

Globally, the adoption of comprehensive pesticide management policies or strategies has been rare so far. Targets for the reduction of pesticide risks and/or use have mainly been established in Europe. Other countries have incorporated pest management principles in their legislation, regulations and associated policies. In countries that lack a comprehensive national pest/vector and pesticide management policy, adopted by the main stakeholders, the elaboration of such a policy would represent an important first step towards setting concrete targets for pesticide risk reduction. [Chapter 3.4.5].

Identifying ways to minimize the human health and environmental impacts of pesticides is part of the mandate of the report. It should be emphasized that pesticide risk reduction may not be a formal or explicit objective of some of the instruments reviewed. The evaluation is therefore not an assessment of the effectiveness of those instruments against their specific objectives, for which other mechanisms may be in place.

To assess whether an international, regional or national instrument contributes to a reduction of pesticide risks or impacts, outcome indicators and output indicators are used. Outcome indicators can be quantitatively related to a likely reduction in pesticide risks. Output indicators may lead to a reduction in pesticide risks if an instrument is implemented effectively.

Outcome indicators:

- Reduction in the authorization or use of pesticides (particularly hazardous ones) and/or banning, for example through cancelling imports or registration of such pesticides and/or replacing them with less hazardous alternatives such as biopesticides or low-risk pesticides, or through using approaches such as integrated pest or vector management (IPM/IVM) and agroecology;

- continued use of pesticides (particularly hazardous ones), but in such a manner that they pose lower risks to human health and the environment, e.g., through use restrictions, user training, less hazardous formulations, lower application rates and frequencies, and more/better use of personal protective equipment (PPE).

Output indicators:

- generation of tools, such as best practice guidelines and training materials for risk assessment, that may lead to reduced pesticide risks;

- implementation of activities, such as technical support for pesticide regulators and training in IPM or IVM, that may lead to reduced pesticide risks.

3.2 International instruments that address pesticide distribution and use

3.2.1 Introduction

A considerable number of international instruments and mechanisms address one or more aspects of the life cycle of pesticides. Some are legally binding, while others are voluntary. Although international instruments may directly address pesticide management, production, distribution and use, they can also provide more general policy context that indirectly addresses how pesticides are managed and used (Figure 3.2-1). The main provisions of these international instruments, insofar as they concern pesticides, are reviewed below.

3.2.2 Legally binding international instruments that directly address pesticides

The Rotterdam Convention

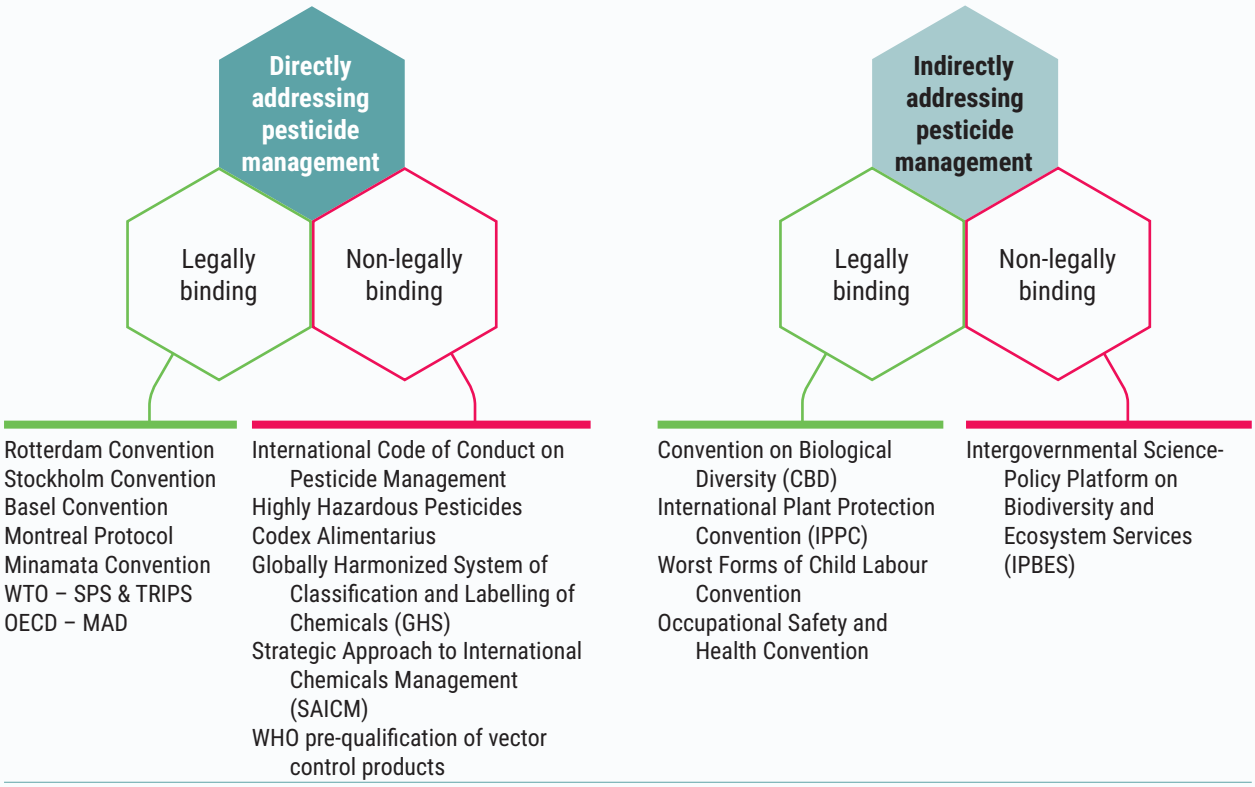
The Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade was adopted in 1998 and entered into force in 2004. As of late 2020, there were 164 Parties to the Convention (Rotterdam Convention 2020a).

Objectives and provisions

The objective of the Rotterdam Convention is to "promote shared responsibility and cooperative efforts among Parties in the international trade of certain hazardous chemicals in order to protect human health and the environment from potential harm and to contribute to their environmentally sound use, by facilitating information exchange about their characteristics, by providing for a national decision-making process on their import and export and by disseminating these decisions to Parties" (Rotterdam Convention 2019).

The Convention provides for listing of pesticides and industrial chemicals in its Annex III. Listing in Annex III does not in itself constitute a ban. Instead, it is based on national bans or severe restrictions for health and/or environmental reasons, notified to the Secretariat, in at least two countries in different prior informed consent (PIC) regions – or, in the case of Severely Hazardous Pesticide Formulations (SHPFs) (Rotterdam Convention n.d. a), nomination by one country. Following the listing of chemicals in Annex III, Parties communicate whether they consent to the importation of these chemicals.

Figure 3.2-1 Key international instruments and mechanisms which address pesticide management and use, either directly or indirectly. Conventions are listed only if they have been ratified by more than 50 countries. Based on Food and Agriclture Organization of the United Nations and World Health Organization [FAO and WHO] (2014).



Note: Conventions are only listed if they are ratified by more than 50 countries.

The main provisions of the Rotterdam Convention are:

- *Notification of bans or severe restrictions*: Parties are required to inform other Parties, through the Secretariat, of any final regulatory action to ban or severely restrict a chemical. Notification of a final regulatory action is a first step that may lead to a decision by the Conference of the Parties (COP) to list a chemical in Annex III and subject it to the prior informed consent (PIC) procedure.

- *Severely hazardous pesticide formulations*: A Party which is a developing country, or a country with an economy in transition, that experiences problems caused by a pesticide formulation under conditions of use in its territory may propose its listing in Annex III as a severely hazardous pesticide formulation.

- *Prior informed consent (PIC)*: Parties are required to indicate whether they consent (with or without conditions) to future importation into their territory of chemicals listed in Annex III. Other Parties are required to respect those decisions.

- *Export notification*: Parties that plan to export a chemical which is banned or severely restricted for use within its territory are required to inform the importing Party that such export will take place.

- *Safety information*: Parties are required to ensure that both chemicals listed in Annex III and chemicals banned or severely restricted in their territory, when exported, are adequately labelled with regard to risks and/or hazards to human health or the environment, and, if the chemicals are used for occupational purposes, are accompanied by a safety data sheet;

- *Information exchange*: Parties shall facilitate the exchange of scientific, technical, economic and legal information concerning the chemicals within the scope of the Convention, including toxicological, ecotoxicological and safety information.

Given the importance of information exchange and transparency about trade in hazardous chemicals, the Rotterdam Convention and its PIC procedure have been described as a system of "global governance by disclosure" (Jansen and Dubois 2014). Nevertheless, the Rotterdam Convention covers only a small fraction of the pesticides currently traded internationally.

The Rotterdam Convention and pesticides

There are 52 chemicals listed in Annex III of the Rotterdam Convention, of which 36 are pesticides (including three severely hazardous pesticide formulations) (Rotterdam Convention n.d. b) (Table 3.2-1). In addition, more than 80 per cent of all bans and severe restrictions notified to the Convention, but not listed yet, concern pesticides. Thus a large percentage of the chemicals within the scope of the Rotterdam Convention are pesticides and the Convention can potentially have a great impact on their trade, management and use. It should be noted, however, that many of the pesticides listed in Annex III are hardly, if at all, produced and used any more.

Table 3.2-1 Pesticides listed in Annex III of the Rotterdam Convention. Rotterdam Convention (n.d.b).

Pesticide	Listed since
2,4,5-T and its salts and esters	1998
Aldrin	1998
Captafol	1998
Chlordane	1998
Chlordimeform	1998
Chlorobenzilate	1998
Dichlorodiphényltrichloroéthane (DDT)	1998
Dieldrin	1998
Dinoseb and its salts and esters	1998
EDB (1,2-dibromoethane)	1998
Fluoroacetamide	1998
HCH (mixed isomers)	1998
Heptachlor	1998
Hexachlorobenzene	1998
Lindane (gamma-HCH)	1998
Mercury compounds, including inorganic mercury compounds, alkyl mercury compounds and alkyloxyalkyl and aryl mercury compounds	1998
Methyl-parathion (emulsifiable concentrates (ECs) at or above 19.5 per cent active ingredient and dusts at or above 1.5 per cent active ingredient	1998
Pentachlorophenol and its salts and esters	1998
Phosphamidon (soluble liquid formulations of the substance that exceed 1,000 grams (g) active ingredient/litre)	1998
Binapacryl	2004
Dinitro-ortho-cresol (DNOC) and its salts (such as ammonium salt, potassium salt and sodium salt)	2004
Dustable powder formulations containing a combination of benomyl at or above 7 per cent, carbofuran at or above 10 per cent, and thiram at or above 15 per cent	2004
Ethylene dichloride	2004
Ethylene oxide	2004
Monocrotophos	2004
Parathion	2004
Toxaphene (camphechlor)	2004
All tributyltin compounds	2008
Alachlor	2011
Aldicarb	2011
Endosulfan	2011
Azinphos-methyl	2013
Methamidophos	2015
Carbofuran	2017
Trichlorfon	2017
Phorate	2019

Only eight new pesticides have been included in Annex III since 2011 (Table 3.2-1). Adding new pesticides which will be subject to the PIC procedure has been slow for several reasons. Notifications of final regulatory actions to ban or severely restrict pesticides have been made to the Convention, but many did not meet the information requirements for consideration as a potential Annex III chemical. Specific attention has therefore been given to facilitating the submission of notifications through technical assistance and the development of a Final Regulatory Action Evaluation Toolkit, among other activities. Furthermore, there have been few proposals to include SHPFs in Annex III and only two of these proposals have reached the Conference of the Parties. No consensus was reached on including these SHPFs in Annex III at the most recent Conference of the Parties (COP-9) in 2019.

Decisions to list a chemical must be taken by consensus. In a number of cases Parties have been unable to agree unanimously that certain chemicals should be added to Annex III, although proposed new listings were recommended by the Chemical Review Committee (a subsidiary body of the Convention which reviews notifications from Parties of final regulatory actions) and the COP subsequently acknowledged that the criteria for listing set out in the Convention had been met. In 2015 this difficulty was formally recognized by the Parties, which mandated a search for options to enhance the effectiveness of the Rotterdam Convention (Rotterdam Convention 2015). Options to improve the PIC procedure all appear to have serious legal and operational implications, and so far no concrete measures have been adopted (Center for International Environmental Law 2018; Rotterdam Convention 2018).

Contribution to pesticide risk reduction

As explained above, the objective of the Rotterdam Convention is not explicitly to reduce the human health and environmental risks posed by hazardous chemicals. However, its various provisions have been established "in order to protect human health and the environment from potential harm" and "to contribute to the environmentally sound use" of such hazardous chemicals.

Reduced risks of continued use of pesticides listed under the Convention

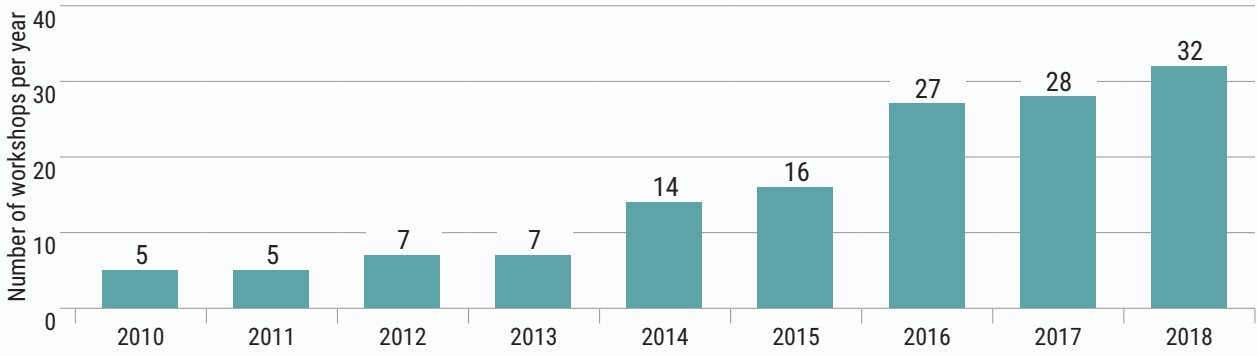
No reviews have yet been conducted to assess whether the pesticides listed in Annex III are used and managed more judiciously by Parties following their listing. However, the principal activity under the Convention is information provision and exchange on chemicals and their risks. This includes, among others, access to Parties' risk assessments and regulatory decisions; dissemination of Decision Guidance Documents (DGDs) and additional information on pesticides listed in Annex III; PIC Circulars (Rotterdam Convention n.d. c); databases on notifications of final regulatory actions and of import responses; and guidance documents on alternatives to certain pesticides within the scope of the Convention.

Capacity-building to support implementation of the Rotterdam Convention has always been an important activity under the Convention. An extensive resource kit, an e-learning tool, training materials, webinars, presentations and workshops have been developed. Some of these activities, such as workshops, have intensified in the last few years (Rotterdam Convention 2020b) (Figure 3.2-2).

Initially, much of this capacity-building focused on procedural matters related to the Convention, e.g., regarding Designated National Authorities (DNAs) or the Chemical Review Committee (CRC) and on building legal and administrative frameworks for implementation, including for transmission of information as set out in the Convention. More recently, technical assistance is also being provided on activities that are essential at the national/regional level to reduce risks posed by hazardous chemicals. Examples are surveying and monitoring pesticide use and impact, risk evaluation and assessment, import/export inspections, identification of alternatives to hazardous pesticides, and strengthening legislative frameworks on pesticide management.

Information exchange and capacity-building are cornerstones of the Convention. While there are always possibilities to further strengthen these activities, the Convention appears to have been successful in raising awareness about the trade

Figure 3.2-2 The Rotterdam Convention has greatly increased its capacity-building initiatives. Shown are the annual number of workshops organized by or with the Rotterdam Convention on pesticides and industrial chemicals combined between 2010 and 2018. Rotterdam Convention (2020b).



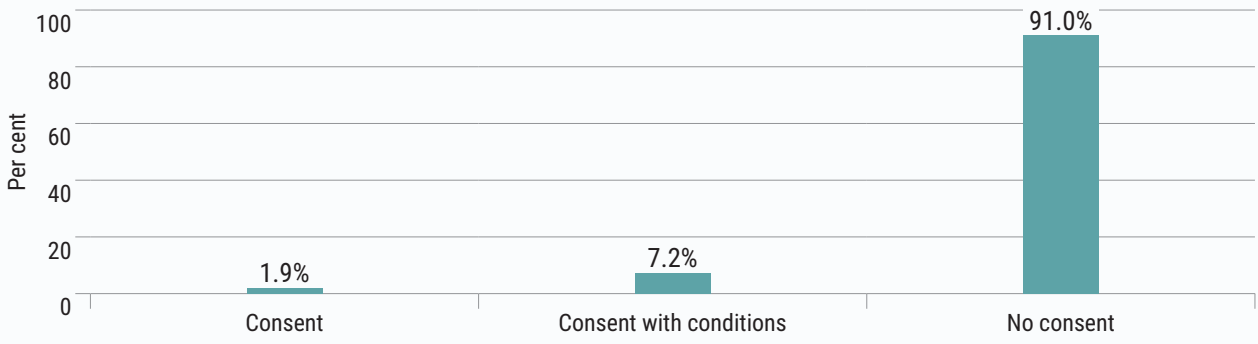
in hazardous chemicals and creating a basis for informed decision-making by Parties. In a review of the PIC procedure, Jansen and Dubois (2014) concluded that more information regarding regulatory decisions on banning and restricting the use of pesticides, together with scientific evidence of harm, had become available. Nevertheless, there are currently no data to show whether this has led, in turn, to reduced human health and environmental risks as a result of continued use of the pesticides listed under the Convention.

Reduced use of pesticides listed under the Rotterdam Convention

Slightly more information is available about the reduction or cancellation of the importation and use of pesticides within the scope of the Convention. It is assumed that reducing or cancelling the importation and/or use of a hazardous pesticide leads to a reduction of the risks it poses.

Through the PIC procedure, Parties are required to communicate a decision about whether to consent to future importation of pesticides listed in Annex III. If, as the result a pesticide being listed in Annex III, a Party decides not to consent to that pesticide being imported (or to consent to its importation only under certain conditions) and that pesticide was imported and used before listing, use of the pesticide in that country will likely be reduced. If, on the other hand, a Party's decision not to authorize the importation and use of a pesticide was taken before it was listed in

Figure 3.2-3 The large majority of responses received by the Rotterdam Convention regarding importation of the pesticides listed in Annex III were "no consent". Situation on 23 April 2019; total number of import responses was 4,404. Rotterdam Convention (2020c).



Annex III, the Convention will have no impact on reducing the use of the pesticide in that country.

By early 2019, 5,600 import responses for Annex III pesticides could in theory have been provided by Parties; slightly more than 4,400 were actually submitted (an overall response rate of about 79 per cent). Thus a large majority of import responses for pesticides listed in Annex III were submitted. More than 90 per cent of the import responses indicated "no consent" to future importation of these pesticides; 7 per cent required certain conditions for importation of the pesticide; and only 2 per cent consented to importation (Rotterdam Convention 2020c) (Figure 3.2-3). No consolidated information is available on whether these import decisions were submitted to the Secretariat of the Convention before or after a national decision on authorization to import or use was taken. While it is unclear whether listing of the pesticide in Annex III was the reason importation was halted or reduced, there are indications of the impact of listing pesticides in Annex III in other studies.

A partial evaluation was conducted by the Secretariat to assess whether listing a chemical in Annex III led to an increase in notifications of bans or severe restrictions (Rotterdam Convention 2017). Six pesticides listed from 2008 onwards were included in this study. No increase in notifications of bans or restrictions was observed following their listing. However, since many Parties had not submitted notifications of all regulatory actions to ban or restrict a pesticide, that result did not fully indicate whether such decisions had been taken by the Parties.

Another study, by the European Commission, assessed the effect of listing four pesticides (alachlor, aldicarb, monocrotophos and parathion) on trade volumes and prices (European Commission [EC] 2017a). It found there was no conclusive evidence of an impact on trade as a result of listing any of these pesticides. In three case studies both prices and trade were broadly similar before and after listing. In one case study, and in some countries, trends were identified suggesting possible effects of listing (e.g., price increase, decrease in trade, switching to alternatives).

Núñez-Rocha and Martínez-Zarzoso (2019) conducted a study of global trade flows of hazardous chemicals, identified through World Customs Organization (WCO) Harmonized Commodity Description and Coding System (HS codes) (World Customs Organization 2020) between 1995 and 2012. They concluded that when an exporting country became a Party to the Convention, and the flow was from an Organisation for Economic Co-operation and Development (OECD) country to a non-OECD country, a limited but statistically significant reduction in imports of hazardous chemicals was observed. The overall effect was a cumulative decrease in imports of about 7 per cent during the period studied.

The Stockholm Convention

The Stockholm Convention on Persistent Organic Pollutants (POPs) was adopted in 2001 and entered into force in 2004. As of the end of 2020, there were 184 Parties to the Convention (Stockholm Convention 2019a).

Objectives and provisions

The objective of the Stockholm Convention is, "mindful of the precautionary approach as set forth in Principle 15 of the Rio Declaration on Environment and Development, to protect human health and the environment from persistent organic pollutants" (Stockholm Convention 2017a).

The main provisions of the Stockholm Convention relevant to pesticides are:

- *Elimination*: Parties are required to take legal or administrative measures to eliminate the production, use, import and export of the chemicals listed in Annex A (Elimination).

- *Restriction*: Parties are required to restrict the production, use, import and export of the chemicals listed in Annex B (Restriction), in accordance with the conditions stipulated therein.

- *Importation*: Parties are required to ensure that importation of chemicals listed in Annexes A and B is only for the purpose of environmentally

sound disposal or for a use or purpose which is permitted for that Party.

- *Export*: Parties are required to ensure that a chemical listed in Annex A for which there is an exemption available, or a chemical listed in Annex B for which there is a specific exemption or an acceptable purpose available, is exported only:
  - for the purpose of environmentally sound disposal; or
  - to a Party permitted to use that chemical; or
  - to a State, not Party to the Convention, which has provided an annual certification to the exporting Party.
- *Authorization*: Parties that have regulatory and assessment schemes for new pesticides are required to prevent the production and use of new pesticides which exhibit the characteristics of persistent organic pollutants. They should also take into consideration the criteria for persistent organic pollutants when conducting assessments of pesticides currently in use.
- *Exemptions*: Parties that have a specific exemption under Annex A or Annex B shall take appropriate measures to ensure that any production or use under such exemption or purpose is carried out in a manner that prevents or minimizes human exposure and release into the environment
- *Stockpiles and waste*: Parties are required to identify stockpiles containing chemicals listed in Annex A or Annex B, as well as wastes containing such chemicals. Any wastes containing such chemicals should be handled, collected, transported, stored and disposed of in an environmentally sound manner.

The Stockholm Convention and pesticides

Of the chemicals listed in Annex A, 16 are pesticides (Stockholm Convention 2019b) (Table 3.2-2). The one pesticide listed in Annex B is dichlorodiphényltrichloroethane (DDT). DDT is only to be used for disease vector control, in accordance with the WHO recommendations and guidelines on its use and when locally safe, effective and affordable alternatives are unavailable. Few pesticides in current use have all the POPs characteristics and would meet the screening criteria which are set out in Annex D to the Convention. POPs characteristics are also considered at the early stage of development of pesticides by the pesticide industry. As a result, fewer pesticides may be listed in the future.

Table 3.2-2 Pesticides listed under the Stockholm Convention. Stockholm Convention (2019b).

Pesticide	Listed since	Pesticide	Listed since
Annex A: Elimination			
Aldrin	2001	Alpha-hexachlorocyclohexane (α-HCH)	2009
Chlordane	2001	Beta-hexachlorocyclohexane (β-HCH)	2009
Dieldrin	2001	Chlordecone	2009
Endrin	2001	Lindane	2009
Heptachlor	2001	Pentachlorobenzene	2009
Hexachlorobenzene	2001	Technical endosulfan and its related isomers	2011
Mirex	2001	Pentachlorophenol and its salts and esters	2015
Toxaphene	2001	Dicofol	2019
Annex B: Restriction			
DDT	2001		

Contribution to pesticide risk reduction

Eight of the 16 pesticides in Annex A were listed after 2004, the year the Convention entered into force. In all but one case (pentachlorophenol) listing of pesticides was done by consensus. All pesticides proposed so far by the POPs Review Committee, a subsidiary body that reviews chemicals nominated for inclusion under the Convention (Stockholm Convention 2019c), have been listed.

Evaluation of the effectiveness of protecting human health and the environment from POPs is an explicit requirement under the Stockholm Convention. The Convention provides a process for evaluating its effectiveness at intervals (Stockholm Convention 2019d). In practice this is done by evaluating, among other things, whether:

- releases from intentional production and use are eliminated or reduced;
- releases from unintentional production are eliminated or reduced;
- releases from stockpiles and wastes are eliminated or reduced;
- environmental levels of POPs are decreasing over time.

To monitor POPs in the environment, the Convention has adopted a Global Monitoring Plan (GMP) for persistent organic pollutants (Stockholm Convention 2017b; Stockholm Convention 2019e). As part of the GMP, POPs pesticides are monitored in air and human tissues. Although most POPs pesticides are no longer used, such monitoring helps to quantify "legacy risks" posed by those which are highly persistent.

The latest effectiveness evaluation was carried out in 2017 (Stockholm Convention 2017c; Stockholm Convention 2019e), while use of DDT was assessed in 2019 (Stockholm Convention 2019f). The main results of these assessments are:

- A decrease in the production, use, export and import was observed for all of the POPs pesticides initially listed in 2001 (Figure 3.2-4). However, most production and use of these pesticides had reportedly ceased before the entry into force of the Stockholm Convention in 2004 except in the case of DDT. No data were available on the continued production of more recently listed POPs pesticides. However, imports of lindane and endosulfan, listed respectively in 2009 and 2011, appeared to show a decreasing trend (Figure 3.2-4).
- Between 2010 and 2017 global DDT use for disease vector control declined by 60 per cent, from about 5,200 tons of active ingredient (a.i.) per year in the period 2003-2009 to about 2,000 tons per year in the period 2015-2017. India was the main user of DDT, primarily to control malaria and leishmaniasis. It was responsible for 95 per cent of global use in the period 2015-2017. Use of DDT for malaria control in sub-Saharan Africa was relatively minor compared to that in India, not exceeding 100 tons a.i. per year in 2015-2017. The number of Parties using POPs pesticides had greatly declined overall since 2001 (Figure 3.2-5). However, this was not the case for the number of Parties using POPs which were industrial chemicals.
- While considerable efforts had been made to dispose of stocks of obsolete POPs pesticides, particularly in Africa, significant stockpiles still existed in some low and middle income countries. These stockpiles could pose a continuing risk to the environment and human health owing to unintentional emissions and illegal use.
- For most of the POPs pesticides initially listed, concentrations in air had declined or remained at low levels due to restrictions that predated the Stockholm Convention and had been maintained at low levels since. The same was true for DDT/dichlorodiphenyldichloroethylene (DDE) levels in human tissues. For some of the more recently listed POPs pesticides, such as hexachlorocyclohexanes (HCHs), concentrations in air in some regions were beginning to show declining tendencies. However, concentrations of endosulfan in other matrices still appeared to be increasing (Stockholm Convention 2017b; Stockholm

Figure 3.2-4 Changes in the quantities of POPs imported for use from before 2004 to 2014. Stockholm Convention (2017c).

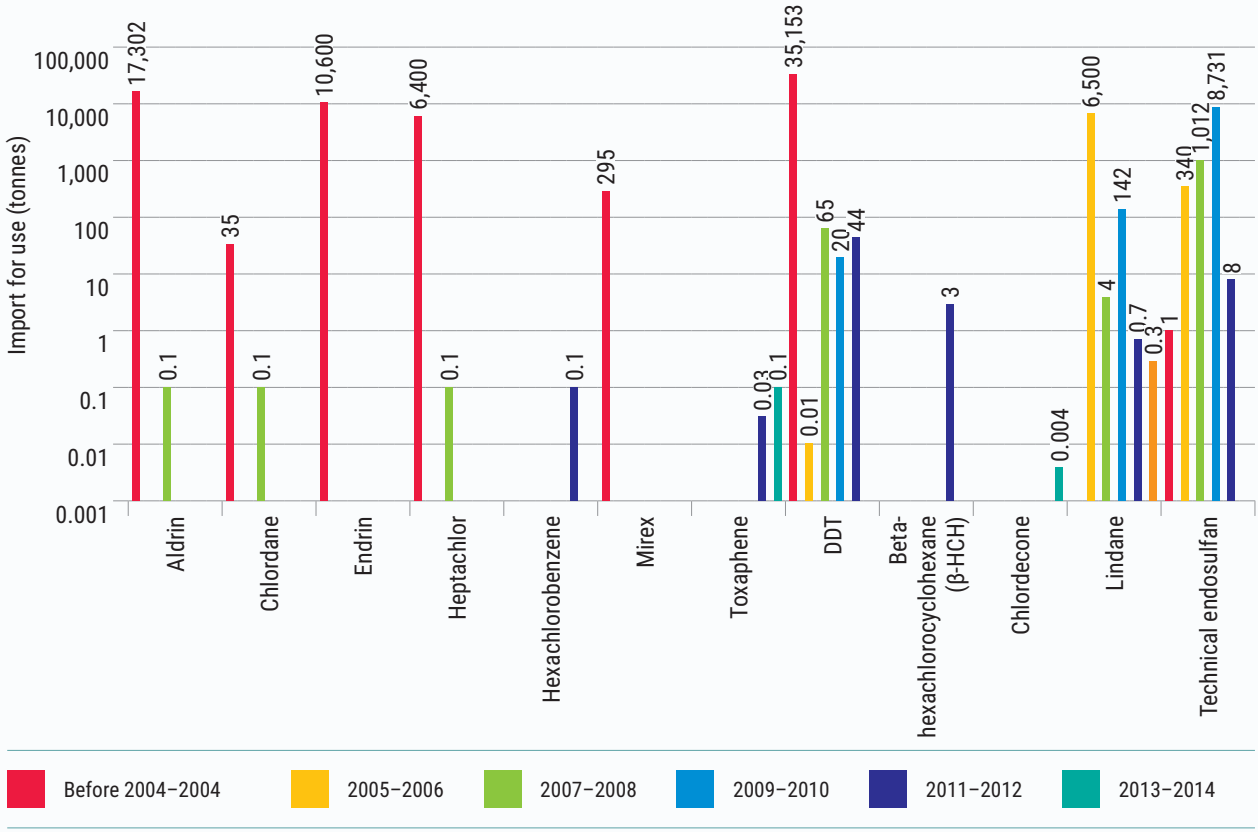
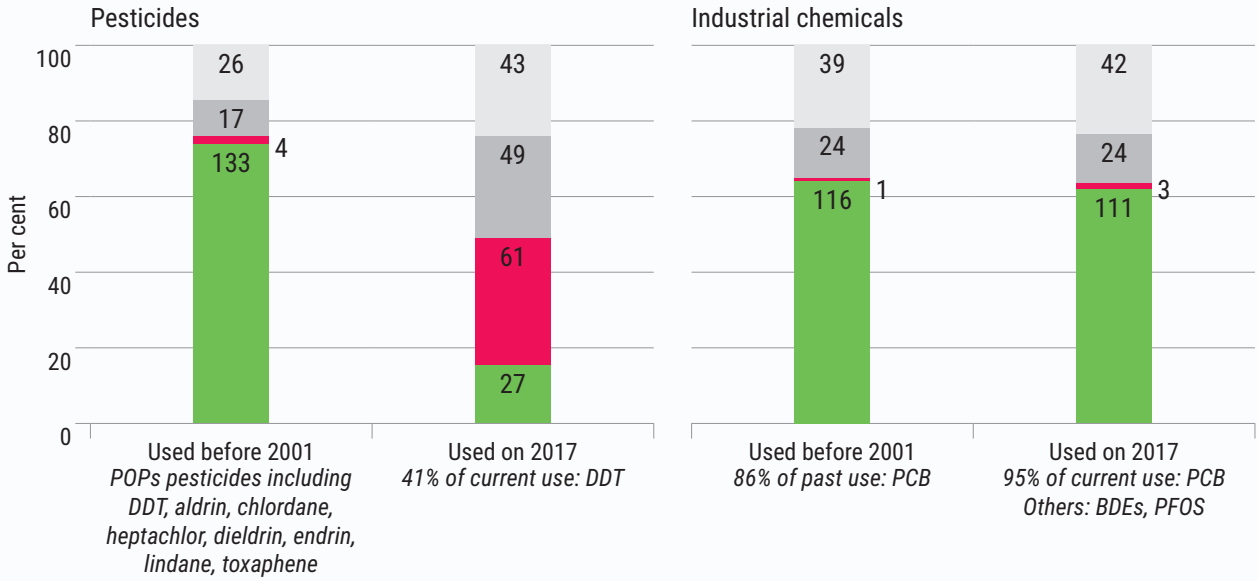


Figure 3.2-5 74 per cent of Parties used POPs pesticides in 2001, compared with 15 per cent in 2017, but the number of Parties using industrial POPs scarcely declined during the same period. Stockholm Convention (2017c).



Numbers in the bars are number of Parties.

Convention 2017c). (See Chapter 4.3.2 for more information on pesticide concentrations in the environment.)

The study by Núñez-Rocha and Martínez-Zarzoso (2019) of global trade flows of hazardous chemicals identified through Harmonized System (HS) Commodity Codes (World Customs Organization 2020) from 1995 to 2012 showed a reduction of about 16 per cent in the trade of POPs by importing countries which had ratified the Convention and in that of POPs shipped from OECD to non-OECD countries.

Another contribution to pesticide risk reduction by the Stockholm Convention is the progressive inclusion of (certain) screening criteria for POPs, set out in Annex D, in regional pesticide management policy and/or national pesticide legislation or decision-making procedures. Consequently, pesticides that have several POPs characteristics are less likely to be authorized for use in these countries. That is the case, for example, in member countries of the Permanent Interstate Committee for Drought Control in the Sahel (CILSS), member countries of the Southern African Development Community, Myanmar, and Member States of the European Union (EU), among others.

The Basel Convention

The Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal was adopted in 1989 and entered into force in 1992. As of late 2020, there were 188 Parties to the Convention (Basel Convention 2020a). An amendment to the Convention, often referred to as the Ban Amendment, was adopted in 1995 and entered into force on 5 December 2019 (Basel Convention 2020b). A Protocol on Liability and Compensation for Damage Resulting from Transboundary Movements of Hazardous Wastes and their Disposal, adopted in 1999, has not yet entered into force (Basel Convention 2019; United Nations Treaty Collection 2020).

Objectives and provisions

The overarching goal of the Basel Convention is to protect human health and the environment from the adverse effects that may result from the generation and management of hazardous and other wastes. The Convention regulates transboundary movements (imports and exports) of hazardous wastes and other wastes. It obliges Parties to ensure that such wastes are managed and disposed of in an environmentally sound manner.

The Convention applies to hazardous and other wastes. Annexes I and VIII list wastes that are classified as hazardous and are subject to the control procedures under the Convention. Annex II identifies wastes that require special consideration ("other wastes").

The provisions of the Convention centre around the following principal aims:

- reduction of hazardous waste generation and promotion of environmentally sound management of hazardous wastes, wherever the place of disposal;
- restriction of transboundary movements of hazardous wastes, except where these are perceived to be in accordance with the principles of environmentally sound management;
- a notice and consent regulatory system applicable to cases in which transboundary movements are permissible.

The Basel Convention and pesticides

Wastes from the production, formulation and use of pesticides (including pesticides which are off-specification, outdated or unfit for their originally intended use), as well as wastes from the manufacture, formulation and use of wood preserving chemicals, fall within the definition of hazardous wastes under the Convention.

For hazardous wastes, including pesticide wastes, the Convention stipulates, among many other measures, that Parties have obligations to:

- ensure that their generation is reduced to a minimum;
- ensure the availability of adequate disposal facilities for their environmentally sound management;
- ensure that the transboundary movement of hazardous wastes and other wastes is conducted in a manner that will protect human health and the environment against the adverse effects which may result from such movement;
- not allow the export of wastes to a Party which has prohibited all imports, or if it has reason to believe that the wastes in question will not be managed in an environmentally sound manner in the importing country. Furthermore, a Party shall not permit wastes to be exported to a non-Party or be imported from a non-Party (except in compliance with Article 11 of the Convention);
- when exporting a hazardous waste, including pesticide waste, notify the competent authority of the countries concerned of any proposed transboundary movement of this waste.

In addition, Parties listed in Annex VII (EU Member States, OECD member countries, and Liechtenstein) which have agreed to be bound by the Basel Convention have an obligation to prohibit all transboundary movements of hazardous wastes destined for final disposal operations from Annex VII States to States not listed therein, and to do the same with regard to certain hazardous wastes destined for recovery or recycling operations

The Parties to the Convention have adopted a series of technical guidelines, including on the environmentally sound management of wastes consisting of, containing or contaminated with POPs listed under the Stockholm Convention.

For hazardous wastes including pesticides, general guidance has been developed for their environmentally sound management as well as technical guidance for treatment and disposal that will also apply to some current use pesticide wastes.

Contribution to pesticide risk reduction

An evaluation of the Basel Convention's effectiveness is required at least every six years. To that end, the Conference of the Parties has adopted a number of indicators for measuring achievement and performance in regard to its Strategic Framework for the Implementation of the Basel Convention for 2012-2021 (Basel Convention 2011; Basel Convention 2020c). Most of these indicators measure the processes or outputs of the Convention, but none address the impacts of reducing hazardous waste generation, the degree of implementation of environmentally sound management of hazardous wastes, or the restriction of transboundary movements of hazardous wastes. Only 19 per cent of the Parties responded to setting a baseline for the indicators (Basel Convention 2014); the final evaluation is expected to be available in 2021.

A review of certain provisions of the Basel Convention was conducted by Kellenberg and Levinson (2014). Trade flows of hazardous wastes from 1988 to 2008 among countries that did and did not ratify the Convention or the Ban Amendment were assessed. The authors concluded that trade in hazardous wastes had greatly increased during the study period and that the Convention had had no effect on the growth of this trade. Reportedly, this may be due in part to the increasing quantities of used electronics and other materials being shipped for recycling, which are also classified as wastes under the Convention. Since no distinction was made in this study between pesticide wastes and other hazardous wastes, it is not clear whether the conclusions apply to pesticide wastes.

It should be noted, however, that the Convention does not aim to eliminate or reduce trade in wastes. Rather, it puts in place trade control measures for transboundary movements of the wastes within its scope.

The Montreal Protocol

The Montreal Protocol on Substances that Deplete the Ozone Layer was adopted in 1987 and entered into force in 1989. The Protocol has been ratified by 198 parties, and as at 4 February 2021

four of its five amendments had been ratified by 197 parties while the fifth one, the Kigali Amendment (adopted in October 2016), had been ratified by 113 parties (UNEP 2020a).

Objectives and provisions

The objective of the Montreal Protocol is to reduce and eliminate the production and consumption of man-made chemicals that destroy the stratospheric ozone layer and harm the climate. This is to be done by controlling (i.e., eliminating or strongly reducing) the production and use of these chemicals, researching and promoting alternatives, and assisting countries to implement such alternatives.

The Montreal Protocol and pesticides

The only pesticide covered by the Montreal Protocol is methyl bromide. This pesticide was used extensively in the past as a fumigant to control a wide range of pests and pathogens present in soils, in post-harvest storage of commodities and in structures. Since alternatives were available for most such applications, they were classified under the Protocol as "controlled uses" which needed to be eliminated (UNEP 2019a).

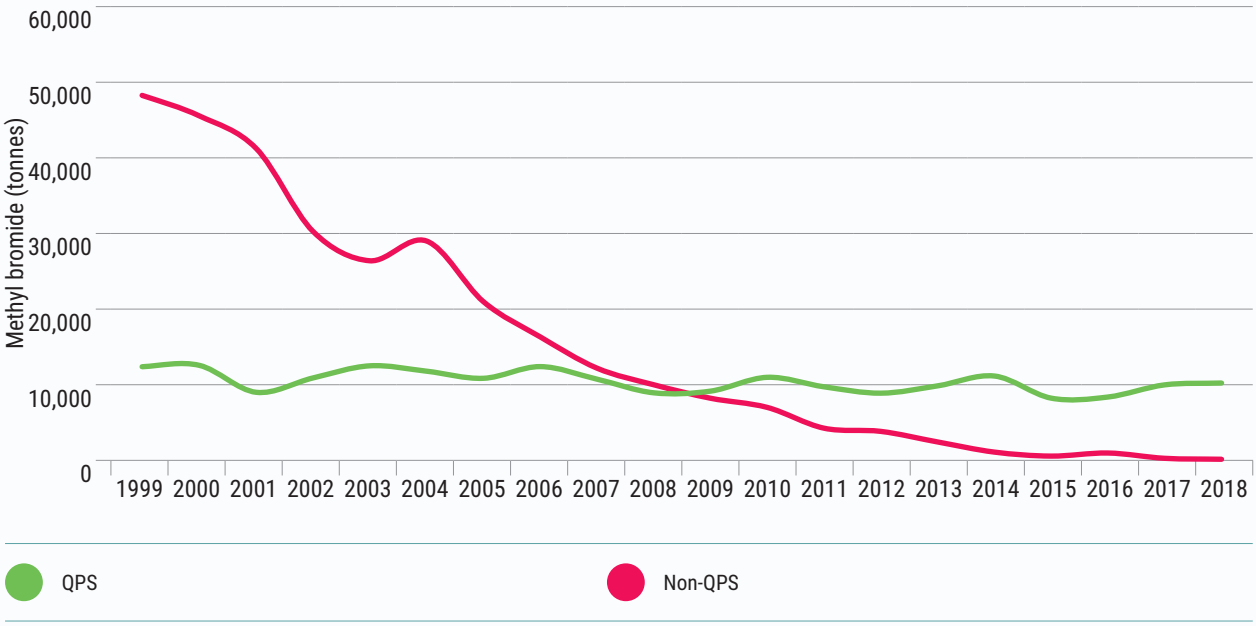
The Protocol includes a provision for "critical uses" which applies to specific cases in which technically or economically viable alternatives to methyl bromide are unavailable. Exemptions are granted annually under this provision on a case-by-case country basis, but requested amounts have fallen by 99 per cent since 2005 as alternatives have been adopted (the Ozone Secretariat, personal communication).

In addition, methyl bromide continues to be used on various traded goods as phytosanitary treatment to control pests and pathogens of quarantine importance. These treatments are known as "Quarantine and Pre-shipment" (QPS) uses. They usually take place before a country exports the traded goods or upon their arrival in the importing country. QPS uses of methyl bromide are not controlled under the Montreal Protocol, but there are annual data reporting requirements (UNEP 2019a).

Contribution to pesticide risk reduction

Global consumption of methyl bromide for controlled uses was reported to be 64,420 tons in 1991 and remained above 60,000 tons until 1998. By 2017 global consumption had fallen

Figure 3.2-6 Estimated global consumption of methyl bromide 1999-2018 for quarantine and pre-shipment (QPS) uses and non-QPS uses. Montreal Protocol (2018) and the Ozone Secretariat (personal communication).



to 245 tons for critical uses, a reduction of 99.6 per cent (Montreal Protocol 2018).

QPS uses of methyl bromide are exempted under the Protocol, but are closely monitored and accounted for. Exempted QPS uses of methyl bromide amounted to about 10,000 tons in 2017, a level that has remained stable during the last two decades (Figure 3.2-6).

Approximately 10,000 tons of methyl bromide are still produced on an annual basis, 86 per cent down from a peak of 74,000 tons in 1991 (Montreal Protocol 2018). The Montreal Protocol has thus been very effective in reducing controlled uses of methyl bromide. QPS uses, however, remain unchanged, representing about 15 per cent of production at the time of entry into force of the Protocol.

The Methyl Bromide Technical Options Committee has indicated that about 35-40 per cent of current QPS uses could be replaced with readily available chemical and non-chemical alternatives, and has provided lists of such alternatives to methyl bromide for QPS uses. Many of the alternatives have been approved by the International Plant Protection Convention, which in principle allows international trade to occur using these quarantine and pre-shipment treatments. However, it is not clear to what extent such alternatives will replace QPS uses of methyl bromide in the short term, as legal, economic and practical constraints still exist.

The Minamata Convention

The Minamata Convention on Mercury is a global treaty was adopted in 2013. It entered into force in 2017. By late 2020 there were 125 Parties to the Convention (UNEP 2020b).

Objectives and provisions

The objective of the Minamata Convention is to protect human health and the environment from anthropogenic emissions and releases of mercury and mercury compounds.

Pesticides and the Minamata Convention

Pesticides containing mercury are listed under "mercury-added products" in Annex A of the Convention. The Convention stipulates that such products should be phased out by all Parties unless a Party demonstrates that it has already reduced to a de minimis level the manufacture, import and export of the large majority of the product(s).

The manufacture, import or export of pesticides, biocides and topical antiseptics containing mercury is not allowed after 2020. So far, four countries have requested and received exemptions from the phase-out of mercury-containing pesticides until 2025.

A wide variety of mercury compounds have been applied in the past as fungicides, herbicides, insecticides and microbicides. Based on current inventories, however, pesticides are no longer a relevant source of mercury emissions (Minamata Convention 2013; Minamata Convention 2019).

Contribution to pesticide risk reduction

Mercury-based pesticides have been listed in Annex III of the Rotterdam Convention since 1998. More than 95 per cent of the Parties to the Rotterdam Convention have indicated that they did not consent to the importation of such pesticides (Rotterdam Convention 2020c), suggesting that significant risk reductions may no longer be reported in this product category in Annex A.

World Trade Organization agreements

Within the World Trade Organization (WTO) three agreements have a direct influence on pesticide management: the Agreement on the Application of Sanitary and Phytosanitary Measures (SPS Agreement), the Technical Barriers to Trade (TBT) Agreement, and the Agreement of Trade-Related Aspects of Intellectual Property Rights (TRIPS).

The SPS Agreement

The Agreement on the Application of Sanitary and Phytosanitary Measures (SPS Agreement) entered into force with the establishment of the WTO on

1 January 1995. It concerns the application of food safety and animal and plant health regulations, including pesticide residues in food (World Trade Organization [WTO] 2010).

The main objective of the SPS Agreement is to ensure that food is safe according to national levels of protection while, at the same time, avoiding the use of health and safety regulations as an excuse to protect domestic producers from foreign competition. To this end, the SPS Agreement recognizes the right of governments to take measures to protect human, animal and plant health as long as these measures are based on science, are necessary for the protection of health, and do not unjustifiably discriminate among foreign sources of supply. At the same time, it encourages governments to "harmonize" or base their national measures on international standards. For pesticides these standards are the maximum residue limits (MRLs) established by the Codex Alimentarius Commission.

On average, about a quarter of all notifications under the SPS Agreement are pesticide-related (i.e., related to MRLs), indicating the importance of pesticide MRLs under this agreement (Figure 3.2-7). There have been no SPS-related disputes concerning pesticide MRLs to date (WTO Agriculture and Commodities Division, personal communication).

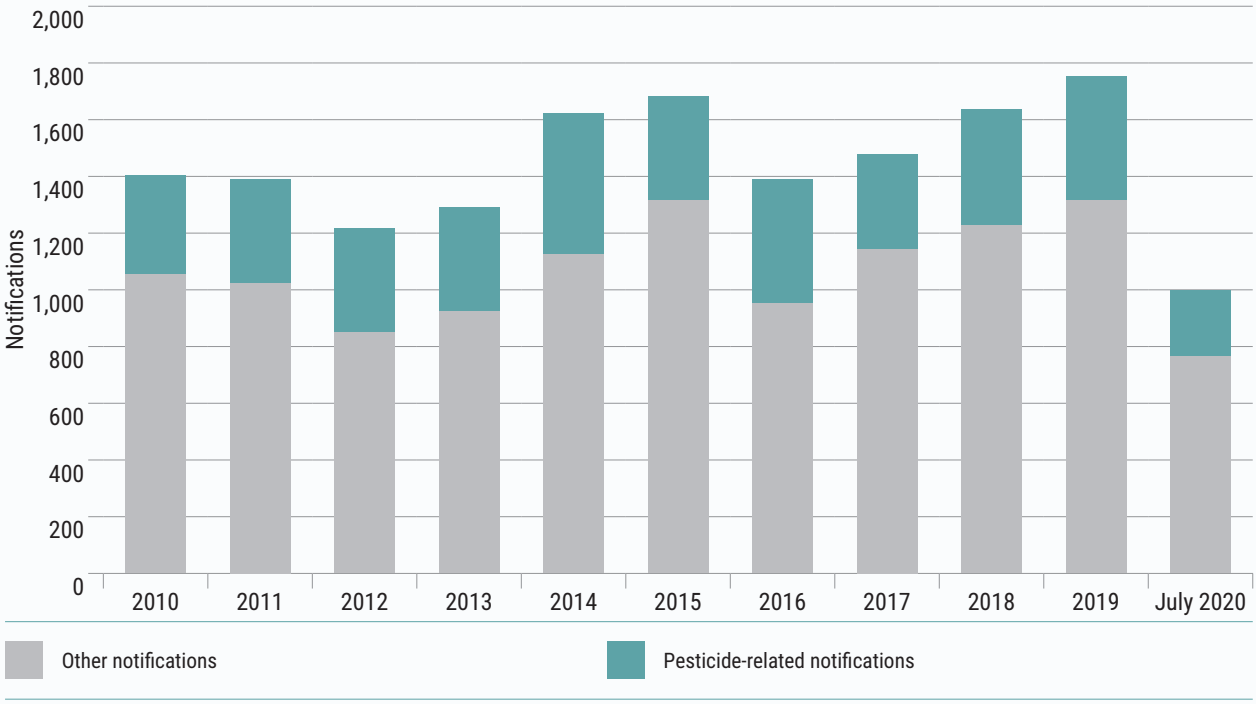
The SPS Agreement is accompanied by a Standards and Trade Development Facility (STDF) which provides technical support to developing countries to build their capacity to implement international standards and gain and maintain market access (Standards and Trade Development Facility 2020).

Figure 3.2-7 Pesticide-related notifications under the Agreement on the Application of Sanitary and Phytosanitary Measures (SPS Agreement) as a share of all SPS notifications, January 2010-July 2020. World Trade Organization (Agriculture and Commodities Division).

TBT Agreement

The WTO Agreement on Technical Barriers to Trade (TBT Agreement) entered into force with the establishment of the WTO on 1 January 1995.

Figure 3.2-7 **Pesticide-related notifications under the Agreement on the Application of Sanitary and Phytosanitary Measures (SPS Agreement) as a share of all SPS notifications, January 2010-July 2020.**
World Trade Organization (Agriculture and Commodities Division, personal communication).



The TBT Agreement aims to ensure that technical regulations, standards, and conformity assessment procedures are non-discriminatory and do not create unnecessary obstacles to trade. At the same time, it recognises WTO members' right to implement measures to achieve legitimate policy objectives, such as the protection of human health and safety, or protection of the environment (WTO 2014).

SPS or TBT?

If a measure is applied to protect human or animal life from risks arising from pesticides in food, beverages or feedstuffs (i.e., food safety issues), SPS applies. On the other hand, TBT applies to other technical regulations and voluntary standards issued to protect the environment or human health from risks posed by pesticides (i.e., not related to food safety). These include specifications to ensure that the pesticide works effectively, specifications to protect farmers from possible harm from handling pesticides (e.g., packaging requirements, usage labelling) and classification of substances, among others.

In some cases WTO members may have a single regulation which jointly covers food safety issues related to pesticides (i.e., SPS-related concerns) and other types of safety concerns, as indicated above (i.e., TBT-related concerns). These types of regulations are normally notified under both Agreements, since the single regulation covers multiple areas (WTO Agriculture and Commodities Division, personal communication).

TRIPS Agreement

The Agreement on Trade-Related Aspects of Intellectual Property Rights (TRIPS Agreement) also entered into force with the establishment of the World Trade Organization (WTO 2017). It sets minimum standards of protection for intellectual property rights. Another key objective of TRIPS is that intellectual property protection should contribute to technical innovation and the transfer of technology.

Of particular relevance to pesticide registration are the provisions on undisclosed information (e.g., trade secrets and test data) under Article 39. The TRIPS Agreement stipulates that test data submitted to governments in order to obtain marketing approval for new agricultural chemicals must be protected against unfair commercial use. In addition, such data must be protected against disclosure, except where necessary to protect the public, or unless steps are taken to ensure that the data are protected against unfair commercial use.

In principle, pesticide regulatory authorities from countries that are members of the WTO should put into place measures to implement TRIPS provisions for data protection.

Contribution to pesticide risk reduction

The contribution of the SPS agreement to pesticide risk reduction depends to a large extent on the level of acceptance by Parties of the Codex Alimentarius MRLs. A large number of countries indeed appear to adopt, or make use of, Codex pesticide MRLs (Chapter 3.2.3; Codex Alimentarius).

The contribution of the TRIPS agreement to pesticide risk reduction depends to a large extent on the level of protection and exclusive use of data submitted to national and regional regulatory authorities for the registration of pesticides. No global review of this aspect is available. So far, no disputes regarding the failure to protect against unfair commercial use of test data submitted for market approval of pesticides (TRIPS Article 39) have been treated by WTO.

OECD – Mutual Acceptance of Data (MAD)

The Organisation for Economic Co-operation and Development (OECD) developed the Mutual Acceptance of Data (MAD) system to avoid conflicting or duplicative national requirements, provide a common basis for cooperation among national authorities, and avoid creating non-tariff barriers to trade. The MAD system is based on three OECD Council Decisions (Organisation for Economic Co-operation and Development [OECD] 2020a):

- The 1981 Council Decision on the Mutual Acceptance of Data in the Assessment of Chemicals (revised in 1997), which states

that test study data generated in any member country in accordance with OECD Test Guidelines and Principles of Good Laboratory Practice (GLP) shall be accepted in other member countries for assessment purposes and other uses relating to the protection of human health and the environment.

- The 1989 Council Decision-Recommendation on Compliance with Principles of Good Laboratory Practices, which establishes procedures for monitoring GLP compliance through government inspections and study audits as well as a framework for international liaison among monitoring and data-receiving authorities.

- The 1997 Council Decision on the Adherence of Non-Member countries to the Council Acts related to the Mutual Acceptance of Data in the Assessment of Chemicals that sets out a step-wise procedure for non-OECD economies to take part as full members in this system.

The objective of the MAD system is to harmonize chemical health and safety testing (including of pesticides) and ensure sharing of such data among OECD member countries as well as non-member adherents. In addition to the 36 OECD member countries, six non-member countries are currently full adherents to MAD.

OECD member countries and adherents to MAD must mutually accept non-clinical health and safety study data if:

- the study has been conducted according to OECD Test Guidelines and OECD Principles of GLP; and

- the study has been conducted in a test facility which has been inspected by a national GLP compliance monitoring programme; and

- the national GLP compliance monitoring programme has undergone a successful evaluation by OECD.

This is the concept of "tested once, accepted for assessment everywhere". However, while the receiving government must accept the study, how it interprets study results is its own prerogative.

The OECD Test Guidelines have been developed for the determination of physico-chemical properties, ecotoxicological studies, environmental fate and behaviour, and human toxicology and health effects, as well as for other areas (e.g., biocides).

Contribution to pesticide risk reduction

The OECD Principles of GLP and Test Guidelines have become a global standard for generation of health and safety data for pesticide registration, including in countries that do not formally adhere to the MAD system. It has been estimated that by reducing duplication of testing, and creating a framework for the sharing of work, the MAD system saves governments and industry around 309 million euros each year, as well as reducing the number of animals used in such testing (OECD 2020b).

Furthermore, it can be argued that through the international review and harmonization of testing methodologies, and the application of the GLP system, pesticide studies are of higher quality, leading to better risk assessments. The MAD system also facilitates exchange of evaluations among regulatory authorities, as well as joint reviews.

3.2.3 Voluntary international instruments and mechanisms that directly address pesticides

International Code of Conduct on Pesticide Management

Objectives and provisions

The International Code of Conduct on Pesticide Management is one of the oldest international instruments addressing the sound management of chemicals (FAO and WHO 2014). Its elaboration started in the early 1980s and the first version was adopted by the FAO Conference in 1985, then under the name International Code of Conduct on the Distribution and Use of Pesticides (Table 3.2-3).

Table 3.2-3 History of the Code of Conduct.

Year of adoption	Version	Main modifications
1985 (FAO Conference)	International Code of Conduct on the Distribution and Use of Pesticides	
1989 (FAO Conference)	Amended version	Inclusion of the prior informed consent (PIC) procedure
2002 (FAO Council)	Revised version	Deletion of the PIC procedure due to the adoption of the Rotterdam Convention Strengthening of the life cycle concept Strengthening of monitoring and observance provisions
2013 (FAO Conference) 2014 (WHO Executive Board endorsement)	International Code of Conduct on Pesticide Management	Incorporation of public health pesticides and vector control Strengthening of health and environmental aspects Introduction of the concept of Highly Hazardous Pesticides (HHP) Co-published by WHO

Table 3.2-4 Outline of the Code of Conduct. (FAO and WHO 2014).

Article		Article	
1	Objectives of the Code	7	Availability and use
2	Terms and definitions	8	Distribution and trade
3	Pesticide management	9	Information exchange
4	Testing of pesticides	10	Labelling, packaging, storage and disposal
5	Reducing health and environmental risks	11	Advertising
6	Regulatory and technical requirements	12	Monitoring and observance of the Code

The Code of Conduct was originally elaborated to establish standards of conduct for public and private entities engaged in the distribution and use of pesticides. It was aimed, in particular, at countries that had no or inadequate pesticide legislation. Subsequent revisions have included the prior informed consent (PIC) procedure, which later developed into the Rotterdam Convention; strengthening of the life cycle concept for pesticide management; and inclusion of public health pesticides and enhanced health and environmental protection. The title of the latest version of the Code of Conduct was broadened to cover all aspects of pesticide management and was published jointly by FAO and WHO.

From a tool to assist countries with inadequate pesticide legislation in the 1980s, the Code of Conduct has developed into a broad voluntary framework for the sound management of pesticides endorsed by all FAO member countries, WHO, and key pesticide industry and civil society organizations.

The provisions of the Code of Conduct are organized around 12 articles covering the main aspects of pesticide regulation, use and management (Table 3.2-4).

Implementation

Since the Code of Conduct was first adopted, FAO has spearheaded its implementation through a variety of activities. A large number of technical assistance projects have been implemented over the years in all parts of the world, focusing on specific provisions of the Code such as strengthening pesticide legislation, capacity-building for pesticide evaluation and registration, improving and harmonizing pesticide

efficacy and residue testing, reducing health and environmental risks, disposing of obsolete pesticides, and management of empty containers.

Furthermore, a considerable number of technical and policy guidelines have been published by FAO and WHO to support the implementation of specific provisions of the Code of Conduct (FAO 2020a; WHO 2020c). FAO has also developed an on-line Pesticide Registration Toolkit to support pesticide registration staff in countries with limited resources (FAO 2020b).

A Panel of Experts has advised FAO on the implementation of the Code of Conduct since its inception. From 2007, WHO has nominated experts to join these discussions, leading to the establishment of the FAO/WHO Joint Meeting on Pesticide Management (JMPM).

Other stakeholders have also taken actions to implement the Code of Conduct. Adherence to the Code is a requirement for membership in CropLife International (CLI). This organization has developed a guide and an e-learning tool for its members (CropLife International [CLI] 2017). It also has a product stewardship programme focusing on responsible distribution and use of pesticides (Chapter 3.5.2).

Shortly after the Code of Conduct was first issued, the International Organization of Consumers Unions (IOCU) and the Pesticide Action Network (PAN) developed a Citizens' Action Guide to the Code of Conduct (Goldenman and Rengam 1987). PAN also published a consolidated guide to the chemical tools and conventions (Goldenman and Pozo Vera 2008) which provided a checklist for implementation of the Code of Conduct. A dedicated Code monitoring module has been developed to help concerned organisations to monitor compliance with the Code of Conduct by governments and industry (Pesticide Action Network Asia and Pacific and Pesticide Action Network UK 2016). In addition, PAN carries out projects that promote implementation of parts of the Code of Conduct, in particular on alternatives to more hazardous pesticides.

Contribution to pesticide risk reduction

The status of implementation of the Code of Conduct has been evaluated several times since its original adoption in 1985. A baseline survey was conducted in 1986 by FAO, and subsequently several surveys have been organized by FAO and/or WHO (Table 3.2-5). All surveys targeted government bodies responsible for pesticide management, generally those responsible for agriculture and for public health.

While the first two surveys were practically identical, later surveys targeted different government bodies and included a variety of aspects and questions, which makes comparison of the outcomes difficult. Despite this variability, however, some trends can be distinguished for certain key provisions of the Code that were included in most of the surveys (Table 3.2-6).

The number of countries that have legislation concerning the marketing, distribution, sales and use of pesticides has significantly increased during the last 30 years. On the other hand, such legislation is clearly lagging behind in the case of public health pesticides. Pesticide

Table 3.2-5 Surveys conducted by FAO and WHO on implementation of the Code of Conduct.

Survey year	Number of responding countries (response rate)	Responding government bodies	Reference
1985	119 (75%)	Agriculture	FAO (1993)
1993	91 (51%)	Agriculture	FAO (1996)
2003	71 (57%)	Health	WHO (2004)
2008	39 (21%)	Agriculture	FAO (2010)
2010	113 (80%)	Health	Matthews *et al.* (2011); WHO (2011a)
2018	52-94 (27-48 %)	Agriculture, health	FAO/WHO (2019c); van den Berg *et al.* (2020)

Table 3.2-6 Trends in the implementation of certain key provisions of the Code of Conduct (percentage of responding countries agreeing with these statements) (for sources, see Table 3.2-5).

Statement on implementation of provisions of the Code of Conduct	Percentage of positive responses						Trend
	1985	1993	2003	2008	2010	2018	
National legislation for marketing and use of agricultural pesticides is in place	80	96	95	97	84	95	Unchanged since 1993
National legislation for marketing and use of public health pesticides is in place	[1]		13	32	74	60	Improved
Pesticides must be registered before they are placed on the market	66	78		96/100[2]			Improved
Pesticide distributors and retailers are regulated	53	57			80 (ag)/ 65 (ph)[3]	79 (ag)/ 47 (ph)	Improved?[4]
Pest control operators are regulated			62		70		Improved?
Pesticides are labelled adequately	52	51		77/94			Improved
The quality of pesticides on the market is inadequate (i.e. substandard, counterfeit)	50	48		68/32	67	40	Unchanged
Regulators have access to pesticide quality control laboratories	11	22	49	37/78	50	59	Improved, but stagnated
IPM/IVM successfully promoted	50	55		32/63	62	76	Improved
Monitoring of pesticide residues in food and feed is in place				21/89		58	Unchanged?
Monitoring of pesticide residues in the environment is in place	42	11		21/75		26	Unchanged
Pesticide import and/or use statistics are collected	45	65	68	100/94	78	88	Improved

1 = data unavailable
2 in red: low and lower middle income countries; in green: upper-middle and high income countries (according to World Bank classification, at the time of the survey)
3 ag = retailers of agricultural pesticides; ph = retailers of public health pesticides
4 ? = trend not clear

labelling appears to have improved over time. Pesticide quality, on the other hand, is still considered inadequate in about half of countries, while access to quality control laboratories has stagnated since the early 2000s. The successful promotion of IPM and IVM as preferred pest management approaches is reported from upper middle and high income countries, but much less so from lower middle and low income ones. More governments throughout income groups are collecting statistics on the importation and/or use of pesticides. However, monitoring of pesticide residues in food and feed does not seem to have improved much, especially in lower income countries, while surveillance of such residues in the environment is only conducted in a quarter of all countries, with no significant improvement over the last three decades.

Implementation of Article 3.6 on avoiding the use of pesticides that require personal protective equipment that is unsuitable for small-scale users in hot and humid climates – an important provision for minimizing occupational exposure to pesticides – has not been assessed by any of the surveys.

The surveys conducted so far by FAO and WHO are self-reporting: government representatives are asked to provide information on implementation of the Code of Conduct. No comprehensive independent evaluation of the effectiveness of the Code of Conduct in improving sound pesticide management has ever been conducted. An independent and comprehensive impact assessment of the main provisions of the Code of Conduct could provide valuable insights on

achievements and constraints regarding sound pesticide use and management. Given the breadth of the Code of Conduct, certain of its provisions have, however, been reviewed separately.

Orozco *et al*. (2009) reframed certain articles of the Code of Conduct in terms of farmers' rights and assessed through surveys, farmer focus groups and direct observations the extent to which these rights were respected in Ecuador and Peru. In particular, they evaluated access to information and training about pesticides and alternatives, availability of good quality products, and PPE. Their assessment was that adherence to provisions of the Code by pesticide companies and the government was far less than adequate.

Member organizations of the Pesticide Action Network (PAN) conducted community monitoring in 13 countries in Africa, Asia and Latin America between 2007 and 2009 (PAN 2010). They assessed pesticide use practices, use of personal protective equipment and self-reported symptoms of pesticide poisoning. Based on this survey, their assessment was that 25 years after initial publication of the Code of Conduct, pesticides in these regions were still exposing farmers to significant health risks.

Monitoring observance of the Code of Conduct is explicitly included in its Article 12. Governments are invited to monitor observance and report on progress on implementation of the Code; pesticide industry to report on stewardship; and non-governmental organizations (NGOs) and other interested entities to monitor activities related to the implementation of the Code. (FAO and WHO 2014). FAO further published Guidelines on Monitoring and Observance of the Code of Conduct to facilitate monitoring by third parties (FAO 2006a). This publication describes a procedure for submission and treatment of both regular and ad hoc monitoring reports.

According to the Code of Conduct, monitoring reports can be addressed to the Directors-General of FAO and WHO and the Executive Director of UNEP. In practice these reports are reviewed by the FAO Panel of Experts on Pesticide Management (FAO 2006), which has been superseded de facto by the FAO/WHO Joint Meeting on Pesticide Management (JMPM).

Since these guidelines were published, three formal submissions of ad hoc monitoring reports have been made by a university and several NGOs and reviewed by the JMPM in 2007 and 2017 (FAO and WHO 2007; FAO and WHO 2017). The latest monitoring report indicated non-adherence to several provisions of the Code by certain multinational pesticide companies. The JMPM heard arguments from submitters and the implicated pesticide companies. However, the submitters subsequently argued that the JMPM response was not concrete enough and would allow such practices to continue (Public Eye 2017). The JMPM began a revision of its guidance on this topic in 2019 (FAO and WHO 2019b).

Because the Code of Conduct is a voluntary instrument, no legally formalized procedures exist for reporting and treating alleged non-observance of the Code; only guidance has been published by FAO. However, to stimulate monitoring by third parties and ensure that outcomes of such activities can be translated into concrete actions that strengthen effective implementation of the Code, the manner in which these reports are reviewed and published by FAO and WHO will require further attention.

It may be concluded that the contribution of the Code of Conduct to pesticide risk reduction is variable, with advances in some aspects of pesticide management and stagnation with regard to many others. Overall, low income countries have the highest incidence of gaps in pesticide life cycle management when compared to the provisions covered by the Code of Conduct (van den Berg *et al*. 2020).

Highly Hazardous Pesticides

Objectives and provisions

The International Code of Conduct currently defines Highly Hazardous Pesticides (HHPs) as "pesticides that are acknowledged to present particularly high levels of acute or chronic hazards to health or environment according to internationally accepted classification systems

such as WHO or GHS or their listing in relevant binding international agreements or conventions. In addition, pesticides that appear to cause severe or irreversible harm to health or the environment under conditions of use in a country may be considered to be and treated as highly hazardous" (FAO and WHO 2014).

While there is no formal international instrument or mechanism specifically addressing HHPs, various initiatives have been taken to reduce the risks posed by HHPs. The main justification for national governments to consider more strict regulation of HHPs is that they consist of a limited number of pesticides which are estimated to cause a disproportionate portion of human health and environmental adverse effects, particularly in lower income countries.

International criteria to identify HHPs were established by the FAO/WHO JMPM in 2007 and formally published by FAO and WHO in 2016 (Table 3.2-7). Their main objective was to identify a limited number of pesticides that were most likely, under use conditions in lower income countries, to cause a major share of adverse effects on human health and the environment.

Criteria for the toxicity of pesticides go back to the first Recommended Classification of Pesticides by Hazard adopted by WHO in 1975, which recognized the classification of pesticides as extremely or highly hazardous to human health (Copplestone 1988). Although the designation of pesticides as HHPs takes into account both their human health and environmental aspects, current FAO and WHO criteria still mainly cover human health hazards. A broader set of HHP criteria, which includes more human health and environmental aspects, is being applied by the Pesticide Action Network International (Pesticide Action Network 2021). The pesticide industry, on the other hand, has long promoted a narrower set of criteria for HHPs than FAO and WHO although it now follows the FAO and WHO criteria (CLI 2020a).

For many years NGOs and a number of scientists have called on national governments to consider stricter regulation or prohibition of HHPs as a way to reduce the risks of pesticide use (Reeves *et al*. 1999; Dinham and Malik 2003; Konradsen *et al*. 2003). In the mid-2000s prohibition of highly toxic or hazardous pesticides also appeared on the agendas of international platforms such as the Intergovernmental Forum on Chemical Safety (IFCS) and the FAO Council (Table 3.2-8).

HHPs were subsequently identified as an "issue of concern" by the 4th International Conference on Chemicals Management (ICCM) in 2015 (Strategic Approach to International Chemicals Management [SAICM] 2015a). A strategy to address these pesticides was developed by FAO,

Table 3.2-7 Highly Hazardous Pesticides (HHPs) are defined by FAO and WHO as having one or more of the following characteristics (FAO and WHO 2016).

1.	Pesticide formulations that meet the criteria of classes Ia or Ib of the WHO Recommended Classification of Pesticides by Hazard;	or
2.	Pesticide active ingredients and their formulations that meet the criteria of carcinogenicity Categories 1A and 1B of the Globally Harmonized System of Classification and Labelling of Chemicals (GHS);	or
3.	Pesticide active ingredients and their formulations that meet the criteria of mutagenicity Categories 1A and 1B of the Globally Harmonized System of Classification and Labelling of Chemicals (GHS);	or
4.	Pesticide active ingredients and their formulations that meet the criteria of reproductive toxicity Categories 1A and 1B of the Globally Harmonized System of Classification and Labelling of Chemicals (GHS);	or
5.	Pesticide active ingredients listed by the Stockholm Convention in its Annexes A and B, and those meeting all the criteria in paragraph 1 of Annex D of the Convention;	or
6.	Pesticide active ingredients and formulations listed by the Rotterdam Convention in its Annex III;	or
7.	Pesticides listed under the Montreal Protocol;	or
8.	Pesticide active ingredients and formulations that have shown a high incidence of severe or irreversible adverse effects on human health or the environment.	

Table 3.2-8 History of relevant international policy for the identification and risk reduction of Highly Hazardous Pesticides (HHPs).

Year	Entity	Policy proposition	Reference
1975	WHO	Adoption of *The WHO Recommended Classification of Pesticides by Hazard*, which included the classes of extremely and highly hazardous pesticides	Copplestone (1988)
2003	4th Session of the Intergovernmental Forum on Chemical Safety (IFCS)	Recommended prohibiting or restricting the availability and use of acutely toxic pesticides … and/or those pesticides associated with frequent and severe poisoning incidents	IFCS (2003)
2006	131st Session of the FAO Council	Regarding FAO's contribution to SAICM, the Council suggested that FAO activities could include [pesticide] risk reduction, including the progressive ban on Highly Hazardous Pesticides (HHPs), promoting good agricultural practices …	FAO (2006b)
2007	1st FAO/WHO Joint Meeting on Pesticide Management (JMPM)	Recommended criteria for HHPs Recommended priority activities for risk reduction of HHPs	FAO/WHO (2007)
2008	2nd JMPM	Recommended minor amendments to the criteria for HHPs	FAO/WHO (2008)
2013	FAO and WHO	The International Code of Conduct on Pesticide Management includes a definition of HHPs Article 7.5 stipulates that "Prohibition of the importation, distribution, sale and purchase of highly hazardous pesticides may be considered if, based on risk assessment, risk mitigation measures or good marketing practices are insufficient to ensure that the product can be handled without unacceptable risk to humans and the environment."	FAO/WHO (2014)
2015	4th International Conference on Chemicals Management (ICCM)	ICCM Resolution IV/3 identifies HHPs as an "issue of concern" FAO, UNEP and WHO, in consultation with stakeholders, develop a Strategy to Address HHPs in the Context of SAICM	Strategic Approach to International Chemicals Management (SAICM) (2015a; 2015b)
2016	FAO and WHO	FAO and WHO publish *Guidelines on Highly Hazardous Pesticides*	FAO/WHO (2016)
2017	WHO	The 70th World Health Assembly approves the Chemicals Roadmap, which recognizes HHPs as a priority health issue and calls for actions focused on risk management WHO lists HHPs as one of 10 chemicals or groups of chemicals of major public health concern	WHO (2017; 2020a)
2020 - 2021	FAO	FAO, in collaboration with WHO and UNEP, elaborates an Action Plan on HHPs	

UNEP and WHO in consultation with stakeholders (SAICM 2015b). This strategy outlines ongoing work by intergovernmental organizations, identifies responsibilities and inputs from key stakeholders, and lists a number of gaps in HHP risk reduction. Based on an assessment carried out, the strategy identified the following main focus areas for concerted action:

- raising the awareness of different stakeholders about the risks of HHPs, and sharing information about risk reduction measures and viable alternatives;
- facilitating the identification of HHPs;

- capacity-building with regard to regulatory control;
- piloting and mainstreaming alternatives.

This was followed in 2016 by the publication by FAO and WHO of the Guidelines on Highly Hazardous Pesticides (FAO and WHO 2016). These guidelines aim to help national or regional pesticide regulators with limited resources design a process to address HHPs that follows three steps: identification, assessment and mitigation.

More recently, an action plan on HHPs is being elaborated by FAO in close collaboration with WHO and UNEP .

Contribution to pesticide risk reduction

HHP risk reduction is not a formal international mechanism, but rather a set of policy provisions and recommendations under various frameworks and codes. Therefore, a comprehensive review of its effectiveness has not been conducted so far.

The WHO Recommended Classification of Pesticides by Hazard (WHO 2019) has been widely used in Africa, Asia and Latin America both to classify pesticides (e.g., for labelling) and as a decision-making tool. Classes Ia and Ib (extremely and highly hazardous) pesticide products have been strictly regulated or are not allowed to be registered in many low and middle income countries.

Countries continue to take regulatory actions to restrict or cancel the registration of pesticides considered to pose unacceptable hazards and/or risk for human health or the environment. Such measures are often taken independently of the above mentioned policy recommendations concerning HHPs. However, on the basis of

Table 3.2-9 Examples of recent restrictions and bans in low and middle income countries of pesticides considered to pose unacceptable risk to the environment and/or human health.

Country	Year	No. of pesticides cancelled	No. of pesticides otherwise more strictly regulated
Mozambique	2014	61 products containing 31 a.i.'s	52 products containing 11 a.i.'s
Sri Lanka	2016	Registration cancelled of all pesticides containing carbaryl, carbofuran and chlorpyrifos	
Permanent Interstate Committee for Drought Control in the Sahel (CILSS)	2017	Registration cancelled of pesticides containing hexazinone and acetochlor	
Viet Nam	2017	Registration cancelled of plant protection products containing paraquat and 2,4-D	
Argentina	2018	Ban on all agricultural pesticides containing dichlorvos and trichlorfon	
Myanmar	2018	Products containing 15 a.i.'s	Products containing 4 a.i.'s
Malaysia	2019	All products containing paraquat	
Pakistan	2019	WHO class Ia and Ib pesticides will no longer be registered	
Viet Nam	2019	Registration cancelled of plant protection products containing chlorpyrifos-ethyl and fipronil	
India	2020	Prohibition to import, manufacture, sell, transport, distribute and use 27 pesticide active ingredients	
Peru	2020	Registration cancelled of pesticides containing methamidophos	
Turkey	2020	Withdrawal from the market of pesticides containing 16 pesticide active ingredients	

FAO and WHO guidance several countries (e.g., Botswana, Costa Rica, Malawi, Tanzania, Zambia and Zimbabwe) have identified HHPs still in use within their territories while others (e.g., Mozambique and Myanmar) have also taken concrete actions to reduce their risks (FAO 2016; ter Horst *et al*. 2018) (Table 3.2-9).

In 2015-16 CropLife International member companies conducted a portfolio review to assess their production of and trade in HHPs. The focus was on realistic conditions of use in low income countries. Of the approximately 6,400 crop protection products evaluated, 85 per cent were not HHPs; 10 per cent were HHPs which companies considered can be used safely and responsibly; 2.5 per cent required risk mitigation measures or were withdrawn from the market; and 2.5 per cent are under further evaluation (CPI 2020a). This corresponds reasonably well with experiences in countries in Africa and Asia where, on average, 5-10 per cent of registered pesticide products were identified as HHPs (FAO 2016; ter Horst *et al*. 2018).

It is not possible currently to determine the impact that global attention to HHPs has in reducing the risks associated with these pesticides. However, an increasing number of countries have recently banned or severely restricted the use of pesticides which in many circumstances meet HHPs criteria (Table 3.2-9).

For measures to reduce the environmental and human health risks posed by HHPs to be effective, they often need to be far-reaching (e.g., prohibition or severe restriction of the pesticide). This requires high-level national policy support. However, sound pest and pesticide management competes with many other national priorities, and policymakers may not see the urgency of reducing the risks posed by HHPs.

A further major obstacle to the phase-out or restriction of HHPs appears to be the apparent lack of cost-effective lower risk alternatives. Regulators are hesitant to take measures unless alternatives are available for farmers or other pesticide users. Experience shows that such alternatives are often available, in both high and lower income countries, that will not adversely affect production or farmer revenues. Examples are the bans of monocrotophos, methamidophos and endosulfan in Sri Lanka (Manuweera *et al*. 2008), coffee production without endosulfan in Latin America (FAO 2015), and bans of WHO class I pesticides in Bangladesh (Chowdhury *et al*. 2018).

In a recent review, Jepson *et al*. (2020) conclude that certain pesticides which pose a high risk in particular to the environment are not identified as HHPs by the current FAO and WHO criteria. They therefore call on these organizations to broaden the HHP definition to account for effects of concern with regard to important ecological services as well as to human health.

However, it has also been argued that care should be taken not to consider a large number of pesticides as HHPs, as this could hamper effective risk management by national regulators. Such a development could counteract the original objective of identifying a limited number of pesticides which require urgent risk mitigation, as they are very likely to cause undeniably unacceptable risks under conditions of use in lower income countries.

The Codex Alimentarius

Objectives and provisions

The Codex Alimentarius is a collection of internationally adopted food standards and related texts which aim to protect consumer health and promote fair practices in the food trade by setting international, science-based food safety and quality standards (FAO and WHO 2019c).

Codex standards are globally agreed recommendations for voluntary application by members, but in many cases they serve as a basis for national legislation. Furthermore, Codex standards are identified under the Word Trade Organization's (WTO) Agreement on the Application of Sanitary and Phytosanitary Measures (SPS) as benchmarks against which national measures and regulations are evaluated (FAO and WHO 2018) (Chapter 3.2.2).

The inaugural meeting of the Codex Alimentarius Commission was held in Rome in 1963.

A joint FAO/WHO Food Standards Programme was also established. Currently the Codex Alimentarius Commission has 189 members made up of 188 Member Countries and one Member Organization (the European Union). In addition, over 200 non-Member governments and intergovernmental and nongovernmental organizations are accredited observers of the Commission; consumers' organizations have been represented at its sessions since 1965 (Codex Alimentarius 2020).

The Codex operates under the umbrella of FAO and WHO, and the Secretariat of the Codex Alimentarius Commission is hosted at FAO headquarters in Rome. Codex committees prepare draft standards for submission to the Commission.

The Codex Alimentarius and pesticides

The Codex Committee on Pesticide Residues (CCPR) is responsible for establishing Codex Maximum Residue Limits (MRLs) for pesticide residues in specific food items or in groups of food or feed that are traded internationally.

Before a Codex MRL can be established, human health risk assessments must be conducted to ensure the food supply is safe. It is the responsibility of the Joint FAO/WHO Meeting on Pesticide Residues (JMPR) to review the appropriate toxicology and data obtained from pesticide residue trials which reflect approved pesticide use in accordance with "good agricultural practice", and to conduct dietary risk assessments. JMPR then recommends specific MRLs to the CCPR which, if considered acceptable, forwards them to the Commission for adoption (Codex Alimentarius 2020).

The Codex has established more than 5,000 pesticide MRLs for over 300 different active ingredients (FAO and WHO 2018).

Contribution to pesticide risk reduction

The globally harmonized Codex MRLs can be considered effective if a majority of countries use them. In recent years there have been concerns that an increasing number of countries are moving away from using the Codex pesticide MRLs and are establishing their own (Yeung *et al*. 2018). Indeed, major economies such as Australia and the United States (as well as the European Union) do set their own MRLs, which are regularly different from those of the Codex (Handford, Elliott and Campbell 2015).

Berry (2016) showed that many countries continue to make use of Codex MRLs and that, even in the case of countries with national standards, Codex MRLs continue to play a role. Ninety-six per cent of countries responded in a recent survey that they recognized, or partly recognized, the MRLs provided by the Codex; 29 per cent of countries reported having established national MRLs that differ from the Codex MRLs (FAO and WHO 2019c). Since most countries use Codex MRLs as a reference point, the Codex process for setting food safety standards and the resulting pesticide MRLs remain very relevant.

There are concerns about the capability of the Codex to harmonize MRLs on a global scale. It has been argued that developing countries, in particular, are adversely affected by the stringent MRLs set by certain countries/regions which create unnecessary trade barriers (Handford, Elliott and Campbell 2015; Yeung *et al*. 2018).

Frustration has also been expressed by Codex members and observers regarding the slow pace of establishing new MRLs by the CCPR and the difficulty of that process, leading to a limited increase in the number of evaluations and in the frequency of JMPR meetings. Nevertheless, it may be several years before Codex MRLs are established for new active ingredients.

The advantages of globally harmonized standard setting for pesticide residues are generally considered to outweigh the difficulties encountered. Science-based pesticide MRLs and associated dietary risk assessments, as established by the CCPR, provide a thorough basis for national food safety regulation, especially in countries with limited resources.

The Globally Harmonized System of Classification and Labelling of Chemicals (GHS)

Objectives and provisions

The main objective of the Globally Harmonized System of Classification and Labelling of Chemicals (GHS) is to ensure that information on the physical hazards and toxicity of chemicals is available in order to enhance the protection of human health and the environment during the handling, transport and use of these chemicals (United Nations Economic Council for Europe [UNECE] 2020a). The GHS also provides a basis for harmonization of rules and regulations on chemicals at national and international levels.

The GHS includes the following elements:

- harmonized criteria for classification of substances and mixtures according to their health, environmental and physical hazards; and
- harmonized hazard communication elements, including requirements for labelling and safety data sheets.
- The first edition of the GHS was adopted in December 2002. It has been updated every two years.

The GHS and pesticides

The International Code of Conduct on Pesticide Management (FAO and WHO 2014) and the FAO and WHO Guidelines on Good Labelling Practice for Pesticides (FAO and WHO 2015) recommend that the GHS classification and labelling provisions be applied to pesticides. Moreover, the GHS classification criteria for carcinogenicity, mutagenicity and reproductive toxicity (CMR) are among the criteria used to identify Highly Hazardous Pesticides (FAO and WHO 2016).

Contribution to pesticide risk reduction

The GHS website lists 72 countries that have implemented (part of) the GHS in national legislation or policy (UNECE 2020b). However, certain countries which are not listed there are applying GHS to the labelling of pesticides, e.g., through national guidelines.

A review by Persson *et al*. (2017) found that, as of 1 April 2017, 50 countries (26 per cent of UN Member States) had fully implemented the GHS in national legislation, 15 countries (8 per cent) had partially implemented it, and 128 countries (66 per cent) had not yet implemented it (these percentages were for the GHS in general, not specifically pesticides). This review noted that the African region was still mainly outside the GHS system, although the Southern African Development Community (SADC) had agreed to implement the GHS by the latest in 2020.

SAICM progress reports, on the other hand, indicate that countries reporting that they labelled pesticides in conformity with the GHS increased from 41 per cent in 2011 to 82 per cent in 2017 (SAICM 2011; SAICM 2019). A recent survey by FAO and WHO (FAO and WHO 2019a) found considerably lower adoption of GHS: 55 per cent of countries reported that pesticide labelling for agriculture was in line with the GHS, and 43 per cent reported that this was the case for public health pesticides.

These somewhat contradictory figures may be partly explained by different survey methods and data sources, but further clarification would be helpful given the role of pesticide labelling in communicating risks and good practices.

GHS criteria have also been used for decision-making for the authorization of pesticides. The European Union generally does not approve pesticide active substances that meet GHS criteria 1A and 1B for carcinogenicity, mutagenicity and reproductive toxicity (EU 2009). Similarly, Myanmar uses criteria for Highly Hazardous Pesticides, which include the GHS CMR criteria, as a decision-making element in its first screening of new pesticide active ingredients (ter Horst *et al*. 2018). This has been criticized by some as not being risk-based, and therefore less relevant for regulatory decision-making. However, the GHS does provide a sound international system to help decision-making in countries that have insufficient resources to conduct extensive local risk

assessments. It has not been assessed to what extent GHS criteria are used in decision-making for pesticide registration around the world.

Persson *et al*. (2017) assessed factors affecting the level of adoption of the GHS by different countries. They conclude that government effectiveness (i.e., the ability to formulate and introduce legislation) was the strongest predictor of GHS implementation, followed by political globalization and commitment to occupational safety. On the other hand, a country's GDP or general commitment to the sound management of chemicals (as measured through participation in international chemicals conventions) were not significantly associated with GHS adoption.

The Strategic Approach to International Chemicals Management (SAICM)

Objectives and provisions

The Strategic Approach to International Chemicals Management (SAICM) is a policy framework to promote chemical safety around the world (SAICM n.d.). Its objective has been to achieve sound management of chemicals throughout their life cycle, so that by the year 2020 they were produced and used in ways that minimized significant adverse impacts on the environment and human health.

Focal points from more than 175 governments are registered with SAICM, as are about 125 non-governmental organizations and 18 intergovernmental organizations. The implementation of SAICM is periodically reviewed by the International Conference on Chemicals Management (ICCM).

SAICM and pesticides

Pesticide risk reduction was proposed from the start as a possible work area in the SAICM Global Plan of Action (GPA) (SAICM n.d.). A large number of pesticide-related activities were identified under three topics:

- risk management and reduction of highly toxic pesticides and promotion of safer pest control measures;
- establishment of pesticide management programmes to regulate the availability, distribution and use of pesticides;
- reduction of health and environmental risks of pesticides.

In addition to work under the GPA, HHPs were identified as an "issue of concern" by SAICM's 4th International Conference on Chemicals Management (ICCM-4) in 2015 (SAICM 2015b). ICCM also identified two "emerging policy issues" which relate to pesticides: endocrine disrupting chemicals and nanotechnology.

Several groups of pesticides have been identified as likely to affect the endocrine system (Chapter 4.3.4). Therefore, risk assessment and reduction activities under this emerging issue may also impact on pesticide use. Nanomaterials are increasingly being used in pesticide formulations (Chapter 2.6.4).

SAICM established a Quick Start Programme (QSP) to support initial enabling capacity-building and implementation activities to meet the SAICM objectives. Additional funding for SAICM-related activities has come from a Global Environment Facility (GEF) project on Global Best Practices on Emerging Policy Issues of Concern, as well as from the UNEP Special Programme on Institutional Strengthening for the Chemicals Cluster (the Special Programme). Nevertheless, SAICM has remained chronically short of funding.

Contribution to pesticide risk reduction

The QSP consisted of a trust fund which supported 184 small-scale projects in 108 countries. Only 17 of these projects concerned agriculture and pesticides; however, some other projects (e.g., those concerned with implementation of the Basel, Rotterdam or Stockholm Conventions) are likely to have had an impact on pesticide management.

SAICM-related activities are currently being funded under the GEF Project and the Special Programme, but neither supports specific activities on pesticide risk reduction although broader

chemicals management initiatives may support pesticide risk reduction.

Progress in the implementation of SAICM was last assessed for the period 2014-2016 (SAICM 2019). The percentage of countries with risk management programmes for pesticides increased from 66 per cent in 2011 to 82 per cent in 2017 (SAICM 2011; SAICM 2019). Pesticides were considered among the top five highest priority risks in 2017. Activities concerning pesticides have usually shown a high response rate in SAICM surveys, with more than 77 per cent of respondents in 2017 reporting activities on pesticides under almost all the SAICM indicators (SAICM 2019).

Regarding HHPs, a few QSP projects have resulted in the identification of HHPs (Costa Rica) or even the banning/restriction of HPPs (Mozambique). SAICM has not established a multi-stakeholder platform to advance work on HHPs, as it has with some other Issues of Concern and Emerging Policy Issues (e.g., lead in paint). No coordinated action has been taken on HHPs through SAICM, but individual stakeholders have made some progress. A Global Plan of Action on HHPs is being elaborated (Chapter 3.2.3, section on HHPs). Slow progress in formal recognition of HHPs as an issue of concern has been a cause of frustration for several SAICM stakeholders (Nurick 2019).

Thus while the period during which SAICM has operated can be associated with certain positive developments regarding pesticide risk reduction, these results cannot be directly related to SAICM activities. This is, to a certain extent, inherent in the policy-oriented approach taken by SAICM, which makes it difficult to quantify the effectiveness of its contributions to pesticide risk reduction. In this respect the independent evaluator of SAICM (Nurick 2019) concluded that the indicators of progress on the GPA had critical limitations as an effective monitoring system to track SAICM's performance over time. He stressed the need for outcome and impact focused indicators to complement existing indicators of progress (i.e., results-based indicators to measure tangible reductions in the environmental and health impacts of chemicals use).

WHO Prequalification of Vector Control Products

Objectives and provisions

The WHO prequalification programme aims at ensuring that vector control insecticides are effective, safe and of good quality, in order to prevent transmission of vector-borne diseases (WHO 2020b). This programme performs the following functions:

- assess the efficacy, safety and quality of vector control products based on dossiers submitted by manufacturers and according to well-defined methods and procedures;
- conduct inspections of the corresponding manufacturing sites;
- build the capacity of national regulatory authorities and quality control laboratories;
- undertake post-qualification market surveillance.

The outputs of this process are:

- a list of prequalified vector control products supported by a positive public health evaluation from WHO;
- a list of prequalified manufacturing sites, that have been and will continue to be inspected by WHO.

WHO pre-qualification and pesticides

The WHO prequalification evaluations cover vector control products, primarily insecticide-treated nets, indoor residual sprays (IRS), space sprays and larvicides.

The WHO prequalification programme for vector control products is the only international assessment addressing the efficacy, quality, human health and environmental risks of a specific group of pesticides (i.e., vector control products) at the global level.

Contribution to pesticide risk reduction

Lists of prequalified vector control products were originally intended to be used by WHO and other multilateral agencies to guide their procurement decisions. However, these lists have progressively become an important tool for many countries and organizations having to purchase vector control products.

In 2018, 20 per cent of surveyed countries used the WHO prequalified list as the sole basis for registering a vector control product, while another 44 per cent used it as supportive information (FAO and WHO 2019a). Therefore, the WHO prequalification assessments have de facto become a global evaluation and authorization tool for vector control products in the majority of countries affected by vector-borne diseases.

3.2.4 Legally binding international instruments indirectly addressing pesticides

The Convention on Biological Diversity

The Convention on Biological Diversity (CBD) was adopted in 1992 and entered into force in late 1993 (Convention on Biological Diversity [CBD] 1992). As of March 2020 the CBD had 196 Parties (CBD 2020a).

Three further protocols adopted under the CBD address specific issues related to biodiversity:

- the Cartagena Protocol on Biosafety;
- the Nagoya-Kuala Lumpur Supplementary Protocol on Liability and Redress to the Cartagena Protocol on Biosafety;
- the Nagoya Protocol on Access and Benefit-sharing.

Objectives and provisions

The CBD has three main objectives:

- conservation of biological diversity;
- sustainable use of the components of biological diversity;
- fair and equitable sharing of the benefits arising out of the utilization of genetic resources.

The Convention attempts to achieve these objectives through a variety of mechanisms, including National Biodiversity Strategies and Action Plans, cooperation and partnerships with other conventions and organizations, certain financial mechanisms to support developing countries in implementing the Convention and a Clearing-House Mechanism for scientific and technical information.

During the last decade the Strategic Plan for Biodiversity 2011-2020 and its Aichi Biodiversity Targets, adopted under the Convention in 2010, have served as a guiding framework for implementation of the CBD. Governments are in the process of developing a post-2020 global biodiversity framework that will guide actions in the decades to come.

The CBD and pesticides

Biodiversity, its components, and many of the Aichi Biodiversity Targets may be impacted by pesticide use. Three targets have indicators that directly address the use of pesticides (Table 3.2-4)

A number of thematic programmes have been established under the Convention, one of which is Agricultural Biodiversity. Issues addressed in this programme are conservation, sustainable use of pollinators and soil biodiversity, among others (CBD 2020b). The recently published The State of the World's Biodiversity for Food and Agriculture (FAO 2019) recognizes that pesticides are among the direct drivers reducing both agricultural and general biodiversity. (The effects of pesticides on biodiversity are further discussed in Chapter 4.)

Contribution to pesticide risk reduction

Much of the work of the CBD deals with biodiversity assessments and inventories, compilation and dissemination of best practices for conservation of biodiversity or its sustainable use, and facilitating financial support to developing countries for the implementation of the Convention. Several of these activities can

Table 3.2-10 Aichi Biodiversity Targets that directly address the management and use of pesticides and selected recommended specific indicators (CBD 2016).

Aichi Biodiversity Target	Specific indicators relevant to pesticide management and use
Target 3	
By 2020, at the latest, incentives, including subsidies, harmful to biodiversity are eliminated, phased out or reformed in order to minimize or avoid negative impacts, and positive incentives for the conservation and sustainable use of biodiversity are developed and applied, consistent and in harmony with the Convention and other relevant international obligations, taking into account national socioeconomic conditions.	• Trends in potentially harmful elements of government support to agriculture (e.g. pesticide subsidies or tax exemptions) • Number of countries with national instruments on biodiversity relevant taxes, charges and fees (e.g. risk discriminating pesticide taxes)
Target 7	
By 2020 areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity.	• Areas of agricultural land under organic production (expected to reduce pesticide use and risks) • Areas of agricultural land under conservation agriculture (may increase herbicide use and risks) • Proportion of agricultural area under productive and sustainable agriculture (expected to reduce pesticide use and risks)
Target 8	
By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity.	• Trends in pesticide use • Mortality rate attributed to unintentional poisoning

be expected to have an indirect positive effect on pesticide risk reduction.

The Global Biodiversity Outlook (GBO) is the CBD's flagship publication. It is a periodic report that summarizes the latest data on the status and trends of biodiversity and draws conclusions relevant to further implementation of the Convention. The most recent version (Global Biodiversity Outlook 5) was launched in September 2020 (CBD 2020c).

Table 3.2-11 shows progress towards the three Aichi Biodiversity with the information provided in that report. The targets mentioned above insofar as they address pesticide management and use. It may be concluded that, by 2019-2020, generally only limited progress has been made towards achieving relevant indicators for these three targets. In the case of pesticide use and unintentional pesticide poisoning, trends are moving away from achievement of the target.

The International Plant Protection Convention

The International Plant Protection Convention (IPPC) was originally adopted in 1951. The currently applicable New Revised Text of the IPPC was adopted in 1997 and came into force in 2005 (International Plant Protection Convention [IPPC] 1997). As of late 2020, the IPPC had Contracting Parties (IPPC 2020).

Objectives and provisions

The IPPC aims to protect the world's plant resources from the spread and introduction of pests and to promote safe trade. It has introduced International Standards for Phytosanitary Measures (ISPMs) as the main tool to achieve its goals, making it the sole global standard-setting organization for plant health. The IPPC standards are one of three recognized by the World Trade Organization's (WTO) Sanitary and Phytosanitary Measures (SPS) Agreement (Chapter 3.2.2)

Table 3.2-11 Progress towards Aichi Biodiversity Indicators 3, 7 and 8 made by 2020 (with particular reference to pesticide risk reduction), based on information provided in the Global Biodiversity Outlook 5 (GBO-5) and the reviews conducted in this report.

Aichi specific indicators	General progress towards achieving the indicator by 2020, as stated in GBO-5[1]	Specific progress towards achieving the indicator with respect to pesticides[2]	Chapter in this report
Target 3			
Trends in potentially harmful elements of government support to agriculture (e.g. pesticide subsidies or tax exemptions)	Little overall progress during the past decade	Limited progress Direct pesticide subsidies have become rare, but tax exemptions are still quite common.	5.3
Number of countries with national instruments on biodiversity relevant taxes, charges and fees (e.g. risk discriminating pesticide taxes)	Some progress towards target but at an insufficient rate	Limited progress Only few countries have established pesticide taxes; however, discriminating taxes based on hazards/risks are common for those countries that do. Some countries facilitate the registration of biopesticides and low risk pesticides.	5.3
Target 7			
Areas of agricultural land under organic production (expected to reduce pesticide use and risks)	There has been a substantial expansion of efforts to promote sustainable agriculture. Progress towards target but at an insufficient rate.	Limited progress Increased, but still minor fraction of total cropland	2.4
Areas of agricultural land under conservation agriculture (may increase herbicide use and risks)		Not assessed in this report	--
Proportion of agricultural area under productive and sustainable agriculture (expected to reduce pesticide use and risks)		Not assessed in this report	--
Target 8			
Trends in pesticide use	Rate of use (per area) of pesticides has stabilized during this decade, globally and in most regions, but rates are higher than for the previous decade by about 14 per cent.	Moving away from target Increase both in global use and in use intensity	2.3
Mortality rate attributed to unintentional poisoning		Moving away from target While global mortality due to unintentional pesticide poisoning has decreased, all cases of unintentional occupational pesticide poisoning have likely significantly increased.	4

1 GBO-5 = Global Biodiversity Outlook 5 (CBD 2020c). Progress towards meeting the targets was assessed in general, rather than with a specific focus on pesticides
2 Specific progress based on data from this report, i.e. for the period 2019-2020

In addition to the establishment of standards, the IPPC focuses on information exchange on, for example, pest status, phytosanitary measures and regulations, as well as on capacity development by national plant protection organizations (NPPOs).

The IPPC and pesticides

IPPC standards indirectly influence pesticide use, especially where they establish principles for the protection of plants, guidelines for pest eradication and recommended phytosanitary treatments.

The IPPC Guidelines for the Export, Shipment, Import and Release of Biological Control Agents and Other Beneficial Organisms (ISPM 3) (Food and Agriculture Organization of the United Nations and International Plant Protection Convention 2017) address biological control agents capable of self-replication and include those packaged or formulated as commercial products. Certain biopesticides are regulated under these guidelines.

Contribution to pesticide risk reduction

Through its programmes for strengthening national plant protection organizations, the IPPC has contributed to sound pest and pesticide management in many countries. However, no specific information is available concerning the IPPC's impact on pesticide use or risk reduction.

The IPPC Guidelines (ISPM 3) are used in many countries. These guidelines indicate that the scope of this standard does not include issues related to registration of biopesticides. However, confusion exists in some countries regarding how the guidelines relate to national regulations and requirements for biopesticide registration. This may lead to unnecessary impediments to the authorization of biological control products.

The ILO Worst Forms of Child Labour Convention

The International Labour Organization (ILO) Convention concerning the Prohibition and Immediate Action for the Elimination of the Worst Forms of Child Labour (No. 182) was adopted in 1999 and entered into force in 2000. As of the end of 2020, it had been ratified by 187 countries (International Labour Organization [ILO] 2020a)

Objectives and provisions

The main objective of the Convention is to prohibit and eliminate of the worst forms of child labour. This comprises, among others, "work which, by its nature or the circumstances in which it is carried out, is likely to harm the health, safety or morals of children" (ILO 1999).

The Worst Forms of Child Labour Convention and pesticides

Handling and use of pesticides by children is likely to harm their health and should therefore be prohibited and eliminated.

Contribution to pesticide risk reduction

The most recent global estimates of child labour indicate an overall reduction of children's involvement in child labour from 16 per cent in 2000 to 9.6 per cent in 2016. The share of children doing hazardous work declined from 11.1 per cent in 2000 to 4.6 per cent in 2016 (ILO 2017a)

Agriculture is the most important sector for child labour by a considerable margin, accounting for 71 per cent of the total. There are no global statistics on the number children handling and applying pesticides, but it is known to be common in certain parts of the world including in some high income countries. For example, children applying pesticides in the production of cocoa, cotton and sugarcane is widespread (ILO 2007; ILO 2016; ILO 2017b) although not limited to these crops.

The ILO Occupational Safety and Health Convention

The ILO Convention concerning Occupational Safety and Health and the Working Environment (No. 155) was adopted in 1981 and entered into force in 1983 (ILO 1981). As of the end of 2020, it had been ratified by 69 countries (ILO 2020b).

Objectives and provisions

The main objective of this Convention is to prevent accidents and injury to health arising out of, linked with or occurring in the course of work. It does so by requiring from Parties that they implement and periodically review a coherent national policy on occupational safety, occupational health and the working environment.

The Occupational Safety and Health Convention and pesticides

The Convention applies to all branches of economic activity, including agriculture and others where pesticides may be handled and

applied. However, Parties are allowed to exclude to particular branches of economic activity from the Convention.

Contribution to pesticide risk reduction

No information is available regarding extent to which the Convention has led to national regulations on prevention of the occupational risks related to the use of pesticides.

3.2.5 Voluntary international instruments and mechanisms indirectly addressing pesticides

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES)

Objectives

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) was established in 2012 as an independent intergovernmental body to strengthen the science-policy interface for biodiversity and ecosystem services for the conservation and sustainable use of biodiversity, long-term human well-being and sustainable development (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services [IPBES] 2020). The IPBES can be seen as a follow-up to the Millennium Ecosystem Assessment conducted in the early 2000s (Millennium Ecosystem Assessment 2005).

IPBES is not a United Nations body. However, at the request of the IPBES Plenary and with the authorization of the UNEP Governing Council in 2013, the United Nations Environment Programme (UNEP) provides secretariat services to IPBES.

The work of IPBES can be broadly grouped in four complementary areas:

- Assessments: conducting assessments on specific themes, methodological issues, and at both the regional and global levels;
- Policy support: identifying policy-relevant tools and methodologies, facilitating their use, and catalyzing their further development;
- Building capacity and knowledge: identifying and meeting the priority capacity, knowledge and data needs of the member States, experts and stakeholders;
- Communications and outreach: ensuring the widest reach and impact of the work of IPBES.

IPBES and pesticides

IPBES has conducted various regional assessments on biodiversity and ecosystem services. The overall scope of these assessments has been to assess status and trends with regard to biodiversity and ecosystem services, the impact of biodiversity and ecosystem services on human well-being, and the effectiveness of responses. A global assessment report on biodiversity and ecosystem services was published in 2019 (IPBES 2019).

IPBES assessments address direct and indirect drivers of change in ecosystems and biodiversity. Both the impact pesticides have on biodiversity and ecosystem services, as well as policies that increase or decrease their use, are discussed in the assessments, though in fairly general terms.

IPBES has also produced an assessment report on pollinators, pollination and food production. It provides a detailed evaluation of existing knowledge and knowledge gaps about the role that pesticides play as a driver of change of pollinators, pollination networks and pollination (Chapter 4.3.3).

Contribution to pesticide risk reduction

IPBES assessments on the state of the science on various aspects of biodiversity are considered authoritative and provide valuable suggestions for the protection and restoration of biodiversity.

The impact of these reports on national policy has not (yet) been measured.

3.2.6 Discussion – International instruments and mechanisms

A considerable number of international instruments and mechanisms address the

management of pesticides. They either directly influence the production, trade and use of pesticides (e.g., the Rotterdam and Stockholm Conventions, the International Code of Conduct on Pesticide Management, the Codex Alimentarius, WTO agreements) or affect the use and risks of pesticides indirectly, generally by establishing requirements for national environmental, health or trade policies and legislation (e.g., the Convention on Biological Diversity, the ILO Occupational Safety and Health Convention), see Figure 3.2-1 above.

Most international instruments, both legally binding and voluntary ones, have substantial provisions and programmes aimed at promoting information exchange, conducting training and awareness building activities, providing guidance on best practices, and strengthening technical and administrative skills. These instruments can therefore be expected to have contributed to the enhancement of the national capacities for judicious pesticide management and knowledge about associated environmental and human health risks. However, improving knowledge, awareness and attitudes with regard to sound pest and pesticide management does not necessarily mean that adverse environmental and health impacts will be significantly reduced in a given country, region or situation (Chapters 2.7.20 and 2.7.21).

Four legally binding instruments (the Rotterdam, Stockholm, Montreal and Minamata Conventions) address individual pesticides. However, at the time of listing of those pesticides under the respective conventions the use of a considerable number of them had already been greatly reduced or discontinued. The prospects of new listings of pesticides of concern have been shown to face constraints due to both technical and political limitations. For other pesticides, such as methyl bromide (under the Montreal Protocol) and endosulfan (under the Rotterdam and Stockholm Conventions), significant reductions in use have been noted after they were listed in the conventions. The risks of some of the pesticides more recently included under the Rotterdam Convention (e.g., methamidophos and carbofuran) may have been reduced due to their listing in Annex III, but data to support these are scant.

Another impact that participation in environmental multilateral agreements may have is a strengthening of national legislation. There are indications this may be the case for pesticides. Brandi *et al*. (2019) found that both environmental international agreements and preferential trade agreements (PTAs) with environmental provisions resulted in significant changes in national environmental legislation referring to pesticides. The positive link was strongest in developing countries: they noted that although this apparent effect of treaties on domestic legislative change does not mean new regulations will fully implement a treaty, the legislation is more stringent than previous legislation, is de facto enforced, or leads to better environmental outcomes. The adoption by developing countries of environmental international agreements and PTAs with environmental provisions appears to stimulate change in, and likely strengthening of, environmental legislation.

Several non-binding international instruments which address pesticides, such as the Code of Conduct, the Codex Alimentarius and the GHS, appear to have improved the way pesticides are evaluated, managed and used. This has occurred through strengthening of legislation and registration of pesticides, harmonization of standards such as MRLs, or improved evaluation and hazard classification of pesticides. Furthermore, an international mechanism, SAICM, has set the stage to globally address issues of concern including risks posed by HHPs.

Owing to the nature of many legally binding and voluntary international instruments, their contribution to pesticide risk reduction cannot always be well evaluated. Appropriate indicators and procedures to allow such evaluations are not in place. An exception is the Global Monitoring Plan of the Stockholm Convention. Implementation of the International Code of Conduct of Conduct on Pesticide Management has been evaluated several times, but the results are not very specific and do often not allow specific needs and actions for improvement to be identified.

The absence of broader impact assessments makes it difficult to quantify the extent to which these international instruments contribute to

reducing the risks posed by pesticides (or by chemicals in general).

It should be emphasized that the conventions discussed above cover only a limited number of pesticides. Even if they significantly reduce the risks of the listed pesticides, they will not be adequate to effectively minimize the adverse environmental and health impacts of all pesticides in a comprehensive and sustainable manner. Furthermore, effective implementation of codes and other voluntary instruments faces operational challenges in addressing important aspects of the management of all pesticides and minimizing their adverse impacts.

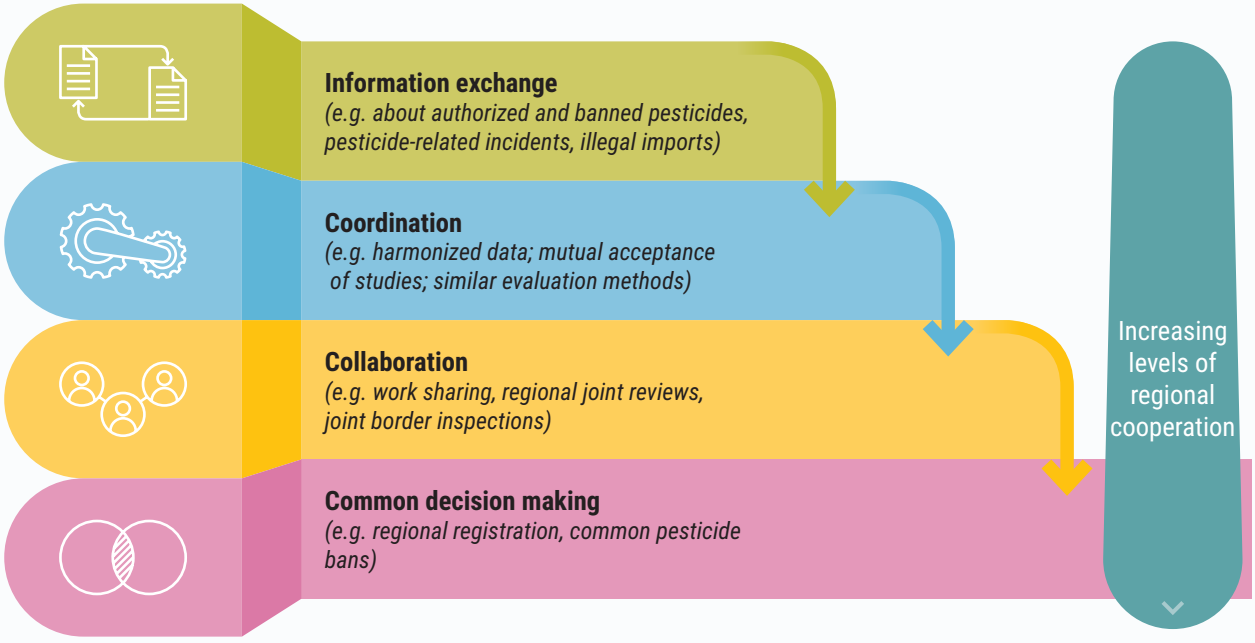
3.3 Regional collaboration on pesticide management

3.3.1 Introduction

The International Code of Conduct calls on governments to "promote the advantages of, and cooperate with other governments in, the establishment of harmonized (regionally or by groups of countries) pesticide registration requirements, procedures and evaluation criteria" (FAO and WHO 2014).

Regional collaboration on pesticide management can take different forms, ranging from simple information exchange mechanisms to more complex common decision-making systems (Figure 3.3-1). Regionalization of activities such as the evaluation, authorization, inspection and control of pesticides will often optimize the use of limited human and financial resources and strengthen the regulatory potential that comes with a larger geographical area. In addition, given the fact that borders are relatively porous in many parts of the world, regional collaboration can enhance the possibilities to control cross-border (illegal) trade in pesticides. Regional harmonization of registration and licensing requirements also facilitates the registration of new (lower risk) pesticide products that would not be economical to do in one country.

Figure 3.3-1 Cooperation on pesticide management and regulation can take many forms, with increasing intensity and complexity



Early examples of regional instruments for collaboration and harmonization of pesticide testing and regulation are the voluntary European and Mediterranean Plant Protection Organization (EPPO) Guidelines on the efficacy evaluation of plant protection products (first published in 1977), the South-East Asian pesticide bioefficacy protocols (published in 1990-92), the legally binding EU Council Directive 91/414/EEC concerning the placing of plant protection

Figure 3.3-2 Examples of legally binding regional legislation and agreements for the management of pesticides

Reglamentos técnicos centroamericanos

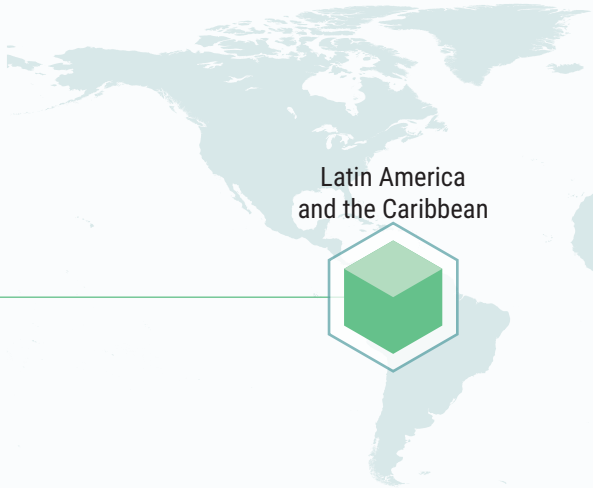
Year of adoption	various	*Number of countries*	5
Main objective	Harmonization of pesticide registration data requirements and labelling		

Manual Técnico Andino para el registro y control de plaguicidas químicos de uso agrícola (Comunidada Andina, Resolución 2075)

Year of adoption	1998 *(revised 2019)*	*Number of countries*	4
Main objective	Regional pesticide registration		

Common regulation for the registration of pesticides in CILSS member states

Year of adoption	1992 *(revised 1999)*	*Number of countries*	13
Main objective	Regional pesticide registration		



Bamako Convention on the ban of the import into Africa and the control of transboundary movement and management of hazardous wastes within Africa

Year of adoption	1998 *(entry into force)*	*Number of countries*	25
Main objective	Prohibit the import of all hazardous wastes into the African continent (including obsolete pesticides)		

Regulation on harmonization of the rules governing pesticide registration in ECOWAS region*

Year of adoption	2008	*Number of countries*	17
Main objective	Regional pesticide registration (not yet operational; will incorporate the CILSS system)		

Réglementation commune sur l'homologation des pesticides dans l'espace CEMAC

Year of adoption	2006	*Number of countries*	6
Main objective	Regional pesticide registration		

SADC Guidelines on pesticide management and risk reduction*

Year of adoption	2019	*Number of countries*	16
Main objective	Collaboration and harmonization on pesticide regulation and management (implementation just started)		

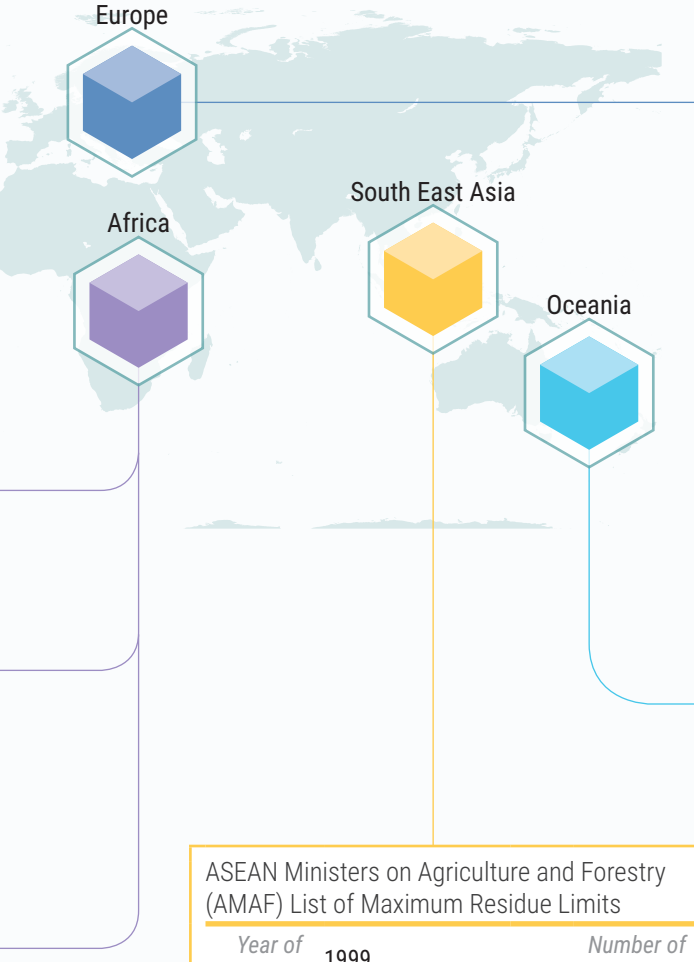
EAC regional pesticide registration guidelines

Year of adoption	2019	*Number of countries*	6
Main objective	Harmonization of pesticide registration data requirements and mutual acceptance of locally generated data		

ASEAN: Association of SouthEast Asian Nations; CEMAC: Communauté Economique et Monétaire de l'Afrique Centrale; CILSS: Permanent Interstate Committee for Drought Control in the Sahel; EAC: East African Community; ECOWAS: Economic Community of West African States; EU: European Union.

products on the market (adopted in 1991), and the Comité inter-État de lutte contre la sécheresse au Sahel (CILSS) Common Regulation for the registration of pesticides (adopted in 1992).

Many regional initiatives for pesticide management have since been initiated, ranging from harmonization of pesticide testing protocols and regional pesticide management action plans to regional pesticide registration. Several of these



EU Regulation 396/2005 on maximum residue levels of pesticides in or on food and feed of plant and animal origin

Year of adoption	2005	*Number of countries*	28
Main objective	Regional pesticide maximum residue limits		

EU Regulation 1107/2009 concerning the placing of plant protection products on the market

Year of adoption	2009 *(originally 1991)*	*Number of countries*	28
Main objective	Regional pesticide registration		

EU Directive 2009/128 establishing a framework for Community action to achieve the sustainable use of pesticides

Year of adoption	2009	*Number of countries*	28
Main objective	Pesticide use and monitoring		

EU Regulation 528/2012 concerning the making available on the market and use of biocidal products

Year of adoption	2019	*Number of countries*	28
Main objective	Regional biocides registration		

Waigani Convention to ban the importation into Forum island countries of hazardous and radioactive wastes and to control the transboundary movement of hazardous wastes within the South Pacific region

Year of adoption	2001 *(entry into force)*	*Number of countries*	12
Main objective	Ban the import and export of hazardous waste to and from the Convention Area		

ASEAN Ministers on Agriculture and Forestry (AMAF) List of Maximum Residue Limits

Year of adoption	1999	*Number of countries*	10
Main objective	Regionally harmonized pesticide maximum residue limits		

* Document not available online.

activities have been actively supported by FAO and WHO, especially in low and lower-middle income countries.

3.3.2 Legally binding instruments

Most legally binding regional instruments that address pesticides are operational in Africa and Europe (Figure 3.3-2). A large regional pesticide management instrument is represented by the EU systems for authorization of plant protection products and of biocides, in which pesticide active ingredients are evaluated and approved by all EU Member States. Subsequent authorization of pesticide products takes place at the national level, although attempts are made to rationalize this process through regional zoning (for plant protection products) or EU authorizations (for biocidal products).

Another long-standing regional pesticide registration scheme is that of the Permanent Inter-State Committee for Drought Control in the Sahel, or Comité Permanent Inter-Etats de Lutte contre la Sécheresse dans le Sahel (CILSS) in West Africa. Unlike the EU approach, all pesticide products rather than only active ingredients are registered by the Sahelian Pesticides Committee for all CILSS members. Originally covering nine countries, this system is now being extended to cover 17 countries in West Africa under the auspices of the Economic Community of West African States (ECOWAS).

So far, most regional agreements have focused on the authorization of pesticides. However, the EU framework to achieve sustainable use of pesticides also promotes the reduction of risks and impacts of pesticide use on the environment and human health, as well as the use of IPM and of non-chemical alternatives to pesticides. A similar broad approach to regional pesticide management has recently been adopted in the Southern African Development Community (SADC) region of southern Africa, but it is not yet fully operational.

3.3.3 Voluntary regional collaboration mechanisms

Several regional collaborative structures and mechanisms have been established which are aimed at optimizing or facilitating different aspects of pesticide management (Figure 3.3-3). In some cases regional institutions (e.g., regional plant protection organizations) may operate pesticide management activities as part of their broader mandate; in others dedicated structures have been established.

Many voluntary collaborative mechanisms function as platforms for information exchange among regulators in a given region. In some cases regional technical guidance documents and protocols are being developed.

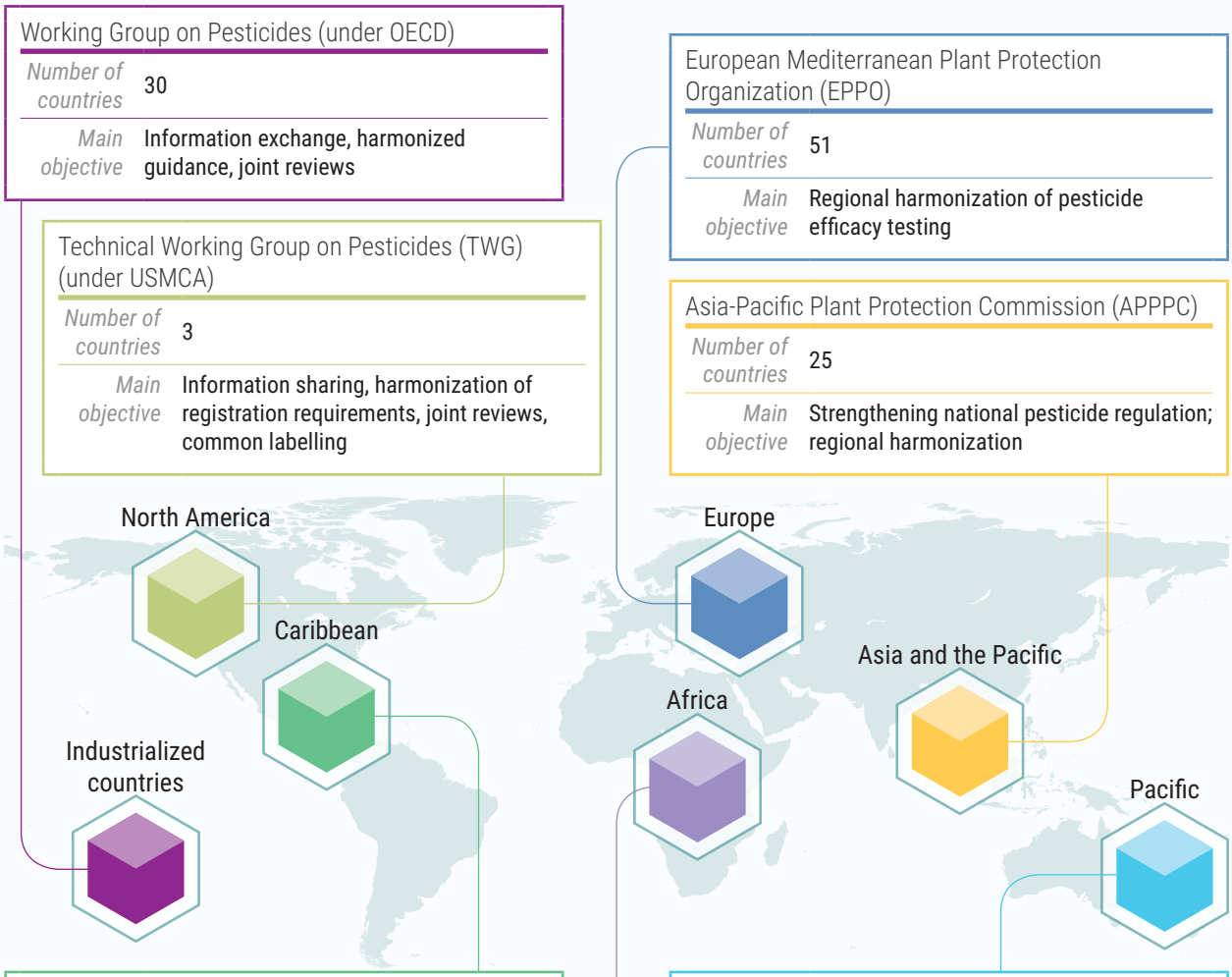
3.3.4 Discussion: Regional agreements and mechanisms

During the last decade there has been increasing interest in establishing or strengthening regional agreements and mechanisms on pesticide regulation and management. Recent initiatives include harmonization of data requirements and mutual acceptance of efficacy and residue data in East Africa, and the development of procedures for joint pesticide evaluations in the Pacific and the Caribbean, among others.

Regional cooperation is a potentially powerful approach for activities that can be conducted over larger geographical regions, such as efficacy and residue testing, or the evaluation, classification and authorization of pesticides, especially if resources are limited. However, regional collaboration has also proven to be effective in high income regions. It tends to be easier to implement in regions with similar administrative cultures, economic and agronomic conditions, and constraints.

The establishment or strengthening of regional collaboration is promoted by international organizations such as FAO. It may be the only effective way forward for resource-intensive aspects of pesticide management such as scientific evaluations, pesticide quality control and border inspections.

Figure 3.3-3 Examples of regional mechanisms for the management of pesticides



* Document not available online.

3.4 National policies and legislation on pesticides

3.4.1 Pesticide legislation

Legislation is intended to regulate the use and management of pesticides by defining the rights and obligations of the actors involved, and to establish measures designed to ensure the observance of these rights and obligations.

At the national level, pesticide legislation refers to legal instruments specifically designed to control pesticides. The term pesticide legislation may refer to a primary instrument, often a law, act or ordinance, as well as to a number of secondary or subsidiary legal instruments such as regulations, decrees, rules or notices (FAO and WHO 2020).

The International Code of Conduct on Pesticide Management prescribes that governments "should introduce the necessary policy and legislation for the regulation of pesticides, their marketing and use throughout their life cycle, and make provisions for its effective coordination and enforcement, including the establishment of appropriate educational, advisory, extension and health-care services" (FAO and WHO 2014). Specific guidance on pesticide legislation has been published under the Code of Conduct, which includes recommendations for countries to regulate pesticides taking into consideration all stages of the pesticide life cycle (FAO and WHO 2020) (Box 3.4-1).

In principle, pesticide legislation should cover all types of pesticides and all aspects of the pesticide life cycle (Figure 3.4-1). In many countries, parts of the pesticide life cycle are regulated by dedicated pesticide legislation. This often includes the authorization, importation, storage and transport,

Box 3.4-1 Elements of pesticide legislation.

Governments regulate pesticides for many reasons. The main objective is to protect human health and the environment from risks associated with pesticide use. This includes protection of pesticide users, consumers, crops, livestock, wildlife, water bodies and many others. Other important objectives include ensuring the effectiveness of pesticide products for their proposed use and safeguarding a fair market for manufacturers, importers and distributors of pesticide products.

Legislation is one of the tools that countries use to achieve these objectives, by regulating the manufacture, registration, importation, transport, storage, sale, use and disposal of pesticides (FAO/WHO 2020). The International Code of Conduct on Pesticide Management recommends governments to regulate all pesticides through all stages of their life cycle (Article 6.1). *The Guidelines on Pesticide Legislation* (FAO/WHO 2020) identify the key elements that should be taken into consideration in pesticide legislation.

A key characteristic of dedicated pesticide legislation is that a pesticide can generally not be imported, traded or used in a country unless it has been explicitly authorized under the law (positive list). This contrary to much other chemicals legislation which often allow the use of an individual chemical unless it is restricted or prohibited (negative list).

Essential elements of pesticide legislation are (based on FAO/WHO 2020):

Scope and definitions	▸ a well-defined scope that makes clear whether the legislation covers all categories of pesticides and all stages of the pesticide life cycle. If not, the law should explicitly identify the groups of pesticides covered
	▸ definitions that are clear, not ambiguous, and aligned with the international reference definitions (e.g. Code of Conduct, Chemical Conventions)
Administration	▸ a competent authority designated to coordinate the implementation of the law, having the necessary powers, including the power to inspect, charge fees and elaborate regulations
	▸ a mandatory registration system for pesticides
	▸ a pesticide registration board composed of different public institutions representing the key regulatory interests, with safeguards in place to prevent conflicts of interest on the part of its members

Registration	key steps in the application procedure for pesticide registration, including information and data requirements to be included in the application and main criteria for decision making on registration
	a provision that a registration can be reviewed at any time when new information has become available and that a negative outcome of such a review can lead to cancellation of the registration
	provisions ensuring confidentiality and protection of intellectual property rights
Import and export	prohibition of the import of pesticides that have not been registered and specification of any exceptions to this provision
	a licensing and/or permit system for importation of pesticides
Licensing	specification of which pesticide-related activities (manufacture, sale, transportation, import, special applications) are allowed only by operators holding a valid license
	establishment of the necessary licensing schemes and designation of the responsible authorities for their implementation
	stipulation of the obligations of license holders and the consequences of noncompliance, while ensuring that licensing schemes are backed up with inspections
Packaging and labelling	requirements for the packaging and labelling of pesticides
Use	prohibition of the use of pesticides for a purpose or in a manner other than that prescribed on the label
	requirements for employers to implement necessary measures to protect worker health and to prevent use by children and other vulnerable groups
	prescription of the use of PPE, proper application equipment, responsible cleaning of application equipment and safe disposal of empty containers to protect users, the public and the environment
Advertising	specific requirements for pesticide advertising
Storage, transport and disposal	requirements for and designation of the authority responsible for overseeing storage, transport and disposal of pesticides
Information collection, monitoring and incident reporting	designation of the powers and responsibilities of the responsible body(ies) for information collection and monitoring with respect to pesticides, including the ability to impose reporting requirements on manufacturers, importers, distributors and sellers of pesticides
	mechanism for the reporting of pesticide-related incidents
Inspection	designation of national authority (or authorities) responsible for inspection
	definition the powers of inspectors and ensure that these are adequate to enable the inspectors to fulfil their duties
Offences and penalties	establishment of the offences under the law and outline of the applicable penalties
	rights and appeals related to enforcement procedures

distribution and sales, licensing, labelling and use of pesticides. Other parts of the pesticide life cycle, such as manufacturing and formulation, aspects to which environmental and health standards are applicable, or disposal, will often be regulated by other legislation.

The regulatory framework for control of pesticides therefore encompasses a much broader set of legislation than that which only directly addresses pesticides. It may, for instance, include legislation on environmental protection, public and occupational health, food safety, water, wildlife, plant protection and general chemicals management. The national regulatory framework also includes obligations under international instruments (Chapter 3.2).

Most countries have some form of dedicated pesticide legislation in place

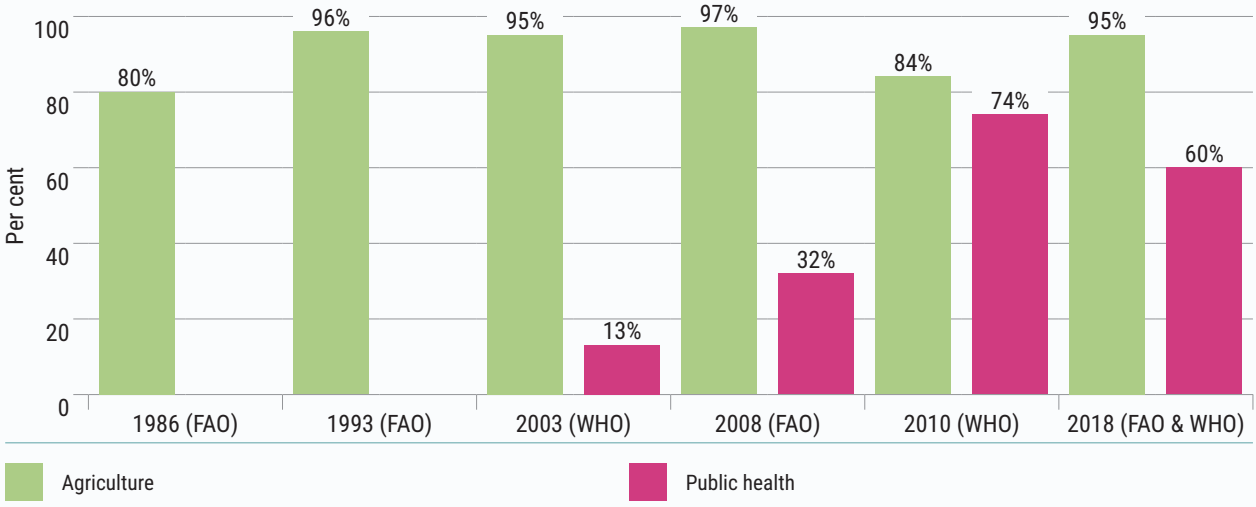
Figure 3.4-1 Elements of the pesticide life cycle and aspects in each step that are often regulated

(Figure 3.4-2). Traditionally, pesticide legislation covers agricultural pesticides. Shortly after the adoption of the first version of the International Code of Conduct on the Distribution and Use of Pesticides, in 1986, about 80 per cent of countries regulated at least agricultural pesticides. This increased to about 95 per cent in 1993 and has remained at that level ever since.

The almost complete global coverage of (agricultural) pesticide legislation provides an excellent legal basis to ensure judicious pesticide management. Public health pesticides and vector control pesticides, on the other hand, were covered by legislation in only 13 per cent of countries in 2003, although this steadily increased in the following decade to 60-70 per cent of countries (Figure 3.4-2). Still, according to the most recent global survey, 40 per cent of countries reported that the distribution and use of public health pesticides are not adequately regulated (FAO and WHO 2019a).

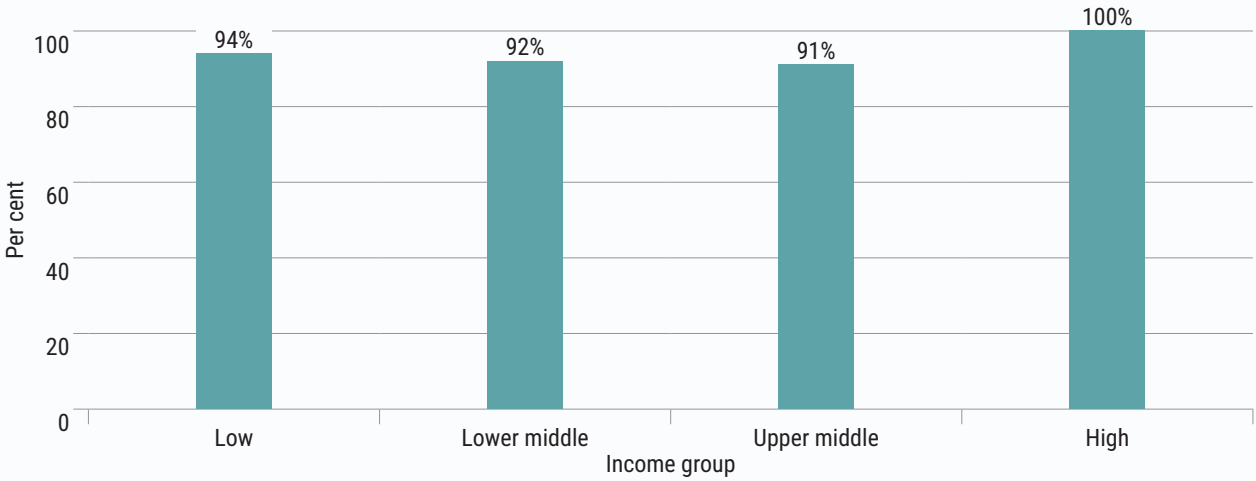
While all high income countries have dedicated pesticide legislation in place, an average of around 92 per cent of low, lower-middle and upper-middle

Figure 3.4-2 **Percentage of countries reporting that they have dedicated legislation for (primarily) agricultural pesticides, or for public health pesticides, based on surveys conducted during the last 30 years. The apparent reduction in 2010 is likely due to a different target being addressed by the survey: ministries responsible for public health vs. ministries responsible for agriculture.**



Sources: 1986 and 1993 = FAO (1996); 2003 = WHO (2004); 2008 = FAO (2010); 2010 = Matthews *et al.* (2011); 2018 = FAO/WHO (2019c).

Figure 3.4-3 **There is little difference in the percentage of countries that have pesticide legislation in place in low, lower-middle and upper-middle income groups.** FAO and WHO (2019a) (data re-analysed).



income countries regulate (some groups of) pesticides (Figure 3.4-3). The small overall difference in percentages in these income groups indicates that national income does not greatly influence the establishment of pesticide legislation.

Although many countries regulate pesticides, legislation does not necessarily cover all elements of the pesticide life cycle. Authorization (or registration) required in order to place pesticide products on the market is included in most pesticide legislation (Table 3.4-1). However, this is less true for other elements of the life cycle such as transport, storage, distribution and use, particularly in the case of public health pesticides. About 15-40 and 40-50 per cent of countries, respectively, responded that they do not adequately regulate these elements of the life cycle of agricultural pesticides and public health pesticides. Recycling and disposal of empty pesticide containers and unused pesticides is regulated in many countries through hazardous waste legislation rather than through pesticide laws. Empty pesticide containers remain an

Table 3.4-1 **Percentage of countries reporting that they regulate specific elements of the pesticide life cycle.** FAO and WHO (2019a).

Life cycle element	Percentage of countries regulating each element (n=59)	
	For agricultural pesticides	For public health pesticides
Manufacturing and formulation	69%	
Authorization (registration)	95%	60%
Importation	n.a.	n.a.
Storage and transport		
Storage	83%	58%
Transport	74%	47%
Distribution and use		
Pesticide retail sales	79%	47%
Online pesticide sales	24%	
Recycling and disposal		
Empty containers	62%	42%
Obsolete pesticides	69%	51%

n.a. = data not available. However, in countries that do not locally manufacture or formulate pesticides, but only import them, registration legislation automatically covers importation.

Table 3.4-2 **Responsible government authority which issues registrations for several types of pesticides. Shown is the percentage of countries answering each question positively, and the number of responding countries (n). Some countries reported there was more than one responsible authority.** FAO and WHO (2019a).

Pesticide types or products	Government ministry or agency					n
	Agriculture	Health	Environment	Other agency	Not regulated	
a. Agricultural pesticides	56%	9%	6%	25%	3%	64
b. Vector control pesticides	28%	40%	3%	23%	5%	60
c. Professional public health pest control products	26%	34%	5%	25%	10%	61
d. Insecticides directly applied on humans	18%	41%	2%	28%	11%	61
e. Household pest control products	24%	34%	3%	27%	12%	59

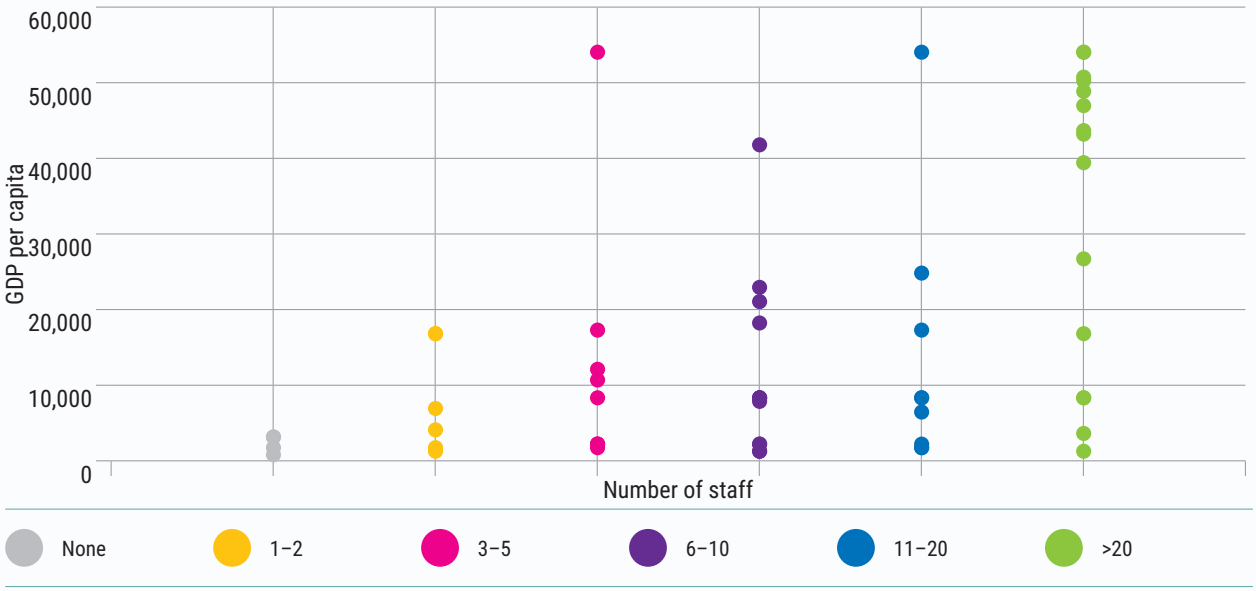
environmental and health problem in many low and middle income countries.

The presence or absence of pesticide legislation does not indicate its quality (i.e., the degree to which legislation allows sound management of pesticides in a country, and the possibilities it gives governments to limit dangerous practices and misuse). No global reviews are available assessing the content and quality of pesticide legislation, nor is there any systematic analysis of legal gaps. International guidance on the elaboration and content of pesticide legislation is however, available with which to assess the comprehensiveness of national pesticide legislation (FAO and WHO 2020)

3.4.2 Pesticide registration

Pesticide registration is the process whereby a responsible national government or regional authority approves the sale and use of a pesticide following the evaluation of scientific data aimed at demonstrating that the product is effective

Figure 3.4-4 Staff capacity for pesticide registration increases with countries' income levels. GDP is per capita purchasing power parity (PPP); 2017 international dollars. FAO and WHO (2019a) (data re-analysed).



for its intended purposes and does not pose an unacceptable risk to the environment, or to human or animal health, under the conditions of use in the country or region (FAO and WHO 2014).

Only 3-12 per cent of countries globally have indicated that they do not register (certain types) of pesticides; all other countries have systems in place to evaluate and authorize the use of pesticides (FAO and WHO 2019a). In many countries more than one government authority is in charge of pesticide registration. The ministry responsible for agriculture generally conducts the registration of plant protection products, while the ministry responsible for health registers public health and domestic pesticides (Table 3.4-2). This does not mean that decisions about the authorization of pesticides is made by only one sector ministry. Many countries have established intersectoral pesticide registration boards or committees that assess registration applications and either decide, or advise on, decisions to authorize the use of a pesticide.

In only one-quarter of countries (mainly in Europe, North America, and parts of Asia and the Pacific) do specialized pesticide registration agencies exist (FAO and WHO 2019a).

The evaluation of pesticides requires specialized staff in various scientific fields, including biology, chemistry, toxicology and environmental sciences. However, many countries do not have many staff dedicated to this task. Low and lower-middle income countries often only employ up to five staff who are involved in pesticide registration; registration authorities with larger pesticide registration bodies tend to be found in richer countries (Figure 3.4-4).

3.4.3 Compliance monitoring and enforcement

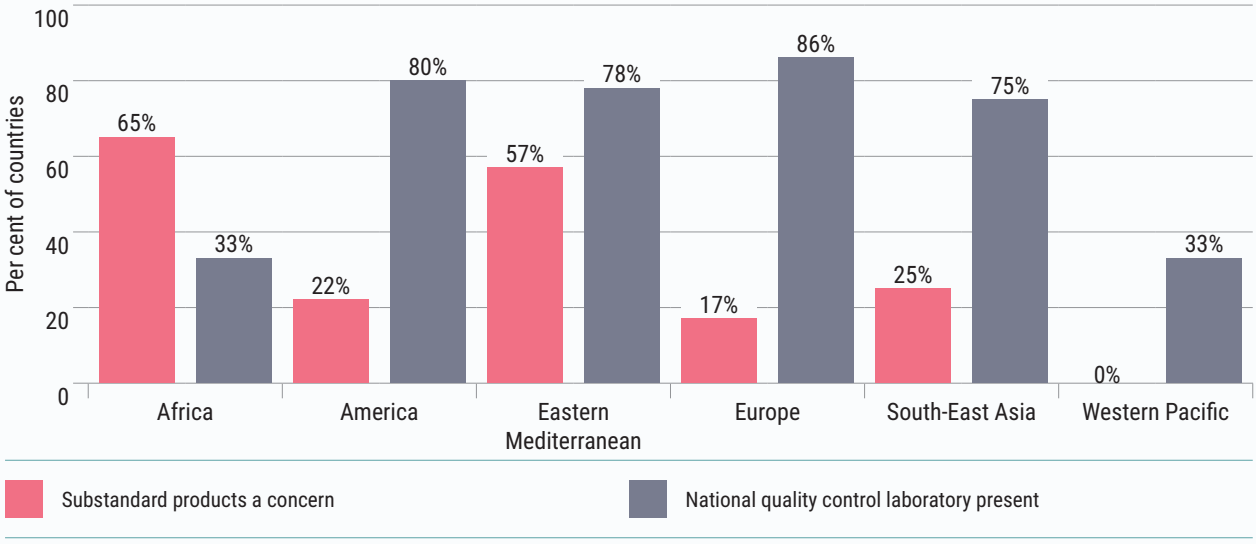
The existence of pesticide legislation, even if this legislation is well elaborated, is not enough to ensure that it will be respected. Legislation without effective compliance monitoring and enforcement tends to be a "paper tiger". On average, only 33-47 per cent of countries have indicated that they are able to routinely monitor the implementation of legislation, with countries in Africa, the Eastern Mediterranean and South Asia well below this average (FAO and WHO 2019a).

Lack of post-registration compliance and enforcement actions at the national level, resulting from a lack of government capacity, is well known to be a fundamental problem in pesticide management (FAO and

Table 3.4-3 **Some factors that could lead to non-compliance with pesticide legislation.** Adapted from OECD (2012) and UNEP (2019b).

By the regulated community	By regulators
Lack of knowledge or understanding of what is required for compliance	Lack of human and/or financial capacity to conduct inspections
High cost of compliance	Inadequate legal power for enforcement
Low likelihood of detection of violation or of prosecution	Lack of collaboration with other enforcement institutions (e.g. for fear of giving up power of control)
Low penalties, which can be internalized as a cost of doing business	Lack of clear responsibilities
Absence of a culture of compliance	High level of risk tolerance

Figure 3.4-5 **The percentage of countries in which substandard pesticide products are a concern is especially high in the African and Eastern Mediterranean regions. Laboratory capacity for quality control is low in the African and the Western Pacific regions.** FAO and WHO (2019a).



WHO 2019d). This problem is not limited to pesticide management. Inadequate compliance and enforcement is also a serious constraint with respect to most environmental legislation, even in countries with adequate resources (McClary and Goldstein 2017; UNEP 2019b).

There may be many reasons that lead to non-compliance, both for the regulated community and for governments (Table 3.4-3). They range from lack of knowledge of what is required for compliance to penalties that are too low to deter violations. Furthermore, if regulatory institutions are unable to effectively carry out inspections, and to prosecute and adjudicate violations of pesticide legislation, the regulated community may reasonably believe that violations will go punished (UNEP 2019b).

An important element of pesticide inspections is quality control of pesticide products at importation and sales. Such quality control should be conducted by dedicated laboratories that are equipped and experienced for the task. Fifty-nine per cent of responding countries reported having access to a national pesticide quality control (testing) laboratory (Figure 3.4-5). However, in the African and Western Pacific

regions only 33 per cent of countries had such laboratories (FAO and WHO 2019a). This is of particular relevance since, in 40 per cent of countries globally, substandard or counterfeit products are considered of "major or moderate" concern, defined as these products being readily available to the general public. The African and Eastern Mediterranean regions are well above this average (FAO and WHO 2019a) (see also Chapter 2.5).

Compliance, implementation and enforcement costs and modalities should always guide the design of regulatory frameworks to ensure successful performance of such frameworks in meeting regulatory objectives (FAO and UNEP 2020).

3.4.4 Other legislation relevant to pesticides

Because pesticide management is cross-sectoral, the way pesticides are authorized, managed and used is not only influenced by dedicated pesticide legislation, but also by laws, regulations and standards elaborated in other sectors.

Legislation on environmental protection, public health, occupational health, water, food safety, plant protection, general chemicals management, transportation and disposal of hazardous substances may be relevant (FAO and WHO 2020). For instance, environmental legislation may include requirements for the siting and construction of pesticide formulation plants or sales outlets, but it may also set surface water standards that need to be taken into account when a pesticide is registered; under food safety legislation pesticide maximum residue limits may be defined, which in turn influences both the authorization of a pesticide and good plant protection practices that should be followed by farmers; and occupational health legislation may set requirements for the protection of workers who apply pesticides or harvest crops. However, national standards (e.g., for pesticide labelling, packaging or PPE) may directly affect pesticide sales and use.

To avoid either potential legal conflicts or gaps in pesticide legislation, it is therefore important that the lead ministry responsible for the pesticide law coordinates effectively with other sector ministries. Multi-sector coordination and cooperation involves the (horizontal linkages) of, for example, health, agriculture, the environment and trade, as well as the (vertical) competences of various levels of administration (FAO and UNEP 2020). In many countries elaborating and operating comprehensive, well-coordinated legislation which effectively addresses all aspects of the pesticide life cycle has proved challenging.

3.4.5 Pest and pesticide management policy

Legislation is intended to regulate the trade, use and management of pesticides. Policy, on the other hand, is a set of principles that guide decision-making. A pest and/or pesticide management policy defines desired changes and is made in response to recognized problems or constraints, or to prevent problems arising in the future (WHO 2011b).

Stand-alone national pesticide management policies are developed and adopted to define how pesticides should be regulated, used and managed in a country. Pest management policies are broader than legislation, as they define how pests or disease vectors will be prevented, managed and controlled. Pest management policies will include pesticides, but also other approaches to pest management. It is important to note that countries without stand-alone pest management policies or strategies, elements of such policies may be incorporated into national legislation, regulations, or policies on related topics.

A recent global survey found that approximately 60 per cent of responding countries have adopted a national IPM policy, while about 50 per cent have a national IVM policy (FAO and WHO 2019a). These policies generally identify IPM/IVM as the preferred pest or vector management approach and define strategies and measures to promote this approach. WHO has developed guidance for public health pesticide management policy development (WHO 2011b).

During the last two decades several European countries have adopted pesticide use or risk reduction policies, recent examples of which are shown in Table 3.4-4. By 2014 EU Member States

Table 3.4-4 Examples of national pesticide management policies and strategies, and achievements insofar as these have been established.

Country	Year	Title	Targets	Results	Reference
China	2015	Action to achieve zero growth of pesticide use by 2020	Keep pesticide use per unit of land area below the average level in the years 2012 to 2014 Increase the use of biocontrol and physical pest control on main crops by 10 per cent compared to 2014	Not yet known	Jin and Zhou (2018)
The Netherlands	2013	Healthy growth, healthy harvest – Second policy for sustainable crop protection 2013-2023	Pesticide use: no target Exceedance of surface water quality standards: 50 per cent reduction by 2018; no exceedance by 2023 IPM: by 2014 all agricultural producers to apply IPM	By 2018: Use of chemical pesticides hardly decreased; fraction of low-risk pesticides remained very low, at 0.1 per cent of total use Fewer exceedances of surface water quality standards (30 per cent reduction for chronic exposure and 50 per cent for acute) Fewer exceedances of MRLs and dietary risk standards in food; target achieved 50 per cent of producers applying IPM	Ministerie van Economische Zaken (MEZ) (2013); Planbureau voor de Leefomgeving (PBL) (2019)
France	2015 2018	Ecophyto II - Ecophyto II+	25 per cent reduction in use of plant protection products by 2020 and 50 per cent reduction by 2025, compared with 2015	By 2018 an increase in use was observed, leading to Ecophyto II+	Government of France (2018)
Denmark	2017	Pesticide strategy 2017-2021	Pesticide Load Indicator (PLI, the Danish pesticide risk indicator) to be at most 1.96 by 2021	PLI reached 1.69 in 2017, an almost 50 per cent reduction compared with 2010-2011	Danish Environmental Protection Agency (EPA) (2017; 2019)
European Union	2009	Framework for Community Action to Achieve the Sustainable Use of Pesticides	All Member States should adopt and update National Action Plans (NAPs) for pesticide risk and impact reduction. All professional pesticides users should be trained and certified. Pesticides should only be sold to professionals by certified persons. IPM and organic farming shall be promoted under professional users	By 2020: 75 per cent of Member States reviewed/updated their NAP Majority of NAPs fail to identify high-level outcome-based target 20 per cent reduction in the EU harmonized risk indicator since 2017 50 per cent increase in risk from emergency authorizations Assessment of implementation of IPM is insufficient	EU (2009a; 2009b)
European Union	2020	Farm to Fork Strategy	Reduction of overall use and risk of chemical pesticides by 50 per cent, and those of more hazardous pesticides by 50 per cent, by 2030	Not yet known	European Commission (EC) (2020b)

were obliged to elaborate National Action Plans (NAPs) to achieve sustainable use of pesticides as a way to implement the Framework to Achieve Sustainable Use of Pesticides (Chapter 3.3). By 2016 all of the (then 28) Member States had drawn up NAPs, although not all of these countries had kept the NAPs up-to-date. Twenty countries had established pesticide risk reduction objectives and nine used defined pesticide use reduction objectives; some had a combination of both (EC 2017b; EC 2020a).

Although IPM is a cornerstone of the Framework, by 2016 countries had not yet set clear criteria to ensure that the principles of IPM were implemented, nor did they systematically verify implementation of IPM by growers. On the other hand, all Member States had established training and certification systems for pesticide users. It was reported that almost 4 million professional operators had been trained. A range of good practices to protect the aquatic environment from pesticides, reduce pesticide use in public spaces, and promote judicious handling and storage of pesticides were also implemented (EC 2017b).

While the EU Sustainable Use Framework did not establish specific pesticide risk reduction targets, the recently published Farm to Fork Strategy aims to reduce the overall use and risks posed by chemical pesticides by 50 per cent by 2030 (EC 2020b).

Stand-alone pesticide management policies are much less common in other parts of the world. Examples are the Pesticide Management Policy for South Africa (Department of Agriculture, Forestry and Fisheries of South Africa 2010) and the policy of zero growth in pesticide use in China (Jin and Zhou 2018). In other countries pesticide management policy elements may be incorporated in pesticide legislation. No global review of the content and implementation of pest or pesticide management policies is currently available.

Experience with pesticide risk/use reduction policies in Europe demonstrates that meeting ambitious pesticide use or risk reduction targets is not straightforward. Denmark achieved its target of 50 per cent reduction in pesticide risks, measured using a national pesticide risk indicator; the Netherlands only partly met its risk reduction targets; while France had to update and strengthen Ecophyto, its national pest and pesticide management policy, due to a lack of progress (Lee, den Uyl and Runhaar 2019). In this respect, Lee, den Uyl and Runhaar (2019) evaluated the effectiveness of public and private policy instruments in European or other high income countries which aimed to reduce pesticide use by farmers. These included regulatory, economic and informative policy instruments. They found that not one individual instrument is guaranteed to reduce pesticide use. Combinations of two, or better three, instruments were most effective in achieving pesticide use reductions.

The use of risk indicators to measure policy progress has become more prominent as they better reflect the potential adverse effects of pesticide use than volumes of pesticide consumption or sales. Risk indicators tend to combine a measure of pesticide exposure with the toxicity/hazard of the pesticides used. An example is the Pesticide Load Indicator (PLI) in Denmark (Kudsk *et al*. 2018). Another approach was taken in harmonized risk indicators established to monitor implementation of the EU Framework to achieve the sustainable use of pesticides, which are based on authorization categories of active substances in the EU (EU 2019).

3.4.6 Discussion – National legislation and policies

Most countries in the world have adopted dedicated pesticide legislation, which provides a national legal basis to manage the trade in and use of pesticides. However, not all aspects of the pesticide life cycle may be effectively covered by national legislation. In other cases, different parts of the pesticide life cycle may be shared across different pieces of legislation implemented by different government entities. While this does not need to pose problems in itself, it does require effective horizontal and vertical coordination within the government to ensure duplications and gaps are minimized. Legislation of domestic and public health pesticides is still inadequate in a large faction of countries. International guidance is available under the International Code of Conduct

on Pesticide Management for strengthening national pesticide legislation.

The majority of countries have put in place a pesticide registration system to evaluate and authorize the use of pesticide products. Especially in low and lower-middle income countries, however, human resources for sound evaluation of pesticides are very limited while the assessment of efficacy, hazards and risk is becoming increasingly complex (see Chapter 4).

Government inspection and control of pesticide-related activities are essential for effective implementation of national pesticide legislation. However, the capacity for enforcement is generally considered to be weak in many countries, often due to limited resources. As a result, illegal pesticides are sold and used, the quality of products on the market is not verified, pesticide sales and application are conducted by insufficiently trained staff, pesticides are disposed of in ways that are harmful to the environment, among others. Strengthening national capacity for inspection and control is therefore a high priority, especially in low and middle income countries.

Globally, the adoption of specific pesticide management policies has been rare so far. More policy instruments, based on regulatory, economic and informative approaches, will generally need to be implemented simultaneously, to successfully achieve pesticide risk reduction (Lee, den Uyl and Runhaar 2019). The elaboration of a comprehensive national pest/vector and pesticide management policy, adopted by all main stakeholders, would be an important first step in achieving that goal.

3.5 Corporate stewardship

3.5.1 Introduction

Within the framework of pesticide management, product stewardship has been defined as "responsible and ethical management of a pesticide product from its discovery through to its ultimate use and beyond". This definition in the International Code of Conduct on Pesticide Management (FAO and WHO 2014) explicitly recognizes that product stewardship is needed throughout the pesticide's life cycle.

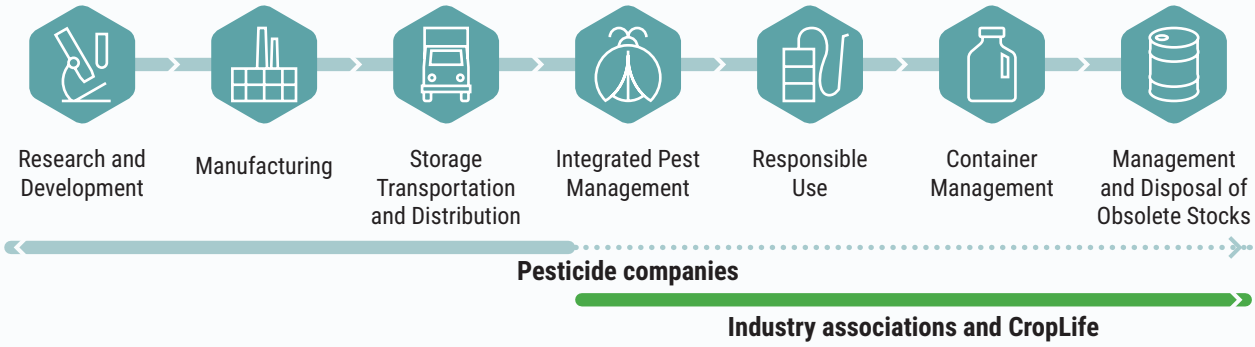
The terms product stewardship and extended producer responsibility (EPR) have been used interchangeably (Tasaki, Tojo and Lindhqvist 2018), while some consider EPR as a mandatory type of product stewardship (Product Stewardship Institute 2020). The OECD defines extended producer responsibility (EPR) as "an environmental policy approach in which a producer's responsibility for a product is extended to the post-consumer stage of a product's life cycle" (OECD 2016). However, OECD guidance on EPR primarily focuses on waste management.

Pesticide manufacturers, distributors and retailers have a primary responsibility to minimize adverse impacts. In this respect the Code of Conduct (FAO and WHO 2014) stipulates that pesticide industry and traders should:

- be capable of providing effective technical support, backed up by full product stewardship to end user level, including advice on and implementation of mechanisms for the effective management of unused and obsolete pesticides and empty pesticide containers;
- retain an active interest in following their products through the entire life cycle, keeping track of major uses and the occurrence of any problems arising from the use of their products, as a basis for determining the need for changes in labelling, directions for use, packaging, formulation or product availability.

In addition to the pesticide industry, pesticide product stewardship can be carried out by the food and commodity industries, particularly during the use of the products, as well as by major pesticide users.

Figure 3.5-1 **Key elements of crop protection stewardship as promoted by CropLife International ("CropLife"). Individual companies focus on the first part of the stewardship life cycle, while industry associations and CropLife International are mainly concerned with the second part.** CLI (2020c).



3.5.2 The pesticide industry

Stewardship is practised by individual pesticide companies and by industry associations.

Research-based pesticide companies

CropLife International (CLI), the association of research-based pesticide companies, has committed itself to promoting effective stewardship (CLI 2011; CLI 2020b; CLI 2020c). It underwrites the life cycle approach promoted by the Code of Conduct and considers that the overall aim of its stewardship approach is to maximize the benefits, and minimize any risk, from using crop protection products.

Stewardship of pesticides or crop protection products at CLI is broken down into seven interrelated elements going from research and development to management and disposal of obsolete pesticides (Figure 3.5-1). Pesticide companies are almost exclusively involved in stewardship at the research, development and manufacturing phases of the product lifecycle, but also conduct stewardship at the field level, where they concentrate on their own products. Regional industry associations and CLI, on the other hand deal more general aspects on stewardship principles that are relevant for all products. These are concentrated on the latter part of the life cycle, from use to waste management.

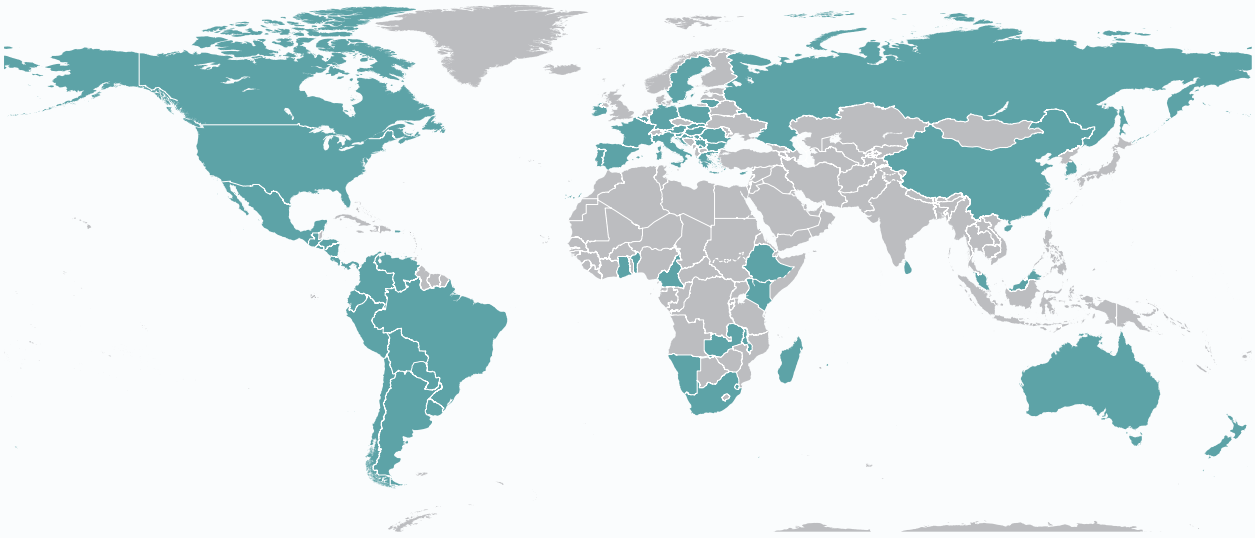
Training in good crop protection practices is considered a vital element of stewardship. CLI provides training on responsible use of crop protection products within the context of promoting IPM, with the underlying principle that a crop protection product should only be used when necessary: "as little as possible, as much as necessary". Training is often carried out in collaboration with sector ministries, farmer organizations and national or international development partners.

During the last decade the CropLife network reportedly trained between 100,000 and 500,000 extension agents, farmers and other pesticide users per year in 70 countries, mainly in Africa, Asia and Latin America (CLI 2020b). Moreover, in a single year (2018) CLI member companies recorded training over 22 million people. National CropLife Associations are also using various media, including the mass media, to reach large audiences with product stewardship information.

To minimize the development of pest resistance to pesticides, CLI member companies have committed themselves to include Mode of Action (MoA) codes on all their product labels by 2023 where this is permitted. Resistance management has been integrated into training programmes.

The management of empty pesticide containers is also part of the industry's stewardship life cycle approach. Its goal is to actively promote the expansion of container management programmes to new regions and countries, based on the lessons learned and "best practices" developed in regions where recycling is now the norm.

Figure 3.5-2 Countries with container management programmes (including pilots) supported by CropLife International and its members as of early 2020. Based on CLI (2020b) and personal communication.



Container management present

Container management programmes are typically undertaken in partnership with local authorities, recycling and/or disposal companies and the national government. Programmes supported by the pesticide industry were taking place in 58 countries in early 2020, an increase from 34 countries in 2010 (CLI 2020b) (Figure 3.5-2).

CLI has been actively involved over the last 25 years in projects to remove obsolete pesticides stocks. Its policy on obsolete stocks, adopted in 1995, is that member companies should consider providing assistance for the disposal of stocks they originally manufactured or supplied. Additionally, management practices and training have been introduced to help prevent future build-up of obsolete stocks, especially for private sector actors.

Work on obsolete stocks has been conducted through projects conducted by organizations such as FAO, the World Bank, the Africa Stockpiles Programme and bilateral donors. CLI has funded the disposal of pesticide stocks that originated from member companies.

CLI has extensively reported on its stewardship activities. Some individual activities have been externally reviewed, for instance if they were implemented through bilateral or multilateral donor programmes (CLI 2020b). However, no independent evaluation of the impact of product stewardship by CLI and its member countries on judicious pest and pesticide management has been conducted so far.

Generic pesticide producers

AgroCare, the association of generic pesticide producers, states that product stewardship is part of its mission (AgroCare n.d.). Nevertheless, no information was made available about stewardship activities by AgroCare or its member associations and companies despite of the fact that the largest share of the global pesticide market is now represented by generic, off-patent pesticides (Chapter 2.7.9).

3.5.3 Food and commodities industries

Pesticide stewardship is also conducted by the food and commodities industries. This primarily concerns the choice of pest management options and pesticides, occupational health precautions, and pesticide handling and application practices.

The most important way in which this stewardship is being shaped is through voluntary sustainability

standards developed by large producers of agricultural commodities, food processing industries and retailers, often in partnership with governments and NGOs. Examples are the Better Cotton Initiative, Fairtrade and Global G.A.P. (Chapter 2.7.12).

3.5.4 Pesticide users

Many pesticide users and their associations (e.g., farmer and grower associations and associations linking land managers (e.g., road, rail, municipalities) may also provide product stewardship. This often takes the form of technical guidance on pesticide handling and use, as well as container disposal.

3.5.5 Discussion: Corporate stewardship

Pesticide product stewardship by the private sector is an essential activity required to ensure optimal cost-efficacy of the pesticides being sold and to minimize health and environmental risks before, during and after their use. Extended producer responsibility for post-use aspects such empty container management as well as prevention and disposal of pesticide waste further contributes to the development of a circular economy.

Research-based pesticide companies conduct an important programme of product stewardship consisting primarily of information provision and training, resistance management, as well as contributions to empty container management and pesticide waste disposal. Companies producing mainly generic pesticides, on the other hand, do not seem to realize many stewardship activities, even though their market share has been significantly increasing.

Regarding information and training, friction may exist between product stewardship operated by pesticide industry and broader public or private objectives to reduce reliance on pesticides for pest and vector management. Pesticide manufacturers and distributors will logically focus their stewardship activities on proper use of the products that they market. This will not necessarily cover all elements needed to establish cropping systems that are less reliant on the use of pesticides.

Pesticide industry stewardship therefore needs to be balanced by independent advice on sustainable pest and pesticide management. This is especially important given the large number of farmers and public and private extension agents that are contacted by outreach activities of the pesticide industry on the one hand, and the decrease in public extension and advisory services observed in many countries on the other.

The food industry, commodity companies and (large) pesticide users conduct stewardship, mainly during the use stage of a pesticide, by ensuring good agricultural practices. Voluntary sustainability standards have become an important tool to promote such practices.

References

AgroCare (n.d.). Mission. https://www.agro-care.org/#mission. Accessed 15 November 2020.

Basel Convention (2011). *Strategic Framework for the Implementation of the Basel Convention for 2012-2021*. Decision BC-10/2. Conference of the Parties to the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposal, Tenth Meeting, 17-21 October 2011. http://www.basel.int/TheConvention/ConferenceoftheParties/ReportsandDecisions/tabid/3303/Default.aspx.

Basel Convention (2014). *Report on the Creation of a Baseline for the Mid-term and Final Evaluations of the Strategic Framework*. Document UNEP/CHW/OEWG.0/INF.3. Open-ended Working Group of the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal. Geneva, 16-19 September 2014. http://www.basel.int/Implementation/StrategicFramework/Evaluation/Midtermevaluation/tabid/3810/Default.aspx.

Basel Convention (2019). *Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal. Protocol on Liability and Compensation for Damage Resulting from Transboundary Movements of Hazardous Wastes and their Disposal. Texts and Annexes. Revised in 2019.* http://www.basel.int/theconvention/overview/textoftheconvention/tabid/1275/default.aspx.

Basel Convention (2020a). Parties to the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal. http://www.basel.int/Countries/StatusofRatifications/PartiesSignatories/tabid/4499/Default.aspx.

Basel Convention (2020b). Basel Convention (2020b). Ban Amendment: Amendment to the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal. http://www.basel.int/Countries/StatusofRatifications/BanAmendment/tabid/1344/Default.aspx. Accessed 17 November 2020.

Basel Convention (2020c). Strategic Framework: Overview. http://www.basel.int/Implementation/StrategicFramework/Overview/tabid/3807/Default.aspx. Accessed 19 November 2020.

Berry, K. (2016). *Codex MRLs – Use and Trends*. White Paper. Seattle: Bryant Christie Inc. https://bcglobal.bryantchristie.com/downloads/whitepaper_Codex_MRLs_Use_and_Trends_globalmrl.pdf.

Brandi, C., Blümer, D. and Morin, J.-F. (2019). When do international treaties matter for domestic environmental legislation? *Global Environmental Politics* 19(4), 14-44. https://doi.org/10.1162/glep_a_00524.

Center for International Environmental Law (2018). *Promoting the Effectiveness of the Rotterdam Convention: Ways Forward and Procedural Implications*. https://www.ciel.org/wp-content/uploads/2017/11/RCEffectiveness_legal-implication-analysis_Final.pdf.

Chowdhury, F.B., Dewan, G., Verma, V.R., Knipe, D.W., Isha, I.T., Faiz, M.A. *et al*. (2018). Bans of WHO Class I Pesticides in Bangladesh – suicide prevention without hampering agricultural output. *International Journal of Epidemiology* 47(1), 175-184. https://doi.org/10.1093/ije/dyx157.

Codex Alimentarius (2020). Codex Alimentarius International Food Standards. http://www.fao.org/fao-who-codexalimentarius/home/en/. Accessed 15 November 2020.

Convention on Biological Diversity (1992). *Text of the Convention*. New York: United Nations. https://www.cbd.int/convention/text/.

Convention on Biological Diversity (2016). *Indicators for the Strategic Plan for Biodiversity 2011 2020 and the Aichi Biodiversity Targets. Decision XIII/28 of the 13th meeting of the Conference of the Parties to the Convention on Biological Diversity. Cancun, Mexico, 4-17 December 2016*. Montreal: Secretariat of the Convention on Biological Diversity. https://www.cbd.int/doc/decisions/cop-13/cop-13-dec-28-en.pdf.

Convention on Biological Diversity (2020a). CBD website. https://www.cbd.int/. Accessed 15 November 2020.

Convention on Biological Diversity (2020b). Agricultural biodiversity. https://www.cbd.int/agro/. Accessed 15 November 2020.

Convention on Biological Diversity (2020c). *Global Biodiversity Outlook 5*. https://www.cbd.int/gbo5. Accessed 19 November 2020.

Copplestone, J.F. (1988). The development of the WHO Recommended Classification of Pesticides by Hazard. *Bulletin of the World Health Organization* 66 (5), 545-551. http://europepmc.org/backend/ptpmcrender.fcgi?accid=PMC2491185&blobtype=pdf.

CropLife International (2011). *A Stocktaking Report: Crop Protection Stewardship Activities of the Plant Science Industry 2005-2011*. https://croplife.org/wp-content/uploads/2014/04/A-Stocktaking-Report-Crop-Protection-Stewardship-Activities-of-the-Plant-Science-Industry-2005-2011.pdf.

CropLife International (2017). International Code of Conduct e-learning tool. https://croplife.org/crop-protection/regulatory/product-management/international-code-of-conduct/international-code-of-conduct-e-learning-tool/. Accessed 15 November 2020.

CropLife International (2020a). A responsible approach to Highly Hazardous Pesticides. Blog by Dr. Christoph Neumann. https://croplife.org/a-responsible-approach-to-highly-hazardous-pesticides/. Accessed 15 November 2020.

CropLife International (2020b). CropLife stewardship report 2005-2018 (in preparation).

CropLife International (2020c). Stewardship. https://croplife.org/crop-protection/stewardship/. Accessed 15 November 2020.

Danish Environmental Protection Agency (2017). *Pesticidstrategi 2017-2021* (Pesticide strategy 2017-2021). https://mst.dk/media/141516/pesticidstrategi2017-2021.pdf. Accessed 15 November 2020.

Danish Environmental Protection Agency (2019). *Bekæmpelsesmiddel-statistik 2017* (Pesticide statistics 2017). https://www2.mst.dk/Udgiv/publikationer/2019/06/978-87-7038-077-5.pdf. Accessed 15 November 2020.

Department of Agriculture, Forestry and Fisheries of South Africa (2010). *Pesticide Management Policy for South Africa*. Government Gazette No. 33899 of 24 December 2010. https://www.nda.agric.za/docs/policy/pesticidemanag.pdf. Accessed 15 November 2020.

Dinham, B. and Malik, S. (2003). Pesticides and human rights. *International Journal of Occupational and Environmental Health* 9(1), 40-52. https://doi.org/10.1179/107735203800328867. Accessed 15 November 2020.

European Commission (2017a). *A Study on the Effect of Listing of Chemicals in Annex III to the Rotterdam Convention. Final Report.* http://www.pic.int/Implementation/EnhancingtheeffectivenessoftheConvention/Impactsoflistingchemicals/tabid/7649/language/en-US/Default.aspx. Accessed 15 November 2020.

European Commission (2017b). *Overview Report on the Implementation of Member States' Measures to Achieve the Sustainable Use of Pesticides under Directive 2009/128/EC*. Report DG(SANTE) 2017-6291. Brussels: Directorate-General for Health and Food Safety.

European Commission (2020a). Report from the Commission to the European Parliament and the Council on the Experience Gained by Member States on the Implementation of National Targets Established in their National Action Plans and on Progress in the Implementation of Directive 2009/128/EC on the Sustainable Use of Pesticides. https://ec.europa.eu/food/sites/food/files/plant/docs/pesticides_sud_report-act_2020_en.pdf.

European Commission (2020b). Farm to fork strategy – for a fair, healthy and environmentally-friendly food system. https://ec.europa.eu/food/farm2fork_en. Accessed 15 November 2020.

European Union (2009a). Regulation (EC) No 1107/2009 of the European Parliament and of the Council of 21 October 2009 concerning the placing of plant protection products on the market. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:02009R1107-20181110. Accessed 15 November 2020.

European Union (2009b). Directive 2009/128/EC of the European Parliament and of the Council of 21 October 2009 establishing a framework for Community action to achieve the sustainable use of pesticides. https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX:02009L0128-20091125. Accessed 15 November 2020.

European Union (2019). Commission Directive 2019/782 of 15 May 2019 amending Directive 2009/128/EC of the European Parliament and of the Council as regards the establishment of harmonised risk indicators. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32019L0782. Accessed 15 November 2020.

Food and Agriculture Organization of the United Nations (1993). *Analysis of Government Responses to the First Questionnaire on the International Code of Conduct on the Distribution and Use of Pesticides*. Rome.

Food and Agriculture Organization of the United Nations (1996). *Analysis of Government Responses to the Second Questionnaire on the State of Implementation of the International Code of Conduct on the Distribution and Use of Pesticides*. Rome. https://www.fao.org/publications/card/es/c/CB5456EN/.

Food and Agriculture Organization of the United Nations (2006a). *International Code of Conduct on the Distribution and Use of Pesticides: Guidelines on Monitoring and Observance of the Code of Conduct*. Rome. http://www.fao.org/fileadmin/templates/agphome/documents/Pests_Pesticides/Code/Monitoring.pdf.

Food and Agriculture Organization of the United Nations (2006b). FAO Council Hundred and Thirty-first Session, Rome, 20-25 November 2006. http://www.fao.org/unfao/bodies/council/cl131/Index_en.htm. Accessed 15 November 2020.

Food and Agriculture Organization of the United Nations (2010). *International Code of Conduct on the Distribution and Use of Pesticides: Regular Monitoring Report – 2008*. Rome. http://www.fao.org/fileadmin/templates/agphome/documents/Pests_Pesticides/Code/Monitorin_questionnaire_2010.pdf.

Food and Agriculture Organization of the United Nations (2015). *Phasing Out Highly Hazardous Pesticides is Possible! Farmer Experiences in Growing Coffee without Endosulfan.* http://www.fao.org/publications/card/en/c/039416ce-83c5-424c-a6aa-25fecf999be9/.

Food and Agriculture Organization of the United Nations (2016). *Addressing Highly Hazardous Pesticides in Mozambique.* http://www.fao.org/publications/card/en/c/4bf666a3-a130-4c00-9184-824033417fe5/.

Food and Agriculture Organization of the United Nations (2019). *The State of the World's Biodiversity for Food and Agriculture*. Bélanger, J. and Pilling, D. (eds.). FAO Commission on Genetic Resources for Food and Agriculture Assessments. Rome. http://www.fao.org/3/CA3129EN/CA3129EN.pdf.

Food and Agriculture Organization of the United Nations (2020a). New list of guidelines. http://www.fao.org/agriculture/crops/thematic-sitemap/theme/pests/code/list-guide-new/en/. Accessed 2 May 2021.

Food and Agriculture Organization of the United Nations (2020b). Plant registration toolkit. http://www.fao.org/pesticide-registration-toolkit/en/. Accessed 2 May 2021.

Food and Agriculture Organization of the United Nations and International Plant Protection Convention (2017). *Guidelines for the Export, Shipment, Import and Release of Biological Control Agents and Other Beneficial Organisms*. https://www.ippc.int/static/media/files/publication/en/2017/05/ISPM_03_2005_En_2017-05-23_PostCPM12_InkAm.pdf.

Food and Agriculture Organization of the United Nations and United Nations Environment Programme (2020). *Legislative Approaches to Sustainable Agriculture and Natural Resources Governance*. FAO Legislative Study No. 114. Rome. https://doi.org/10.4060/ca8728en.

Food and Agriculture Organization of the United Nations and World Health Organization (2007). *Report: 1st FAO/WHO Joint Meeting in Pesticides Management and 3rd Session of the FAO Panel of Experts on Pesticide Management. 22-26 October 2007*. Rome. http://www.fao.org/fileadmin/templates/agphome/documents/Pests_Pesticides/Code/JMPM_2007_Report.pdf.

Food and Agriculture Organization of the United Nations and World Health Organization (2008). *Report: 2nd FAO/WHO Joint Meeting on Pesticide Management and 4th Session of the FAO Panel of Experts on Pesticide Management. 6-8 October 2008*. Geneva. http://www.fao.org/agriculture/crops/thematic-sitemap/theme/pests/code/panelcode/en/.

Food and Agriculture Organization of the United Nations and World Health Organization (2014). *The International Code of Conduct on Pesticide Management*. Rome. http://www.fao.org/agriculture/crops/thematic-sitemap/theme/pests/code/en/.

Food and Agriculture Organization of the United Nations and World Health Organization (2015). *International Code of Conduct on Pesticide Management: Guidelines on Good Labelling Practice for Pesticides (Revised). August 2015*. *Rome.* http://www.fao.org/3/a-i4854e.pdf.

Food and Agriculture Organization of the United Nations and World Health Organization (2016). *International Code of Conduct on Pesticide Management: Guidelines on Highly Hazardous Pesticides*. Rome. http://www.fao.org/3/a-i5566e.pdf.

Food and Agriculture Organization of the United Nations and World Health Organization (2017). *Report: 10th FAO/WHO Joint Meeting on Pesticide Management. 10-13 April 2017. New Delhi*. Rome. http://www.fao.org/fileadmin/templates/agphome/documents/Pests_Pesticides/Code/JMPM_2017_Report.pdf.

Food and Agriculture Organization of the United Nations and World Health Organization (2018). *Codex Alimentarius: Understanding Codex*. 5th edition. Rome. http://www.fao.org/3/CA1176EN/ca1176en.pdf.

Food and Agriculture Organization of the United Nations and World Health Organization (2019a). *Global Situation of Pesticide Management in Agriculture and Public Health. Report of a 2018 WHO-FAO Survey*. Rome and Geneva. https://apps.who.int/iris/handle/10665/329971.

Food and Agriculture Organization of the United Nations and World Health Organization (2019b). *Report: 12th FAO/WHO Joint Meeting on Pesticide Management. 19-22 November 2019, Geneva, Switzerland.* Geneva. https://apps.who.int/iris/handle/10665/331620.

Food and Agriculture Organization of the United Nations and World Health Organization (2019c). *Codex Strategic Plan 2020-2025*. Rome and Geneva. http://www.fao.org/3/ca5645en/CA5645EN.pdf.

Food and Agriculture Organization of the United Nations and World Health Organization (2019d). *Report: 11th FAO/WHO Joint Meeting on Pesticide Management.* Geneva. http://www.fao.org/agriculture/crops/thematic-sitemap/theme/pests/code/panelcode/en/.

Food and Agriculture Organization of the United Nations and World Health Organization (2020). *International Code of Conduct on Pesticide Management: Guidelines on Pesticide Legislation. Second edition*. Rome. https://doi.org/10.4060/cb0916en.

Goldenman, G. and Pozo Vera, E. (eds.) (2008). *International Tools for Preventing Local Pesticide Problems: A Consolidated Guide to the Chemical Codes and Conventions*. Brighton, UK: Pesticide Action Network. https://issuu.com/pan-uk/docs/consolidated_guide_to_chemical_code?e=28041656/64929590.

Goldenman, G. and Rengam, S. (1987). *Problem Pesticides, Pesticide Problems: A Citizens' Action Guide to the International Code of Conduct on the Distribution and Use of Pesticides*. Penang, Malaysia: International Organization of Consumers Unions and Pesticide Action Network.

Government of France (2018). Écophyto II+: Reducing and Improving the Use of Plant Protection Products. Ministère de la Transition écologique et solidaire. https://ec.europa.eu/food/sites/food/files/plant/docs/pesticides_sup_nap_fra-ecophyto-2plus_plan_en.pdf.

Handford, C.A., Elliott, C.T. and Campbell, K. (2015). A review of the global pesticide legislation and the scale of challenge in reaching the global harmonization of food safety standards. *Integrated Environmental Assessment and Management* 11(4), 525-536. https://doi.org/10.1002/ieam.1635. Accessed 15 November 2020

Intergovernmental Forum on Chemical Safety (2003). *Fourth Session of the Intergovernmental Forum on Chemical Safety, Bangkok, Thailand, 1-7 November 2003. Final Report, Executive Summary.* https://www.who.int/ifcs/documents/forums/forum4/final_report/en/.

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (2019). *Summary for Policymakers of the Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. Díaz, S., Settele, J., Brondízio, E.S., Ngo, H.T., Guèze, M., Agard, J. et al. (eds.). Bonn: IPBES Secretariat. https://doi.org/10.5281/zenodo.3553579.

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (2020). IPBES website. https://ipbes.net/. Accessed 15 November 2020.

International Labour Organization (1981). C155 - Occupational Safety and Health Convention, 1981 (No. 155). https://www.ilo.org/dyn/normlex/en/f?p=NORMLEXPUB:12100:0::NO:12100:P12100_INSTRUMENT_ID:312300:NO. Accessed 15 November 2020.

International Labour Organization (1999). C182 – Worst Forms of Child Labour Convention, 1999 (No. 182). https://www.ilo.org/dyn/normlex/en/f?p=NORMLEXPUB:12100:0::NO:12100:P12100_INSTRUMENT_ID:312327:NO. Accessed 15 November 2020.

International Labour Organization (2007). *Rooting Out Child Labour from Cocoa Farms. Paper No. 2: Safety and Health Hazards*. Geneva. http://www.ilo.org/ipecinfo/product/viewProduct.do?productId=6445.

International Labour Organization (2016). *Child Labour in Cotton: A Briefing*. Geneva. http://www.ilo.org/ipec/Informationresources/WCMS_IPEC_PUB_29655/lang--en/index.htm.

International Labour Organization (2017a). *Global Estimates of Child Labour: Results and Trends, 2012-2016*. Geneva. https://www.ilo.org/global/publications/books/WCMS_575499/lang--en/index.htm.

International Labour Organization (2017b). *Child Labour in the Primary Production of Sugarcane*. Geneva. http://www.ilo.org/ipec/Informationresources/WCMS_IPEC_PUB_29635/lang--en/index.htm.

International Labour Organization (2020a). Ratifications of C182 – Worst Forms of Child Labour Convention, 1999 (No. 182). https://www.ilo.org/dyn/normlex/en/f?p=1000:11300:0::NO:11300:P11300_INSTRUMENT_ID:312327. Accessed 19 November 2020.

International Labour Organization (2020b). Ratifications of C155 – Occupational Safety and Health Convention, 1981 (No. 155). https://www.ilo.org/dyn/normlex/en/f?p=1000:11300:0::NO:11300:P11300_INSTRUMENT_ID:312300. Accessed 19 November 2020.

International Plant Protection Convention (1997). *1997 – International Plant Protection Convention (New Revised Text).* https://www.ippc.int/en/publications/131/.

International Plant Protection Convention (2020). IPPC website. https://www.ippc.int/en/. Accessed 15 November 2020.

Jansen, K. and Dubois, M. (2014). Global pesticide governance by disclosure: Prior Informed Consent and the Rotterdam Convention. In *Transparency in Global Environmental Governance: Critical Perspectives*. Gupta, A. and Mason, M. (eds.). Cambridge, MA: MIT Press. 107-131. https://mitpress.mit.edu/books/transparency-global-environmental-governance.

Jepson, P.C., Murray, K., Bach, O., Bonilla, M.A. and Neumeister, L. (2020). Selection of pesticides to reduce human and environmental health risks: A global guideline and minimum pesticides list. *The Lancet Planetary Health*, 4(2), e56-e63. https://doi.org/10.1016/S2542-5196(19)30266-9.

Jin, S. and Zhou, F. (2018). Zero growth of chemical fertilizer and pesticide use: China's objectives, progress and challenges. *Journal of Resources and Ecology* 9(1), 50-58. https://doi.org/10.5814/j.issn.1674-764x.2018.01.006.

Kellenberg, D. and Levinson, A. (2014). Waste of effort? International Environmental Agreements. *Journal of the Association of Environmental and Resource Economists* 1(1/2), 135-169. https://www.journals.uchicago.edu/doi/full/10.1086/676037.

Konradsen, F., van der Hoek, W., Cole, D.C., Hutchinson, G., Daisley, H., Singh, S. and Eddleston, M. (2003). Reducing acute poisoning in developing countries – options for restricting the availability of pesticides. *Toxicology* 192(203), 249-261. https://doi.org/10.1016/S0300-483X(03)00339-1.

Kudsk, P., Nistrup Jørgensen, L. and Ørum, J.E. (2018). Pesticide Load – A new Danish pesticide risk indicator with multiple applications. *Land Use Policy* 70, 384-393. https://doi.org/10.1016/j.landusepol.2017.11.010.

Lee, R., den Uyl, R. and Runhaar, H. (2019). Assessment of policy instruments for pesticide use reduction in Europe; Learning from a systematic literature review. *Crop Protection* 126, 104929. https://doi.org/10.1016/j.cropro.2019.104929.

Manuweera, G., Eddleston, M., Egodage, S. and Buckley, N.A. (2008). Do targeted bans of insecticides to prevent deaths from self-poisoning result in reduced agricultural output? *Environmental Health Perspectives* 116(4), 492-495. https://doi.org/10.1289/ehp.11029.

Matthews, G., Zaim, M., Singh Yadav, R., Soares, A., Hii, J., Ameneshewa, B. *et al*. (2011). Status of legislation and regulatory control of public health pesticides in countries endemic with or at risk of major vector-borne diseases. Environmental Health Perspectives 119, 1517-1522. http://dx.doi.org/10.1289/ehp.1103637.

McClary, M.J. and Goldstein, J.B. (2017). FIFRA at 40: The need for felonies for pesticide crimes. *Environmental Law Reporter 9*, 10767-10785. https://www.eli.org/sites/default/files/elr/featuredarticles/47.10767.pdf.

Millennium Ecosystem Assessment (2005). Guide to the Millennium Ecosystem Assessment Reports. https://www.millenniumassessment.org/en/index.html. Accessed 14 November 2020.

Minamata Convention (2013). *Global Mercury Assessment 2013: Sources, Emissions, Releases and Environmental Transport*. Geneva: United Nations Environment Programme. https://wedocs.unep.org/bitstream/handle/20.500.11822/7984/-Global%20Mercury%20Assessment-201367.pdf?sequence=3&isAllowed=yhttps://wedocs.unep.org/handle/20.500.11822/7984.

Minamata Convention (2017). *Minamata Convention on Mercury. Text and Annexes.* http://mercuryconvention.org/Convention/Text/tabid/3426/language/en-US/Default.aspx.

Minamata Convention (2019). *Global Mercury Assessment 2018*. Geneva: United Nations Environment Programme. https://www.unenvironment.org/resources/publication/global-mercury-assessment-2018.

Ministerie van Economische Zaken (The Netherlands) (2013) Gezonde groei, duurzame oogst – Tweede nota duurzame gewasbescherming periode 2013 tot 2023. Ministerie van Economische Zaken, The Hague https://www.rijksoverheid.nl/documenten/rapporten/2013/05/14/gezonde-groei-duurzame-oogst-tweede-nota-duurzame-gewasbescherming. Accessed 16 November 2020.

Montreal Protocol (2018). *Methyl Bromide Technical Options Committee: 2018 Assessment Report.* United Nations Environment Programme, Ozone Secretariat. Nairobi. https://ozone.unep.org/sites/default/files/2019-04/MBTOC-assessment-report-2018_1.pdf.

Núñez-Rocha, T. and Martínez-Zarzoso, I. (2019). Are international environmental policies effective? The case of the Rotterdam and the Stockholm Conventions. *Economic Modelling* 81, 480-502. https://doi.org/10.1016/j.econmod.2018.04.013. Accessed 16 November 2020.

Nurick, R. (2019). *Independent Evaluation of the Strategic Approach from 2006-2015. Final Report.* http://www.saicm.org/Portals/12/Documents/reporting/FinalReport_Independent-Evaluation-SAICM-2006-2015.pdf.

Organisation for Economic Co-operation and Development (2012). *OECD Guidance on Pesticide Compliance and Enforcement Best Practices*. Series on Pesticides No. 71. https://www.oecd.org/env/ehs/pesticides-biocides/Pesticides_Compliance_Guidance.pdf.

Organisation for Economic Co-operation and Development (2016). *Extended Producer Responsibility – Updated Guidance for Efficient Waste Management.* https://doi.org/10.1787/9789264256385-en.

Organisation for Economic Co-operation and Development (2020a) Mutual Acceptance of Data (MAD). https://www.oecd.org/chemicalsafety/testing/mutualacceptanceofdatamad.htm. Accessed 18 November 2020.

Organisation for Economic Co-operation and Development (2020b). *Saving Costs in Chemicals Management: How the OECD Ensures Benefits to Society.* https://doi.org/10.1787/9789264311718-en.

Orozco, F.A., Code, D.C., Forbes, G., Kroschel, J., Wanigaratne, S. and Arica, D. (2009). Monitoring adherence to the International Code of Conduct – Highly Hazardous Pesticides in Central American agriculture and farmers' rights to health. *International Journal of Occupational and Environmental Health* 15, 255-268. https://doi.org/10.1179/oeh.2009.15.3.255.

Persson, L., Karlsson-Vinkhuyzen, S., Lai, A., Persson, Å and Fick, S. (2017). The Globally Harmonized System of Classification and Labelling of Chemicals – explaining the legal implementation gap. *Sustainability* 9, 2176. https://doi.org/10.3390/su9122176.

Pesticide Action Network (2010). *Communities in Peril: Global Report on Health Impacts of Pesticide Use in Agriculture*. Penang: Pesticide Action Network Asia and the Pacific. http://paninternational.org/wp-content/uploads/2013/12/PAN-Global-Report-communities-peril.pdf.

Pesticide Action Network (2021). *PAN International List of Highly Hazardous Pesticides (PAN List of HHPs). March 2021*. Hamburg: Pesticide Action Network International. http://pan-international.org/wp-content/uploads/PAN_HHP_List.pdf.

Pesticide Action Network Asia and the Pacific and Pesticide Action Network UK (2016). *FAO Code Monitoring Module: An updated users' guide to monitoring the implementation of the FAO/WHO International Code of Conduct on Pesticide Management, 2014*. Updated 2016. https://files.panap.net/resources/FAO-Code-Monitoring-Module.pdf.

Planbureau voor de Leefomgeving (The Netherlands) (2019). Geïntegreerde gewasbescherming nader beschouwd . Tussenevaluatie van de nota Gezonde Groei, Duurzame Oogst. https://www.pbl.nl/publicaties/geintegreerde-gewasbescherming-nader-beschouwd. Accessed 15 November 2020.

Product Stewardship Institute (2020). What is product stewardship? https://www.productstewardship.us/page/Definitions. Accessed 15 November 2020.

Public Eye (2017). UN Experts fail to deliver assessment of Bayer and Syngenta compliance with the International Code of Conduct on Pesticide Management. Press release, 23 November. https://www.publiceye.ch/en/news/detail/un-experts-fail-to-deliver-assessment-of-bayer-and-syngenta-compliance-with-the-international-code-of-conduct-on-pesticide-management. Accessed 15 November 2020.

Reeves, M., Schafer, K., Hallward, K. and Katten, A. (1999). *Fields of Poison: California Farmworkers and Pesticides*. San Francisco: Pesticide Action Network North America. https://www.panna.org/sites/default/files/FieldsofPoison2002Eng.pdf. Accessed 15 November 2020.

Rotterdam Convention (2015). *Report of the Conference of the Parties to the Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade on the Work of Its Seventh Meeting. Geneva, 4-15 May 2015*. UNEP/FAO/ RC/COP.7/21. http://www.pic.int/TheConvention/ConferenceoftheParties/Meetings/COP7/Overview/tabid/4252/language/en-US/Default.aspx.

Rotterdam Convention (2017). *Study of Information on the Impacts of Listing Chemicals in Annex III to the Rotterdam Convention*. Conference of the Parties to the Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade. Eighth meeting Geneva, 24 April-5 May 2017. UNEP/FAO/RC/COP.8/INF/21. http://www.pic.int/Implementation/EnhancingtheeffectivenessoftheConvention/Impactsoflistingchemicals/tabid/7649/language/en-US/Default.aspx.

Rotterdam Convention (2018). *Report of the Working Group on Enhancing the Effectiveness of the Rotterdam Convention on the Work of Its First Meeting*. Riga, Latvia, 4-6 June 2018. UNEP/FAO/RC/EFF.1/4. http://www.pic.int/?tabid=6306.

Rotterdam Convention (2019). *Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade. Text and Annexes. Revised in 2019.* http://www.pic.int/TheConvention/Overview/TextoftheConvention/tabid/1048/language/en-US/Default.aspx.

Rotterdam Convention (2020a). Status of ratifications. http://www.pic.int/Countries/Statusofratifications/tabid/1072/language/en-US/Default.aspx. Accessed 17 November 2020.

Rotterdam Convention (2020b). Workshops. http://www.pic.int/Implementation/TechnicalAssistance/Workshops/tabid/4637/language/en-US/Default.aspx. Accessed 16 November 2020.

Rotterdam Convention (2020c). Import responses. http://www.pic.int/Procedures/ImportResponses/Database/tabid/1370/language/en-US/Default.aspx. Accessed 18 November 2020.

Rotterdam Convention (n.d. a). Severely Hazardous Pesticide Formulations: Overview. http://www.pic.int/Default.aspx?tabid=1191. Accessed 18 November 2020.

Rotterdam Convention (n.d. b). Annex III chemicals. http://www.pic.int/TheConvention/Chemicals/AnnexIIIChemicals. Accessed 18 November 2020.

Rotterdam Convention (n.d.c). Rotterdam Convention. PIC Circulars. http://www.pic.int/Implementation/PICCircular/tabid/1168/language/en-US/Default.aspx. Accessed 16 November 2020.

Standards and Trade Development Facility (2020). STDF: Standards and Trade Development Facility. https://www.standardsfacility.org. Accessed 1 May 2021.

Stockholm Convention (2017a). Text of the Convention. http://chm.pops.int/TheConvention/Overview/TextoftheConvention/tabid/2232/Default.aspx. Accessed 18 November 2020.

Stockholm Convention (2017b). *Second Global Monitoring Report. Global Monitoring Plan for Persistent Organic Pollutants*. Document UNEP/POPS/COP.8/INF/38. Conference of the Parties to the Stockholm Convention on Persistent Organic Pollutants. Eighth meeting, Geneva, 24 April-5 May 2017. United Nations Environment Programme, Geneva. Accessed 17 November 2020.

Stockholm Convention (2017c). *Report on the Effectiveness Evaluation of the Stockholm Convention on Persistent Organic Pollutants*. UNEP/POPS/COP.8/INF/40. Geneva: United Nations Environment Programme. http://chm.pops.int/TheConvention/ConferenceoftheParties/Meetings/COP8/tabid/5309/Default.aspx.

Stockholm Convention (2019a). Status of ratification. http://chm.pops.int/Countries/StatusofRatifications/PartiesandSignatoires/tabid/4500/Default.aspx. Accessed 17 November 2020.

Stockholm Convention (2019b). All POPs listed in the Stockholm Convention. http://chm.pops.int/TheConvention/ThePOPs/AllPOPs/tabid/2509/Default.aspx. Accessed 17 November 2020.

Stockholm Convention (2019c). Persistent Organic Pollutant Review Committee: Overview and Mandate. http://chm.pops.int/Default.aspx?tabid=2806. Accessed 17 November 2020.

Stockholm Convention (2019d). Effectiveness evaluation: Overview. http://chm.pops.int/Implementation/EffectivenessEvaluation/Overview/tabid/369/Default.aspx. Accessed 17 November 2020.

Stockholm Convention (2019e). Global Monitoring Plan. http://chm.pops.int/Implementation/GlobalMonitoringPlan/Overview/tabid/83/Default.aspx. Accessed 15 November 2020.

Stockholm Convention (2019f). *Report of the DDT Expert Group on the Assessment of the Production and use of DDT and its alternatives for Disease Vector Control*. UNEP/POPS/COP.9/INF/6. http://www.pops.int/TheConvention/ConferenceoftheParties/Meetings/COP9/tabid/7521/Default.aspx.

Strategic Approach to International Chemicals Management (2011). *First Progress Report on Implementation of Progress in Implementation of the Strategic Approach: Results of Preliminary Data Collection for 2009 and 2010*. SAICM/OEWG.1/INF/2. http://www.saicm.org/Portals/12/documents/meetings/OEWG1/inf/OEWG1%20INF%202_first%20progress%20report-final.pdf.

Strategic Approach to International Chemicals Management (2015a). *Strategy to Address Highly Hazardous Pesticides in the Context of the Strategic Approach to International Chemicals Management.* http://www.saicm.org/Portals/12/Documents/EPI/HHP%20strategy%20English.pdf.

Strategic Approach to International Chemicals Management (2015b). Fourth Session of the International Conference on Chemicals Management (ICCM4). http://www.saicm.org/Meetings/ICCM4/tabid/5464/language/en-US/Default.aspx. Accessed 15 November 2020.

Strategic Approach to International Chemicals Management (2019). *Progress in Strategic Approach Implementation for 2014-2016*. SAICM/OEWG.3/INF/4. http://www.saicm.org/Portals/12/Documents/meetings/OEWG3/inf/OEWG3-INF-4-2014-16-third-Progress-Report-.pdf.

Strategic Approach to International Chemicals Management (n.d.). *Strategic Approach to International Chemicals Management: SAICM Texts and Resolutions of the International Conference on Chemicals Management.* http://www.saicm.org/Portals/12/Documents/saicmtexts/New%20SAICM%20Text%20with%20ICCM%20resolutions_E.pdf.

Tasaki, T., Tojo, N. and Lindhqvist, T. (2018). Differences in perception of extended producer responsibility and product stewardship among stakeholders: An international questionnaire survey and statistical analysis. *Journal of Industrial Ecology* 23(2), 438-451. https://doi.org/10.1111/jiec.12815.

ter Horst, M., Deneer, J., van der Valk, H., Aung, K.O., San Lwin, S., Raw, S. and Krook, J. (2018). *Risk Reduction of Pesticides in Myanmar: Interim Report of the Dutch-Myanmar Project of Pesticide Registration and Pesticide Risk Eduction 2016-2018*. Wageningen, the Netherlands: Wageningen University and Research. https://www.wur.nl/nl/Publicatie-details.htm?publicationId=publication-way-353337353338.

United Nations Economic Commission for Europe (2020a). *Globally Harmonized System of Classification and Labelling of Chemicals (GHS). Eighth Revised Edition.* https://www.unece.org/trans/danger/publi/ghs/ghs_rev08/08files_e.html.

United Nations Economic Commission for Europe (2020b). GHS implementation. https://www.unece.org/trans/danger/publi/ghs/implementation_e.html. Accessed 15 November 2020.

United Nations Environment Programme (2019a). Methyl bromide. https://www.unenvironment.org/ozonaction/what-we-do/methyl-bromide. Accessed 15 May 2020.

United Nations Environment Programme (2019b). *Environmental Rule of Law: First Global Report*. Nairobi. https://wedocs.unep.org/bitstream/handle/20.500.11822/27279/Environmental_rule_of_law.pdf?sequence=1&isAllowed=y. Accessed 15 May 2020.

United Nations Environment Programme (2020a). The Montreal Protocol on Substances that Deplete the Ozone Layer. https://ozone.unep.org/treaties/montreal-protocol. Accessed 17 November 2020.

United Nations Environment Programme (2020b). Minamata Convention on Mercury: Parties and Signatories. http://www.mercuryconvention.org/Countries/Parties/tabid/3428/language/en-US/Default.aspx. Accessed 18 November 2020.

United Nations Environment Programme (2020c). The Bamako convention. https://www.unenvironment.org/explore-topics/environmental-rights-and-governance/what-we-do/meeting-international-environmental. Accessed 19 November 2020.

United Nations Environment Programme (2022). *Synthesis Report on Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them*. https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

United Nations Treaty Collection (2020). 3.b. Basel Protocol on Liability and Compensation for Damage Resulting from Transboundary Movements of Hazardous Wastes and their Disposal. https://treaties.un.org/Pages/ViewDetails.aspx?src=IND&mtdsg_no=XXVII-3-b&chapter=27&clang=_en. Accessed 16 November 2020.

van den Berg, H., Gu, B., Grenier, B., Kohlschmid, E., Al-Eryani, S., da Silva Bezerra H.S. *et al*. (2020). Pesticide lifecycle management in agriculture and public health: Where are the gaps? *Science of The Total Environment* 742, 140598. https://doi.org/10.1016/j.scitotenv.2020.140598.

van den Berg, M., Kypke, K., Kotz, A., Tritscher, A., Lee, S.Y., Magulova, K. *et al*. (2017). WHO/UNEP global surveys of PCDDs, PCDFs, PCBs and DDTs in human milk and benefit-risk evaluation of breastfeeding. *Archives of Toxicology* 91(1), 83-96. https://doi.org/10.1007/s00204-016-1802-z.

World Customs Organization (2020). What is the Harmonized System (HS)? http://www.wcoomd.org/en/topics/nomenclature/overview/what-is-the-harmonized-system.aspx. Accessed 14 November 2020.

World Health Organization (2004). *Public Health Pesticide Management Practices by WHO Member States – Report of a Survey 2003-2004*. Geneva. https://apps.who.int/iris/handle/10665/68697/.

World Health Organization (2011a). *Public Health Pesticide Registration and Management Practices by WHO Member States – Report of a 2010 Survey*. Zaim, M./WHO Pesticide Evaluation Scheme (WHOPES) (eds.). Geneva. https://apps.who.int/iris/handle/10665/44542.

World Health Organization (2011b). *Guidelines on Public Health Pesticide Management Policy for the WHO African Region*. Zaim, M./WHO Pesticide Evaluation Scheme (WHOPES) (eds.). Geneva. https://apps.who.int/iris/handle/10665/97939.

World Health Organization (2017). *Chemicals Road Map: Road Map to Enhance Health Sector Engagement in the Strategic Approach to International Chemicals Management Towards the 2020 Goal and Beyond*. https://www.who.int/publications-detail/chemicals-road-map.

World Health Organization (2019). *The WHO Recommended Classification of Pesticides by Hazard and Guidelines to Classification, 2019 Edition*. Geneva. https://www.who.int/publications/i/item/9789240005662.

World Health Organization (2020a). Ten chemicals of major public health concern. https://www.who.int/teams/environment-climate-change-and-health/chemical-safety-and-health/health-impacts/chemicals. Accessed 14 November 2020.

World Health Organization (2020b). Vector control product prequalifiction. https://extranet.who.int/pqweb/vector-control-products. Accessed 2 May 2021.

World Health Organization (2020c). Control of neglected tropical diseases. https://www.who.int/teams/control-of-neglected-tropical-diseases/vector-ecology-and-management/vector-control. Accessed 2 May 2021.

World Trade Organization (2010). *The WTO Agreements Series: Sanitary and Phytosanitary Measures*. Geneva. https://www.wto.org/english/res_e/booksp_e/agrmntseries4_sps_e.pdf.

World Trade Organization (2014). *The WTO Agreements Series: Technical Barriers to Trade*. Geneva. https://www.wto.org/english/res_e/publications_e/tbttotrade_e.pdf.

World Trade Organization (2017). TRIPS agreement (as amended on 23 January 2017). https://www.wto.org/english/docs_e/legal_e/31bis_trips_01_e.htm. Accessed 14 November 2020.

Yeung, M., Kerr, W.A., Coomber, B., Lantz, M. and McConnell, A. (2018). Declining harmonization in maximum residue levels for pesticides. *British Food Journal* 120(4), 901-913. https://doi.org/10.1108/BFJ-05-2017-0291.

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them

Envisioning A Chemical-Safe World

Chapter 4 of 12

Environmental and health effects of pesticide use

Contents

1 Global drivers, actors and policies affecting pesticides and fertilizer use

2 Status and trends of pesticide use

3 The regulatory and policy environment for pesticide management

About vi

4 Environmental and health effects of pesticide use 1

4.1 Overview 1

4.2 Pesticide hazards and risks 2

4.3 Adverse environmental effects of pesticide use 5

4.3.1 Overview 5
4.3.2 Pesticide concentrations in the environment 10
4.3.3 Sustainability of agricultural production and vector control 25
4.3.4 The terrestrial environment 37
4.3.5 The aquatic environment 50
4.3.6 Pesticides and biodiversity 55
4.3.7 Knowledge gaps on environmental effects of pesticides 59

4.4 Adverse human health effects of pesticides 62

4.4.1 Overview 62
4.4.2 Acute unintentional pesticide poisoning 66
4.4.3 Chronic pesticide poisoning 75
4.4.4 Self-poisoning 81
4.4.5 Factors affecting occupational and residential pesticide exposure 83
4.4.6 Pesticide residues in food 88
4.4.7 Pesticides and antimicrobial resistance 98
4.4.8 Knowledge gaps on human health effects of pesticides 105

References 107

5. The environmental, human health and economic impacts of pesticides

6 Current pesticide risk reduction and risk management

7 Status and trends of fertilizer use

8 The regulatory and policy environment of fertilizer management and use

9 Environmental and health effects of fertilizer use

10 The impact of fertilizer use

11 Current fertilizer risk reduction and risk management

12 Transformative actions to minimize the adverse impacts of pesticide and fertilizers

List of Boxes

Box 4.2-1	Common hazard and risk assessment terminology applied in this report.	3
Box 4.2-2	Many risk assessments are generally conducted, as part of the pesticide registration process, to assess whether a pesticide may pose unacceptable risks to human or animal health or the environment.	4
Box 4.3-1	Principles of environmental risk assessment of pesticides.	9
Box 4.3.4-1	Selected examples of major functions performed by soil organisms.	37
Box 4.3.4-2	Effects of anticoagulent rodenticides on vertebrate wildlife.	42
Box 4.3.4-3	Endocrine disruption and wildlife.	46
Box 4.3.4-4	Effects observed on reptiles as a result of insecticides applied for locust control in Africa and Australia are highly dependent on the type of insecticide and the way it is applied.	47
Box 4.4-1	Aspects of pesticide epidemiology.	74
Box 4.4-3	Systematic review and meta-analyses of epidemiological studies examining the association between pesticide exposure and health outcomes.	78
Box 4.4-3	Maximum Residue Limits (MRLs).	90

List of Figures

Figure 4.2-1	Pesticide risk assessments are often prospective, in the sense that they are carried out before a pesticide is put on the market. Monitoring or studies of (adverse) effects of pesticide use are consecutive evaluations made after a pesticide is in use. As sufficient data are available, information about pesticide effects contributes to an evaluation of their impact on human health, the environment or society.	2
Figure 4.3.2-1	Pesticides can contaminate surface water and groundwater through many different routes.	11
Figure 4.3.2-2	Global crop area and the distribution of regulatory threshold level exceedance rates for reported measured insecticide concentrations (MIC) aggregated in 1° grid cells.	13
Figure 4.3.2-3	Proportion of sampling sites in European freshwater ecosystems showing exceedance of acute regulatory thresholds by different chemical groups.	14
Figure 4.3.2-4	The proportion of streams in the United States with one or more pesticides that exceeded a chronic aquatic life benchmark was very similar in the periods 1991-2001 and 2002-2011 for agricultural and mixed land use streams, but much greater in the period 2002-2011 for urban streams.	15
Figure 4.3.2-5	Time trends for γ-HCH (lindane) measured in air at the Canadian Great Lakes site, Point Petre. Atmospheric decline rates of γ-HCH have accelerated in the Arctic and the Great Lakes region following North American restriction of the pesticide lindane.	16
Figure 4.3.2-6	Global mean soil concentrations of most organochlorine pesticides in agricultural soils (A) and background soils (B) did not significantly decrease over time (decade one (1993-2002) to decade two (2003- 2012).	17
Figure 4.3.2-7	Factors influencing pesticide leaching to groundwater.	20
Figure 4.3.2-8	Relative availability of local and national groundwater monitoring/review studies per continent (total = 103 publications between 2010 and 2020).	20
Figure 4.3.2-9	Geographical distribution of available local and national groundwater monitoring/ review studies (total = 103 publications between 2010 and 2020).	21
Figure 4.3.2-10	Temporal trend in (A) chlordecone (CLD) and (B) its metabolite chlordecone-5b-hydro (5bCLD) between 2009 and 2015 in Chalvet (Martinique).	22

Figure 4.3.3-1 Global increase of unique cases of herbicide resistance. A unique case is a weed species combined with a herbicide mode of action. 26

Figure 4.3.3-2 The evolution of pesticide resistance in arthropods shows a steady increase in the number of species, pesticide compounds and cases from 1914 to 2019. 27

Figure 4.3.3-3 Arthropods were becoming resistant to a pesticide as rapidly in 2010 as in the 1960s, indicating that there had not been a change in the "durability" of a pesticide over time. No significant relation was observed between the number of generations it takes for an arthropod to become resistant and the year of introduction of a pesticide. 27

Figure 4.3.3-4 Number of classes to which resistance was confirmed in at least one malaria vector in at least one monitoring site, 2010–2019. 28

Figure 4.3.3-5 Global trend in pest resistance to Bt crops: (A) hectares planted with Bt crops (B) cumulative cases of field-evolved practical resistance to Bt crops. 29

Figure 4.3.3-6 Main routes of exposure of honeybees to pesticides; similar routes of exposure are likely for other bees. 31

Figure 4.3.4-1 Pathways through which pesticide application can ultimately lead to population declines in terrestrial vertebrates. 40

Figure 4.3.4-2 Potential kill of corn bunting due to the application of the 294 pesticides included in an analysis in Great Britain fell by about 80 per cent between 1990 and 2016. 41

Figure 4.3.5-1 Risk quotients for algae (squares), fish (circles) and invertebrates (triangles) as ratios of maximum detected global field concentrations and acute standard toxicity data. Open symbols indicate that toxicity was provided as "greater than" values. Risk quotients > 0.01 (dashed line) and > 0.1 (solid line) indicate moderate and high risks, respectively. 53

Figure 4.3.6-1 The main drivers (A) and factors (B) associated with insect declines. Pesticides are identified as the second most important factor associated with declines of insect species' abundance and biodiversity. Shown is (A) the number and (B) the percentage of studies in which specific drivers and factors of decline were determined. 58

Figure 4.3.6-2 Concentration-response dependence between the mean pesticide concentration and mean overall aquatic macroinvertebrate family richness measured in sampling sites in Europe and Australia. The dashed horizontal lines indicate maximum and minimum mean richness and are marked with the percentages compared to maximum richness. The error bars indicate 95 per cent confidence intervals. 58

Figure 4.4-1 Human health effects of pesticides can show up in the short term (acute) or after a long period (chronic) and may be the result of activities leading to both acute and chronic exposure. Shown are the most common routes of poisoning (less likely options are indicated by thinner arrows). 62

Figure 4.4-2 Poison centres are primarily concentrated in high and middle income countries. They are much rarer in some low income parts of the world. 67

Figure 4.4-3 Simplified representation of how pesticide poisoning information may, or may not, reach a poison centre database. Poison centres may use other sources to complete their poisoning statistics, such as surveillance reports, dedicated monitoring studies or media releases. 68

Figure 4.4-4 Pesticide poisoning as a fraction of total reported poisoning cases is positively correlated with the percentage rural population in the respective countries. 70

Figure 4.4-5 Regional distribution of the estimates of non-fatal and fatal cases of pesticide poisoning, all severities combined. 72

Figure 4.4-6 The large majority of studies on occupational exposure to pesticides have been conducted in high and upper-middle income countries. 76

Figure 4.4-7 Use of PPE items in different regions of the world. 85

Figure 4.4-8 On average, 58 per cent of countries have systems in place for monitoring pesticide residues in food and feed, but large differences exist among world regions (total number of responding countries = 50). 88

Figure 4.4-9 Pesticide MRL exceedances of food commodities imported into the EU+ (EU Member States, Norway and Iceland) between 2010 and 2018 ranged between 5.5 and 8.5 per cent of samples (orange bars). Lower MRL exceedances are found for food originating from within the EU+, but these have been steadily increasing from 1.5 in 2010 to 3.1 per cent in 2018 (blue bars). 92

Figure 4.4-10 The number of fungal species with reported antifungal resistance has been increasing over time, both in agriculture and in human health. Increasing colour intensity reflects a higher number of reports. The plant maps depict records of resistance of crop pathogens to azole fungicides (blue scale). 100

List of Tables

Table 4.2-1 Some examples of environmental or health effects of pesticides which had not been identified during prospective risk assessments, but became known after their use was authorized. 4

Table 4.3.1-1 Principal lines of evidence investigated for this report to identify the environmental effects of pesticide use. 5

Table 4.3.3-1 Time line of fungicide resistance in crop diseases for some major fungicide classes. 29

Table 4.3.3-2 Large-scale field studies on the effects on bees of seed treatments with neonicotinoids, not included in the review by Wood and Goulson (2017). 33

Table 4.3.3-3 Examples of pest resurgence or secondary pest development induced by the use of pesticides. 36

Table 4.3.4-1 Recent studies in the United Kingdom associating pesticide use with trends in terrestrial arthropod population abundance or biomass. 49

Table 4.3.5-1 Studies, covering at least five agricultural water bodies, which involved pesticide residue sampling and macroinvertebrate community sampling. Effects from urban and industrial land use were excluded. Studies reanalysing the same data are not considered independently. 51

Table 4.3.6-1 Global biodiversity reviews and status reports citing pesticide use as one of the drivers of biodiversity loss. 57

Table 4.4-1 Principal lines of evidence investigated for this report in order to identify human health effects of pesticide use. 64

Table 4.4-2 Poisoning severity score, as applied by the World Health Organization (WHO) and many national poison centres. 66

Table 4.4-3 Levels of under-reporting of acute pesticide poisoning (APP) cases, as quantified in different parts of the world. 69

Table 4.4-4 Average number of reported pesticide poisoning cases in countries with national coverage; averages calculated over the last three years reported (where available). 70

Table 4.4-5 Fraction of pesticide poisoning according to circumstances of exposure and severity of pesticide poisoning cases, calculated as average percentage of moderate, severe and fatal cases (values are average percentages over the last three years reported). 71

Table 4.4-6 Examples of large cohort studies addressing the health effects of pesticide exposure. 75

Table 4.4-7 Characteristics of the three major reviews of chronic adverse health effects of pesticides. 77

Table 4.4-8 Associations between pesticide exposure and diseases or other health outcomes, as concluded in two recent comprehensive reviews. 79

Table 4.4-14 Global estimates of self-poisoning and suicides (deaths) by pesticides. 82

Table 4.4-10 Factors found to increase or reduce occupational pesticide exposure and/or health effects: outcomes from selected reviews and medium- or large-scale studies. 83

Table 4.4-11 Examples of national pesticide residue monitoring programmes. 89

Table 4.4-12 Recent levels of exceedance of pesticide maximum residue limits (MRLs) in countries with regular pesticide residue monitoring programmes in food commodities. 92

Table 4.4-13 Recent regional or national dietary risk assessments of pesticide residues in food: assessments are of the risks of exposure to single pesticides and of combined exposure to multiple pesticides. 95

Table 4.4-14 Antimicrobials used as both pesticides and human medicines. 99

About

In December 2017, Resolution 4 of the 3rd Session of the United Nations Environment Assembly (UNEA 3) requested "the Executive Director to present a report on the environmental and health impacts of pesticides and fertilizers and ways of minimizing them, given the lack of data in that regard, in collaboration with the World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO) and other relevant organizations by the fifth session of the United Nations Environment Assembly". In response to this request, UNEP published a *Synthesis Report on the Environmental and Health Impacts of Pesticides and Fertilizers and Ways to Minimize Them*[1] in February 2022 (United Nations Environment Programme [UNEP] 2022).

The overall goal of the synthesis report is to provide the information base to enable other advocacy actions to be taken by stakeholders to minimize the adverse impacts of pesticides and fertilizers. Specific objectives of the synthesis report are to:

- Update understanding of current pesticide and fertilizer use practices;
- Present major environmental and health effects of pesticides and fertilizers, during their life cycle, and identify key knowledge gaps;
- Review current management practices, legislation and policies aimed at reducing risks in the context of the global chemicals, environmental and health agenda;
- Identify opportunities to minimize environmental and health impacts, including proven and innovative approaches.

This chapter on "Environmental and health effects of pesticide use" is the 4th in a series of 12 chapters that make up a comprehensive compilation of scientific information. The chapters were developed to both inform and further elaborate on the information provided in the synthesis report. Please note that the disclaimers and copyright from the synthesis report apply

1 The Synthesis report is available at https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

4 Environmental and health effects of pesticide use

Key findings for Chapters 4.3 (environmental effects) and 4.4 (health effects) are found at the beginning of each of those sections.

The annexes referred to in this chapter are provided as supporting information on the web page of the main report.[1]

4.1 Overview

Pesticides are by definition biologically active compounds. They are among the few types of chemicals that are purposely administered in the environment rather than being a by-product of other processes. Their use can pose a risk to humans and other non-target organisms.

This report is not intended to provide a comprehensive review of all the environmental and health risks posed by different groups pesticides. The use of pesticides will virtually always pose certain risks. The likelihood and importance of these risks may depend on, for example, the dose, the use situation, the exposed organisms or ecosystems, and the timing of exposure. Nevertheless, pesticides are widely used because their risks have been judged to be acceptable, although often on the condition that specific risk mitigation measures are applied.

In this chapter the environmental and health effects of pesticides are reviewed. The focus of the chapter is on pesticides' observed effects following their actual use. Pesticide toxicity, or potential environmental and health risks, are not extensively covered. These topics are reviewed in detail elsewhere (Krieger 2010; Brock *et al*. 2010; National Research Council 2013; Roberts and Reigart 2013).

Furthermore, extensive regulatory hazard and risk assessments of individual pesticides have been published by international entities such as the Joint Meeting on Pesticide Residues (JMPR) (FAO 2020) and the International Agency for Research on Cancer (IARC) (2020a), as well as by major national or regional regulatory authorities such as the European Food Safety Authority (EFSA) (2020a), the United States Environmental Protection Agency (US EPA) (2020a) and the Australian Pesticides and Veterinary Medicines Authority (APVMA) (2000), among others.

2 https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing

Chapter 4.2 briefly introduces the hazard and risk assessment of pesticides. Chapter 4.3 discusses the adverse environmental effects of pesticide use. Chapter 4.4 reviews its adverse health effects. Major knowledge gaps with regard to the environmental and health effects of pesticide use are presented at the end of these chapters.

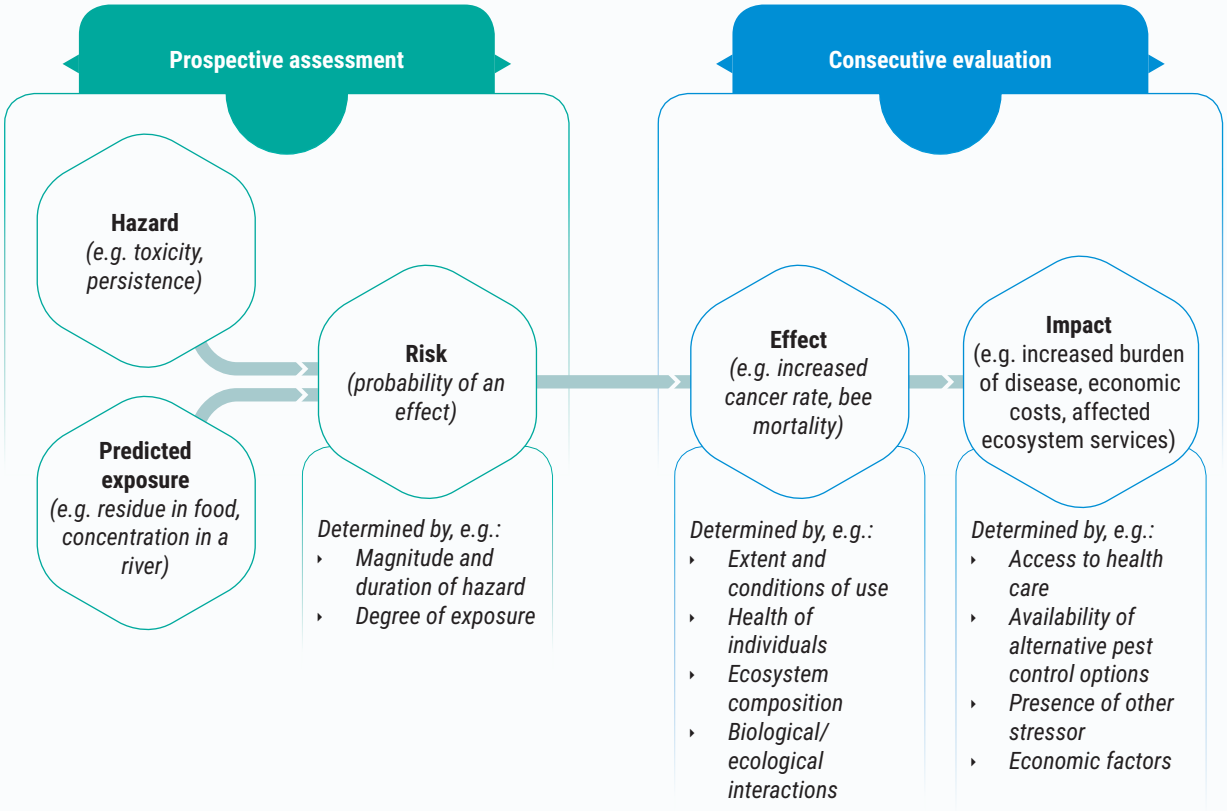
4.2 Pesticide hazards and risks

In many countries and regions, the risks of a pesticide to the environment and health are evaluated before it is authorized for use. This is done during the pesticide registration process. Only if a pesticide is judged not to pose unacceptable risks to the environment, or to human or animal health, under the conditions prevailing in a country/region will it be authorized.

In a risk assessment the hazard of a pesticide (e.g., its toxicity, persistence or bioaccumulative potential) is assessed against a predicted exposure that may occur if the product is used (e.g., determined by the application rate, chemical properties, use of personal protective equipment, environmental conditions) (Box 4.2-1).

In some countries, risks are also weighed against a product's expected benefits. The principle of risk assessment is the same for environmental and human health risks, but the methods used are different. These are prospective assessments,

Figure 4.2-1 Pesticide risk assessments are often prospective, in the sense that they are carried out before a pesticide is put on the market. Monitoring or studies of (adverse) effects of pesticide use are consecutive evaluations made after a pesticide is in use. As sufficient data are available, information about pesticide effects contributes to an evaluation of their impact on human health, the environment or society.



in the sense that they are carried out before a pesticide is authorized or re-authorized for use (Figure 4.2-1).

Only after a pesticide is marketed can the actual effects of the product be assessed under local conditions of use. This assessment may take place through regular monitoring, specific scientific studies, or feedback about incidents. As sufficient reliable information becomes available about the postulated (adverse) effects of the pesticide, its impact is evaluated (e.g., farmers' revenues, or gains/costs for society) (Box 4.2-1). These are referred to as consecutive evaluations (Figure 4.2-1).

For a new active ingredient or new pesticide formulation, prospective assessments provide the main information on risks because the pesticide has not yet been used. In the last few decades pesticide risk assessment methods for the environment and human and animal health have required more data. These methods have also become more precise, more locally specific, and often more complex. In addition, for compounds belonging to already assessed classes, the accumulated evidence related to the class serves as prior knowledge. A large number of environmental and health aspects are evaluated during the pesticide registration process (Box 4.2-2). Although there are limitations to the risk assessment process, pesticides are without doubt among the chemicals that are most thoroughly evaluated before being placed on the market.

Nevertheless, despite increasingly comprehensive prospective risk assessments, either unexpected or greater than expected environmental or health effects may be observed following the introduction of a pesticide (Boyd 2018; Storck, Karpouzas and Martin-Laurent 2017; Vijver *et al*. 2017). This is because not all the potential effects of pesticides are being evaluated in the commonly used risk assessments under current testing and assessment requirements, nor can all environmental conditions or situations of use be modelled in advance. Moreover, certain effects simply cannot be known at the time of the prospective assessment. The identification of effects that become known post-authorization of a pesticide is not necessarily limited to historical cases, but continues to occur up until the present (Table 4.2-1).

Post-registration monitoring and studies that complement prospective risk assessments are therefore important tools, especially given that data on environmental settings and human populations can only become available after the market authorization of a compound. Many pesticide registration authorities will

Box 4.2-1 Common hazard and risk assessment terminology applied in this report.

Hazard	Inherent property of a substance, agent or situation having the potential to cause undesirable consequences (e.g. properties that can cause adverse effects or damage to health, the environment or property). (Food and Agriculture Organization of the United Nations and World Health Organization [FAO and WHO] 2014).
Risk	Probability and severity of an adverse health or environmental effect occurring as a function of a hazard and the likelihood and the extent of exposure to a pesticide or fertilizer. (based on FAO and WHO 2014)
Effect	Change in the state or dynamics of an organism, system, or (sub)population caused by the exposure to an agent. (World Health Organization [WHO] 2004)
Adverse effect	Change in the morphology, physiology, growth, development, reproduction, or life span of an organism, system, or (sub)population that results in an impairment of functional capacity, an impairment of the capacity to compensate for additional stress, or an increase in susceptibility to other influences. (WHO 2004)
Exposure	Concentration or amount of a particular agent that reaches a target organism, system, or (sub)population in a specific frequency for a defined duration. (WHO 2004)
Impact	A durable change in the condition of people or their environment brought about by the (adverse) effect(s) of a pesticide or fertilizer. (based on Hearn & Buffardi 2016)

Table 4.2-1 Some examples of environmental or health effects of pesticides which had not been identified during prospective risk assessments, but became known after their use was authorized.

Pesticide	Effects identified post-registration	Period when the effects observed	Selected reference (also see Chapters 4.3 and 4.4)
DDT	Population declines in birds of prey due to eggshell thinning	1950s-early 1970s	Peakall (1993)
Highly acutely toxic pesticides	High rate of fatalities from self-poisoning	1960s-present	Karunarathne *et al*. (2019)
Various pesticides	Population declines of amphibians	2000s-present	European Food Safety Authority (EFSA) (2018a)
Organophosphate pesticides	Prenatal exposure resulting in high risk of cognitive/behavioural deficits and neurodevelopmental disorders in children	2000s-present	Hertz-Picciotto *et al*. (2018)
Neonicotinoids	Possible declines of wild bee populations	2000s-present	Hladik, Main and Goulson (2018)

amend or cancel registrations of a product if consecutive evaluations show that use of a pesticide under local conditions results in unacceptable risks. Pesticide registrations are often subject to a periodic review process for re-authorization, which presents an opportunity to review, and take action on, the outcomes of post-registration monitoring and studies.

In this chapter the (adverse) environmental and health impacts of pesticides are reviewed, as reported through observational studies, large-scale field experiments and monitoring. These are environmental and health effects observed following actual use of and exposure to such substances. However, a major limitation of an assessment of actual effects is that they depend

Box 4.2-2 Many risk assessments are generally conducted, as part of the pesticide registration process, to assess whether a pesticide may pose unacceptable risks to human or animal health or the environment.

Human health	
Type of risk	**Questions being assessed**
Dietary	Will residues of the pesticide in agricultural commodities or drinking water pose unacceptable risks to consumers?
Occupational	Does the handling and application of the pesticide pose unacceptable risks to the user?
Residential	Will the use of the pesticide pose unacceptable risks to persons living in or close to the treated areas?
Bystander	Will the use of the pesticide pose unacceptable risks to persons?

Animal health and the environment	
Type of risk	**Questions being assessed**
Surface waters	Will the use of the pesticide lead to concentrations in e.g. rivers, lakes, estuaries that pose unacceptable risks to fish, crustaceans, algae and other aquatic organisms?
Soils	Will the use of the pesticide lead to concentrations in soils that pose unacceptable risks to earthworms, springtails and other soil organisms, or adversely affect soil microbial processes such as nitrogen cycling?
Beneficial arthropods	Will the use of the pesticide lead to unacceptable effects on pollinators, natural enemies of pests or other beneficial arthropods?
Terrestrial vertebrates	Will the use of the pesticide pose unacceptable risks to birds, mammals, reptiles or other terrestrial vertebrates?
Plants	Will the use of the pesticide lead to unacceptable affect to non-target vascular plants?

very much on local capacity for monitoring, which is weak in most parts of the world. As a result, reported effects are almost always incomplete.

The assessment of actual effects is also limited by the difficulty of causal inference (i.e., linking the adverse effects to an individual or class of pesticides). Furthermore, many chronic ecosystem/population effects or health effects only materialize after many years of use of a substance. Reports of the effects of more recently introduced pesticides will therefore be limited.

Finally, the population studies that provide valuable information on the proposed associations between pesticide exposure and environmental or health effects exhibit variable precision and validity.

Quantitative information on the impact of (adverse) effect(s) of pesticides – for example, on the global burden of disease, biodiversity and ecosystem health, and economic costs and benefits – are reviewed in Chapter 6.

4.3 Adverse environmental effects of pesticide use

4.3.1 Overview

Pesticides are applied in the environment on purpose. They will therefore, almost by definition, pose risks to non-target organisms. Depending on pesticide use patterns, the toxicity of the pesticide, the conditions of exposure of non-target organisms, and the type of agro-ecosystem exposed, environmental risks will range from very high to virtually absent.

The focus of this chapter is on observed or likely environmental effects of pesticides under current conditions of use. The review in the chapter is therefore essentially a consecutive evaluation of pesticide impact after pesticide products have been authorized (or registered) for use, as outlined in Figure 4.2-1.

The chapter is based on recent existing global or regional reviews, whenever these were available. If no recent reviews were available for a given topic, large-scale national studies were identified which could contribute to an understanding of pesticides' environmental impact. Literature reviews were initially conducted for relevant scientific publications published in the period 2010-2020, although older studies were sometimes included if they were particularly relevant.

Different lines of evidence regarding the environmental effects of pesticides were subsequently evaluated in this report (Table 4.3.1-1):

Table 4.3.1-1 Principal lines of evidence investigated for this report to identify the environmental effects of pesticide use.

Type of evidence	**Pesticides in the environment**	Pesticide effects			**Pesticide risks**
Type of study	Monitoring of environmental concentrations	Monitoring and field studies of effects on sustainability	Monitoring of environmental effects	Field studies of environmental effects	Measured environmental concentrations
Type of outcomes	Magnitude of residues Geographical distribution Time trends	Presence/ absence of effects Time trends	Presence/absence of effects Time trends	Presence/absence of effects Dose-response data	Risks under actual pesticide use conditions (consecutive assessment)

Pesticides end up in different compartments of the environment, both after their intended and authorized use and following misuse or accidents. Residues of pesticides and their metabolites are therefore found in air, surface waters, groundwater, soil and biota. Regular monitoring of pesticide residues in the environment is mainly limited to high income countries. There are substantial information gaps in large parts of Africa, Asia and Latin America Ad hoc studies of pesticide residues in the environment, as well as their behaviour and fate, are conducted more widely, although data from low and lower-middle income countries remain scarce (Stehle and Schulz 2015). [Chapter 4.3.2].

Wherever they have been measured in surface waters, pesticides have been found to be ubiquitous. In many cases the measured concentrations have exceeded national surface water standards and posed risks to aquatic organisms (European Union 2013). In most cases where time trends were reported, pesticide concentrations did not show downward trends over time. This holds true for both legacy organochlorine pesticides (OCPs) and current use pesticides (CUPs) (Stone, Gilliom and Ryberg 2014; Nesser *et al*. 2016; Wolfram *et al*. 2018; Bruce-Vanderpuije *et al*. 2019). [Chapter 4.3.2]

Atmospheric pesticide concentrations are less well documented than concentrations in other environmental compartments. A decrease in atmospheric concentrations of many OCPs has been observed globally. Nevertheless, OCPs are still detected in air almost everywhere in the world, often decades after their use has been ended (Shunthirasingham *et al*. 2010; UNEP 2017a; Rauert *et al*. 2018). Data on CUPs are more limited, and trends in their atmospheric concentrations have not been systematically reviewed. [Chapter 4.3.2]

Pesticide residues appear to be omnipresent in agricultural soils, but are also detected in non-treated soils of organic production systems (Silva *et al*. 2019; Pelosi *et al*. (2021). OCP residues have been found in soils across the world, with concentrations often not showing significant declines over time (Camenzuli, Scheringer and Hungerbühler 2016). Most reports of soil concentrations of current use pesticides were from Europe and China, where the large majority of agricultural soils contained pesticide residues. Certain herbicides and fungicides, as well as neonicotinoid insecticides, were most often encountered (Annex 4.3-3). [Chapter 4.3.2]

Data on pesticide concentrations in groundwater and drinking water are generally scarce for many parts of the world and for many pesticides. Legacy and current use pesticides, and (especially) their transformation products, have frequently been detected at concentrations above their legal limits in groundwater and drinking water (Postigo and Barceló 2015; Pirsaheb *et al*. 2017; Pietrzak *et al*. 2019). Globally, herbicides are most often detected in groundwater (Close and Skinner 2012; McManus *et al*. 2014; Lopez *et al*. 2015; Karki *et al*. 2020). [Chapter 4.3.2] While it is difficult to establish trends over time, pesticides may remain present in groundwater for decades after their authorization has been discontinued (Lopez *et al*. 2015).

Pesticide use may have a direct or indirect impact on the sustainability of agricultural production or the effectiveness of disease vector control. As a result, rather than increasing production or reducing disease vector populations, pesticide use may have a negative mid- and long-term impact on agriculture and public health.

The development of resistance against pesticides in arthropods, diseases, weeds and rodents may lead to failure to control them and to subsequent increases in crop losses or disease prevalence. It also often results in an increase in pesticide use, with associated adverse environmental and health effects (FAO 2012). [Chapter 4.3.3]

Despite efforts to manage resistance, and some clear successes in slowing down its development (e.g., in Bt transgenic crops), the overall trend during the last decades has been a continuous increase in field-evolved pesticide resistance in arthropods, weeds and diseases (Gould, Brown and Kuzma 2018; Gould *et al*. 2018).

Acute mortality of honeybees due to pesticide applications has been reported regularly in the past. However, it appears to be declining

in countries with effective regulation or enforcement of mitigation measures. Incidents of insecticide-associated honeybee mortality may occur more often in other countries, but monitoring data to validate this are lacking (Kovács-Hostyánszki *et al*. 2016). [Chapter 4.3.3]

It remains unclear to what extent sublethal effects of pesticide exposure, which are recorded for individual insects, affect colonies and populations of managed bees and wild pollinators in the field, especially over the longer term. There are indications, however, that wild bees may be more affected than managed honeybee colonies (Kovács-Hostyánszki *et al*. 2016; Woodcock *et al*. 2016; Wood and Goulson 2017).

The use of insecticides, as well as other groups of pesticides, often adversely affects the abundance of natural enemies of pests (Cloyd 2012; Roubos, Rodriguez-Saona and Isaacs 2014). This may have an impact on natural pest control and can lead to pest resurgence or secondary pest development (Dutcher 2007; Wu et al. 2020).

Despite changes in pesticide chemistry, and an increased focus on integrated pest management, both pest resurgence and secondary pest development due to pesticide applications persist (Roubos, Rodriguez-Saona and Isaacs 2014; Hill, MacFadyen and Nash 2017). However, it is unclear whether resurgence and secondary pest development are less frequent today than they were in the past. [Chapter 4.3.3]

Pesticide applications in the field have led to variable effects on soil microbial communities and activity. There is only limited evidence that the observed effects have led to significant and long-lasting decreases in soil functions (Imfeld and Vuilleumier 2012; FAO and Intergovernmental Technical Panel on Soils [ITPS] 2017). [Chapter 4.3.4]

No review was available of post-registration monitoring of earthworm populations following real pesticide applications at recommended rates. However, several studies from Europe suggest that especially chronic risks to earthworms may be higher than previously estimated. Pesticide effects on soil arthropods are less well studied than on earthworms (Pelosi *et al*. 2014).

Despite the importance of soil quality for agricultural production, relatively little monitoring and few large scale field studies have been conducted to assess the impact of current use pesticides on soil organisms and processes.

Pesticides may have a direct (lethal or sublethal) toxic effects on birds, or can act indirectly through food depletion or habitat alteration (EFSA 2009; Amaral *et al*. 2012a; Mineau 2013a; EFSA 2018a; Stanton, Morrissey and Clark 2018). [Chapter 4.3.4]

Although certain current use pesticides can have toxic effects on birds at recommended application rates, there does not appear to be much evidence of significant population reductions arising from direct effects of pesticides (Jahn *et al*. 2014; Tassin de Montaigu and Goulson 2020), with the possible exception of the use of anticoagulant rodenticides and deliberate wildlife poisoning with pesticides (López-Perea and Mateo 2018; Nakayama *et al*. 2019).

There are indications, however, that currently used pesticides may be one of the drivers of observed bird declines and, in some cases, may be the leading cause, likely through indirect effects (Jahn *et al*. 2014; Coates *et al*. 2017; Brain and Anderson 2019; Spiller and Dettmers 2019). Whether or not pesticides are the principal factor in bird population reductions is likely to depend on the type of pesticide applied, the species concerned, and pesticide application parameters such as dose and frequency of treatments (Bouvier et al. 2011; Mineau and Whiteside 2013; Emmerson *et al*. 2016; Stanton, Morrissey and Clark 2018; Miao and Khanna 2020).

Pesticides are likely to affect amphibian survival and health at environmentally relevant concentrations (Egea-Serrano *et al*. 2012; Baker, Bancroft and Garcia 2013). However, important data gaps still exist, particularly with regard to studies on non-anuran amphibians. Furthermore, it is still very difficult to link adverse effects in amphibians to individual pesticides, partly due to the virtual absence of in-depth field monitoring

studies and the presence of other stressors (Bishop *et al*. 2012; North *et al*. 2015; Orton and Tyler 2015).

Reptiles are not yet well studied with regard to the acute and chronic effects caused by pesticides or the different (qualitative and quantitative) ways in which reptiles are exposed to them (Köhler and Triebskorn 2013; EFSA 2018a). The limited information available indicates that adverse (sub)lethal effects may occur at realistic field exposure levels following the application of certain pesticides that are currently authorized for use (Wagner *et al*. 2015; Mingo, Lötters and Wagner 2016; Mingo, Lötters and Wagner 2017). In the absence of validated and robust risk assessment methods, pesticide registration procedures may not fully capture specific risks to this group of non-target vertebrates [Chapter 4.3.4].

Field monitoring of the effects of real pesticide use on aquatic organisms is not frequently conducted, nor are field studies evaluating the effects of pesticides on aquatic ecosystem structure and functioning despite an increase in the number of these studies conducted in recent years (Schäfer 2019; Zubrod *et al*. 2019; Rosic *et al*. 2020; Schepker *et al*. 2020). Nevertheless, available studies generally indicate high pesticide risks to aquatic organisms under current agricultural practices, especially in freshwater systems (Beketov *et al*. 2013; Sánchez-Bayo and Wyckhuys 2019; Schäfer 2019; Ito *et al*. 2020). [Chapter 4.3.5]

In major global biodiversity status reports, pesticides are often listed as a key driver of biodiversity loss in agricultural and natural ecosystems (FAO 2019; IPBES 2019; FAO *et al*. 2020; Sud 2020). However, the number of studies that are able to directly link pesticide use with adverse effects on biodiversity parameters, such as species richness, are relatively rare. Moreover, virtually all studies have been conducted in North America and Europe. There is an almost complete absence of data from other parts of the world (Annex 4.3-6). [Chapter 4.3.6]

Nevertheless, where large-scale studies have been conducted, pesticide use has frequently been associated with adverse effects on biodiversity (Brittain *et al*. 2010; Beketov *et al*. 2013; Emmerson *et al*. 2016; Chiron *et al*. 2014; Forister *et al*. 2016; Sattler *et al*. 2020).

Substantial knowledge gaps have been identified with regard to the effects of pesticide use on the environment. They include the need to conduct more systematic reviews and meta-analyses of existing scientific data, particularly on topics of specific interest in regulatory decision-making (Group of Chief Scientific Advisors 2018). [Chapter 4.3.7]

Despite significant improvements in prospective regulatory risk assessment, unexpected environmental impacts are frequently identified (Boyd 2018; Group of Chief Scientific Advisors 2018; Brühl and Zaller 2019; Schäfer *et al*. 2019; Topping, Aldrich and Berny 2020). Some principal directions for improving pesticide environmental risk assessment have been proposed. [Chapter 4.3.7]

Due to the limitations in the current prospective risk assessment procedures discussed, and the lack of systematic post-registration monitoring, significant gaps in knowledge about adverse effects of current pesticide use on the environment persist. Strengthening post-registration environmental monitoring of pesticide concentrations and effects should therefore have the highest priority (Milner and Boyd 2017; Vijver *et al*. 2017; Group of Chief Scientific Advisors 2018; Rico *et al*. 2020; Topping, Aldrich and Berny 2020).

This chapter does not provide an in-depth review of approaches, methods and procedures for the prospective environmental risk assessment of pesticides. However, possible difficulties of (and gaps in) environmental risk assessment have been indicated where they might explain the observed environmental effects and consecutive risks of actual pesticide use. In order to better situate possible gaps in current risk assessment, the general principles of prospective environmental risk assessment, as conducted by major regulators, are summarized in Box 4.3-1.

Box 4.3-1 Principles of environmental risk assessment of pesticides.

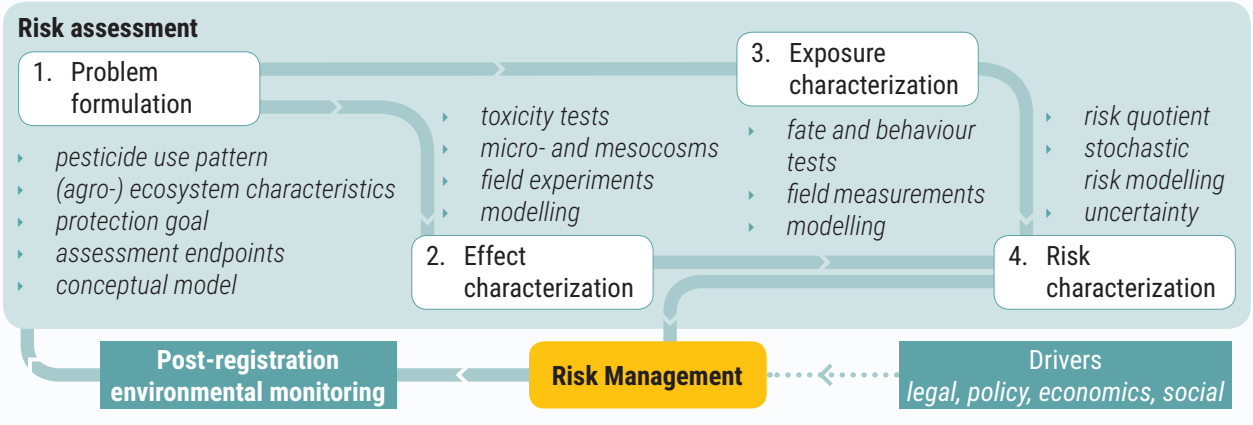
Regulatory (prospective and retrospective) environmental risk assessment of pesticides is conducted based on similar principles for different non-target organisms and in different parts of the world. It tends to be built upon the following elements, as illustrated in the figure below (adapted from Babut *et al*. 2013; Boivin and Poulson 2017; United States Environmental Protection Agency [US EPA] 2020).

1. **Problem formulation**
   The problem formulation outlines the objectives of the risk assessment and defines the pesticide use pattern(s), (agro-)ecosystem characteristics, protection goals (what needs to be protected from the pesticide, where, and how strictly), assessment endpoints, and a conceptual model that describes the main expected relationships between the pesticide and the assessment endpoint(s). Generally, an environmental risk assessment is conducted for an individual pesticide and its relevant metabolites.
2. **Effects characterization**
   The effects characterization describes the types of effects a pesticide may produce in an organism and how those effects depend on the pesticide exposure levels. It is generally based on toxicity studies conducted in the laboratory (bioassays), in micro- and mesocosms, or in the field. Effect models of biological, toxicological and ecological processes have received increasing attention in recent years. The outcomes of the effects characterization are one or more assessment endpoints, generally a predicted no effect concentrations (PNEC) or a similar endpoint.

   Since laboratory toxicity tests tend to be conducted with single species, while the protection goal is often an ecological community, extrapolations are made from one to more species and from the laboratory to the field. This can be done through modelling, using species sensitivity distribution, and/or applying an uncertainty factor (also referred to as a safety or assessment factor).

   If an uncertainty factor is integrated into the effects characterization phase, the assessment endpoint is referred to as, for example, a regulatory acceptable concentration (RAC) or level of concern (LoC). However, the safety factor may also be integrated into the risk characterization phase.
3. **Exposure characterization**
   The exposure characterization estimates the potential exposure of environmental components (e.g. surface waters, soil) where non-target organisms are present. This characterization includes information about the estimated quantity, frequency and duration of the exposure of an organism to a pesticide.

   Laboratory and field-derived data of pesticide fate and behaviour in the environment are at the basis of exposure characterization. Modelling tends to be used to generate exposure assessment endpoints, such as a predicted environmental concentration (PEC) or an estimated environmental concentration (EEC). A PEC is calculated on the basis of the intended use of the pesticide (e.g. application rate, frequency and timing, type of crop or other target), pesticide properties (e.g. solubility, vapour pressure, degradation rates), and environmental conditions or scenarios (e.g. wind speed, slope, weather).
4. **Risk characterization**
   In the risk characterization phase, exposure and ecological effects endpoints are integrated into a risk estimation or risk quotient. If uncertainty factors have not been taken into account in the effects characterization phase, they can be incorporated into the risk estimation. Either way, the risk of the pesticide is generally considered acceptable if the predicted level of exposure is lower than the regulatory acceptable concentration or the level of concern. The integrated risk characterization therefore includes the assumptions, uncertainties, and strengths and limitations of the different risk assessment phases.



On the basis of the results of the risk assessment, risk management decisions about the pesticide are taken, e.g. authorization of the pesticide with or without risk mitigation measures, restricted use registration, or denial of authorization. Risk management decisions are based not only on the scientific risk assessment, but also on legal, policy, economic and social considerations, which are generally country-specific.

In many countries a tiered approach to risk assessment is followed. As a first tier, the results of standard toxicity tests with a limited number of species are compared to a reasonable worst-case PEC. If the first-tier assessment does not indicate an unacceptable risk, further assessments are generally not required. If, on the other hand, a risk is identified in the first tier, more realistic toxicity data and/or exposure estimates are needed. Higher tiers better reflect processes and characteristics of natural ecosystems and are tailored based on the outcomes of the lower tiers.

Regulatory pesticide risk assessments have become increasingly complex, with the aim of addressing risks for a wider range of non-target organisms, being more ecologically realistic, and reducing uncertainty. A large number of guidance documents and models have been published covering key environmental compartments and non-target organisms:

Selected regulatory environmental risk assessment guidance

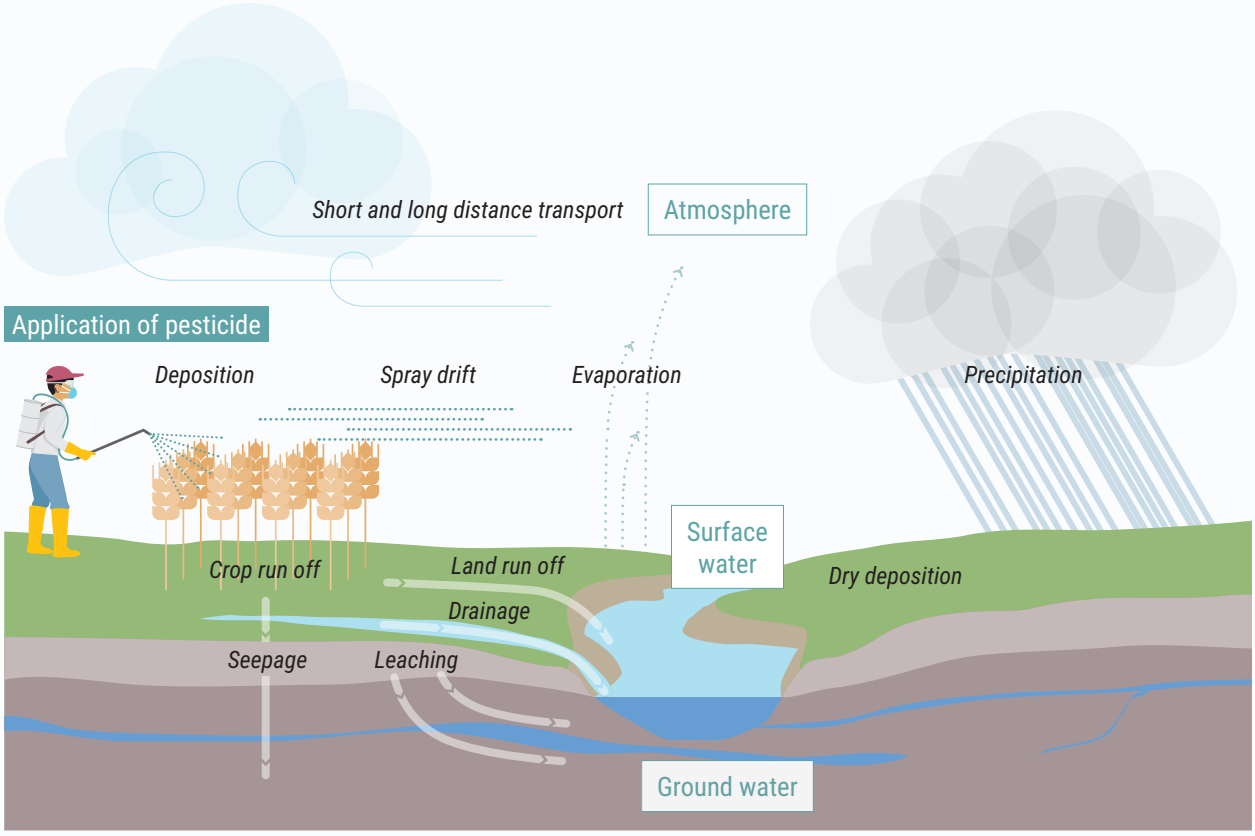
Protection goal	Region	Guidance	Reference
General	Australia	*Risk Assessment Manual – Environment*	Australian Pesticides and Veterinary Medicines Authority (APVMA) 2019
	Brazil	*Manual para requerimento de avaliação ambiental: agrotóxico e afins*	Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis (IBAMA) 2009
	China	*Environmental Risk assessment Handbook for Pesticide Registration in China*	Peeters *et al*. 2014
	United States	*Generic Ecological Assessment Endpoints (GEAEs) for Ecological Risk Assessment*	US EPA 2016
	United States	*Ecological risk assessment for pesticides: Technical overview*	US EPA 2020
Pollinators	EU	*Guidance on the risk assessment of plant protection products on bees (Apis mellifera, Bombus spp. and solitary bees)*	European Food Safety Authority (EFSA) 2013a)
	North America	*Guidance for Assessing Pesticide Risks to Bees*	US EPA, Heath Canada Pest Management Agency, and California Department of Pesticide Regulation 2014
Aquatic organisms	EU	*Guidance on tiered risk assessment for plant protection products for aquatic organisms in edge-of-field surface waters*	EFSA 2013b
Terrestrial vertebrates	EU	*Guidance document on risk assessment for birds and mammals*	EFSA 2009

4.3.2 Pesticide concentrations in the environment

Pesticides will end up in different compartments of the environment after their intended and authorized use, or following misuse or accidents. Residues of pesticides and their metabolites are found in air, surface waters, groundwater, soil and biota (Figure 4.3.2-1). The degree of contamination of the environment by pesticides provides an important indication of their risks to the environment and health. Therefore, increasingly, pesticide residues are monitored in the environment on a more or less regular basis.

In this chapter global and regional reviews concerning pesticide concentrations in the environment are evaluated and their potential

Figure 4.3.2-1 Pesticides can contaminate surface water and groundwater through many different routes.



risks described. Reviews published since 2010 were compiled. Recent large-scale national studies not included in such global or regional reviews were also identified, especially if they included time trends.

Key questions addressed are: What geographical differences exist in observed pesticide residues in the environment? What time trends can be identified, and have pesticide concentrations been increasing or decreasing over time? Are certain (groups of) pesticides found more frequently in the environment than others? And what are the environmental risks of these concentrations (often assessed through exceedances of regulatory acceptable concentrations)? Data gaps are also identified. A distinction is made between persistent organic pollutant (POP) pesticides, which are internationally regulated under the Stockholm Convention on Persistent Organic Pollutants and are generally not used anymore (i.e., legacy pesticides), and other pesticides generally in current use (current use pesticides).

Pesticides in surface waters

Surface waters, such as rivers, lakes, streams and ponds, can be exposed to pesticides in many ways. These include direct application to water bodies, drift of droplets during spraying, drainage of irrigation water, surface run-off during rain events, leaching through the soil, and deposition of evaporated pesticides (Figure 4.3.2-1 above). Pesticides can also contaminate water accidentally, e.g., after spills or as a result of inappropriate handling such as washing pesticide application equipment close to waterways. Finally, surface water may be exposed through illegal use of pesticides (e.g., in fishing).

Monitoring of pesticide concentrations in surface water bodies has been conducted in certain regions of the world for several decades through systematic and ad hoc monitoring or event-triggered sampling. For example, the United States Geological Survey's National Water Quality Assessment (United States Geological Survey 2020) has monitored pollutants such as pesticides

in streams, rivers and groundwater since 1991. The Swedish National Environmental Monitoring Programme of Pesticides in Surface Waters has conducted analyses in surface waters of almost all pesticides registered in Sweden for more than 15 years (Boye et al. 2019). In the Netherlands, although water quality monitoring had been conducted for many years, a harmonized national monitoring network for pesticides in surface waters was only established in 2014 to evaluate implementation of the national plant protection policy (de Weert *et al*. 2014). The European Environment Agency hosts the Waterbase – Water Quality database, which contains Europe-wide data on pesticide concentrations in rivers and lakes (European Environment Agency [EEA] 2019).

In many other parts of the world, especially Africa, Asia and Latin America, very limited if any surface water monitoring is carried out. Stehle and Schulz (2015) estimated that for about 90 per cent of high-intensity agricultural areas in the world, no scientific investigations of insecticide surface water exposure exist (Figure 4.3.2-2).

Monitoring of pesticide concentrations in surface water is constrained by the fact that pesticides tend to be applied only once or a few times per growing season. They often dissipate rapidly from the water column because of adsorption to sediment and organic matter in the water, degradation, and water flow (the latter in rivers and streams). This leads to discrete and often short-term exposure of water bodies, between which there may be long periods without exposure (Stehle, Knäbel and Schulz 2013). Nevertheless, pesticides may be highly toxic to aquatic life, so that even short-term exposure peaks can lead to significant adverse effects. It has been estimated that water monitoring based on fixed intervals, even if technically well conducted, would still miss 99 per cent or more insecticide exposure events that could cause adverse effects (Stehle, Knäbel and Schulz 2013). Monitoring programmes are thus likely to (greatly) underestimate concentrations of pesticides in surface waters. The fact that no pesticides were measured in a sample does not necessarily mean they were not present in the (recent) past or will not be found in the (near) future.

Global reviews

Only three recent global reviews of pesticides in surface water were identified (Annex 4.3-1)[3].

No global reviews of POP pesticides in surface water were available. The Global Monitoring Programme of the Stockholm Convention does not assess POP pesticides in surface water bodies.

Stehle and Schulz (2015) and Stehle, Bub and Schulz (2018) conducted the largest global review of insecticide surface water concentrations so far, covering 838 peer-reviewed studies, at >2,500 sites in 73 countries. They reported that there is a complete lack of scientific monitoring data for ~90 per cent of global cropland (Figure 4.3.2-2).

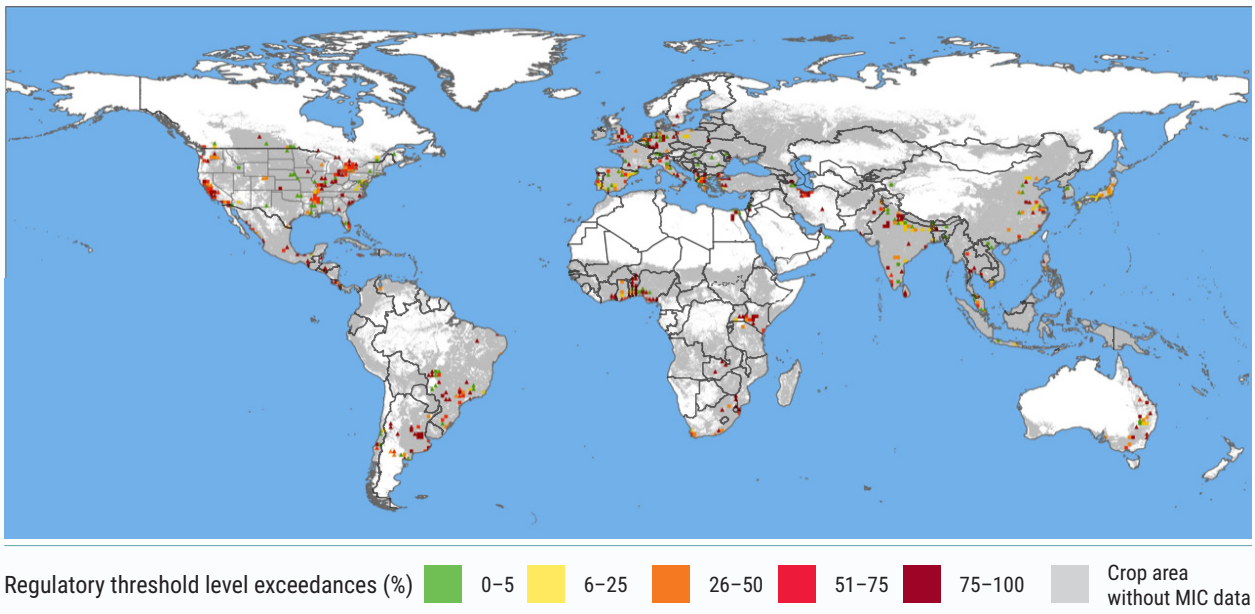
In 97 per cent of samples no insecticides were measured. However, of the remaining 11,300 samples, 52.4 per cent (5,915 cases; 68.5 per cent of sites) exceeded the legally accepted regulatory threshold levels for either surface water or sediments. Median concentrations of neonicotinoids, although limited in number, exceeded those of organochlorines and organophosphates by a factor of about 3 and those of pyrethroids by a factor 10. While most of the available insecticide monitoring data were from North America, Asia and Europe, the highest insecticide concentrations were detected in Africa, Asia and South America.

Recent global reviews were also available for neonicotinoids (Morrissey *et al*. 2015; Sánchez-Bayo, Goka and Hayasaka 2016) and for pyrethroids (Tang *et al*. 2018). Insecticides from these groups were widely found in surface water across all global regions. Average concentrations of neonicotinoids exceeded ecological thresholds in 74 per cent of cases. Sanchez-Bayo, Goka and Hayasaka (2016) found that neonicotinoid concentrations in surface water had increased during the previous 15 years.

In a global review of occurrences of pesticides in surface water published between 2012 and

3 **Annex 4.3-1 is found at:** https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing

Figure 4.3.2-2 Global crop area and the distribution of regulatory threshold level exceedance rates for reported measured insecticide concentrations (MIC) aggregated in 1° grid cells. Stehle and Schulz (2015).



2019, de Souza *et al*. (2020) found that the most frequently reported pesticides were the herbicides atrazine and its metabolites, as well as metolachlor, the insecticide chlorpyrifos, and the fungicides carbendazim and tebuconazole.

Regional and national reviews

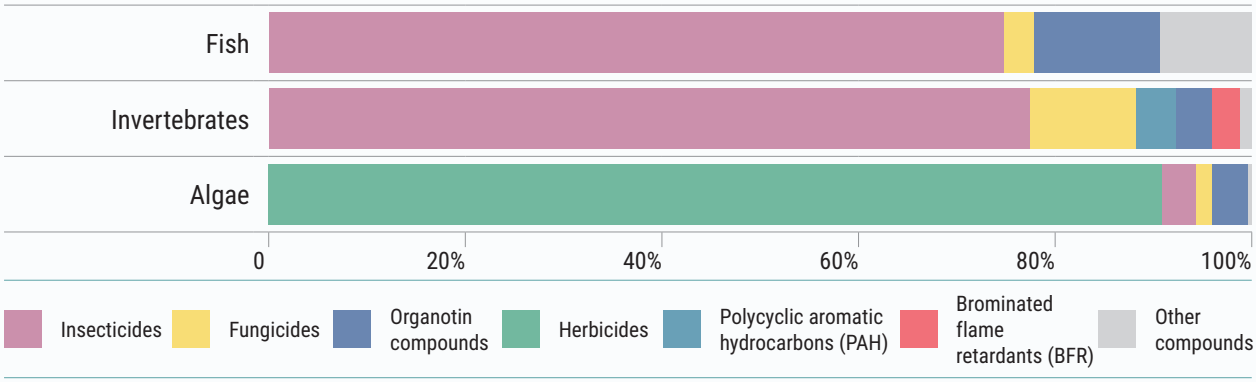
The presence of organochlorine pesticides (OCPs) in surface water has been reviewed for Caribbean and Pacific marine environments (Menzies *et al*. 2013) and freshwater systems in Africa (Taiwo 2019) and South Asia (Ali *et al*. 2014) (Annex 4.3-1). OCPs were encountered in almost all samples taken, even from locations far from human habitation and when the use of such pesticides had been prohibited for many years. Concentrations in freshwater bodies in Africa and Asia very often exceeded quality standards for surface water applied in Europe (European Union 2013).

The ubiquitous presence of OCPs in aquatic environments observed in these regional reviews was confirmed by recent national studies and reviews in surface water and sediments in Africa, for example Egypt (Dahshan *et al*. 2016), Ethiopia (Dirbaba *et al*. 2018), Ghana (Bruce-Vanderpuije *et al*. 2019), South Africa (Ansara-Ross *et al*. 2012), Sudan (Nesser *et al*. 2016) and Tanzania (Elibariki and Maguta 2017), and in Asia, for example China (Grung *et al*. 2015; Li, Li and Liu 2015) and India (Yadav *et al*. 2015).

Clear trends over time have generally not been established, with a reduction in OCP detections implied for the Nile in Sudan (Nesser *et al*. 2016) and significant increases in concentrations found in Ghana (Bruce-Vanderpuije *et al*. 2019). The studies reviewed did not provide indications that concentrations of POP pesticides in surface water showed downward trends in Africa and Asia.

Malaj *et al*. (2014) reviewed the presence and risks of organic chemicals at more than 4,000 sampling locations in freshwater ecosystems in Europe. The chemical risk per river basin was calculated by comparing reported pesticide concentrations with an acute and a chronic risk threshold for each organism group evaluated (fish, invertebrates and algae). Of all the monitoring sites, 14 per cent were likely to be acutely affected by organic chemicals, and 42 per cent to be affected chronically, for at least one organism group. Of the 223 chemicals monitored, pesticides were among the major contributors to chemical risk. They were responsible for 81 per cent, 87 per cent, and 96 per cent of observed exceedances of acute

Figure 4.3.2-3 Proportion of sampling sites in European freshwater ecosystems showing exceedance of acute regulatory thresholds by different chemical groups. Malaj *et al.* (2014).



regulatory thresholds related to fish, invertebrates and algae, respectively (Figure 4.3.2-3).

More recently, Mohaupt *et al*. (2020) evaluated data from the European Environment Agency's Waterbase – Water Quality database on the presence of 180 pesticides in surface water reported by 34 European countries between 2007 and 2017. They found that 5-15 per cent of samples exceeded the environmental quality standards for herbicides and 3-8 per cent exceeded those for insecticides; exceedances of fungicides were less prevalent.

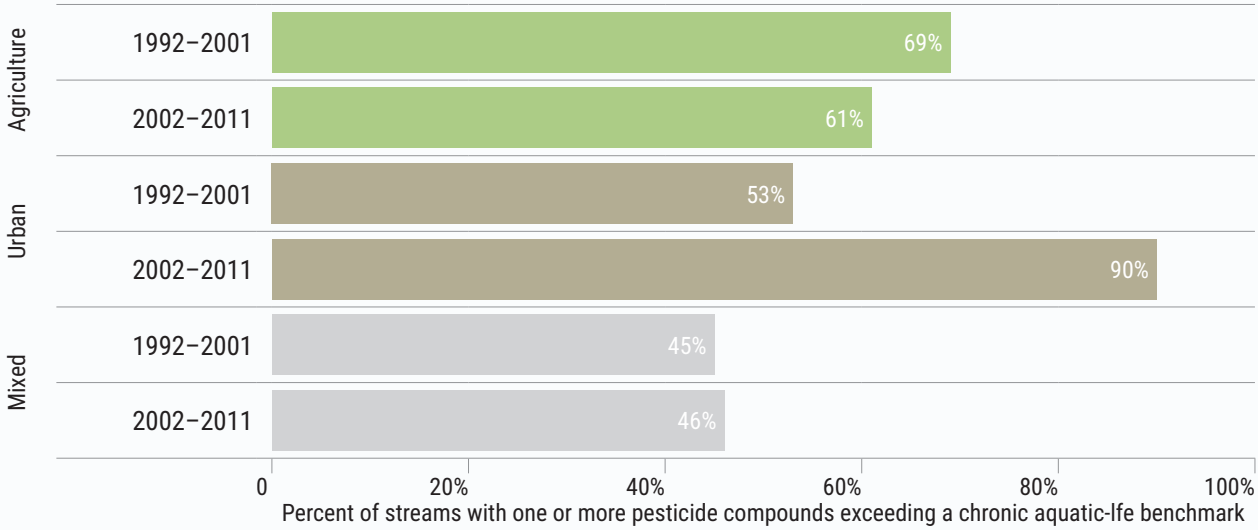
K'oroje *et al*. (2020) reviewed the literature regarding the occurrence of pesticides in Africa. They found that both POPs and current use pesticides are frequently reported in surface water on the continent at concentrations in the same order of magnitude as those found in Brazil, China and India.

Residues of current use pesticides in surface water were also recently assessed in national studies such as those in Brazil (Albuquerque *et al*. 2016), China (Chen *et al*. 2019), Hungary (Székács, Mörtl and Darvas 2015), Italy (Meffe and de Bustamente 2014), Japan (Derbalah et al. 2019), Romania (Schreiner *et al*. 2021), Switzerland (Knauer 2016) and the United States (Stone, Gilliom and Ryberg 2014; Hladik and Kolpin 2016; Wolfram *et al*. 2018), among other countries (Annex 4.3-1). While many studies on OCPs were conducted in Africa and Asia, current use pesticide concentrations were monitored more frequently in Europe and the United States.

Current use pesticides were frequently found in surface water at concentrations which could result in adverse effects on aquatic organisms. For example, Albuquerque *et al*. (2016) reviewed all published studies on pesticide concentrations in Brazilian freshwater systems and identified a potential risk to aquatic life for 59 per cent of the pesticides on which they had data. Chen *et al*. (2019), when monitoring neonicotinoids in river water in Eastern China, concluded that 27 per cent exceeded thresholds for acute and 84 per cent of chronic ecological risks. Meffe and de Bustamente (2014) found that 54 per cent of maximum pesticide concentrations measured in Italian surface water were above environmental quality standards. Wolfram *et al*. (2018) concluded, when reviewing 259 studies that covered 644 sampling sites in surface water in the United States, that 49 per cent of measured pesticide concentrations exceeded regulatory threshold levels. The only recent study which found that most pesticide concentrations in surface water did not appear to pose high risks was in Switzerland (Knauer 2016), where it was concluded that 95 per cent of measured concentrations were below national regulatory acceptable concentration.

Trends over time in water concentrations of current use pesticides were assessed in the United States. Both Stone, Gilliom and Ryberg (2014) and Wolfram *et al*. (2018) concluded that risks

Figure 4.3.2-4 The proportion of streams in the United States with one or more pesticides that exceeded a chronic aquatic life benchmark was very similar in the periods 1991-2001 and 2002-2011 for agricultural and mixed land use streams, but much greater in the period 2002-2011 for urban streams. Wolfram *et al.* (2018).



for aquatic organisms had not declined during the previous five decades (Figure 4.3.2-4). In the Netherlands, on the other hand, the number of exceedances of national water quality criteria fell between 2013 and 2018 by 15 per cent for chronic exposure and 30 per cent for acute exposure (Netherlands Environmental Assessment Agency 2019).

Pesticides in air

Contamination of air by pesticides may occur through spray drift, volatilization from soils, plants and surface waters, and wind erosion of soil particles containing adsorbed pesticides (FOCUS 2008). Taking into account all these processes, the fraction of pesticides emitted to ambient air may be more than 30 per cent of the applied dose (Coscollà and Yusà 2016).

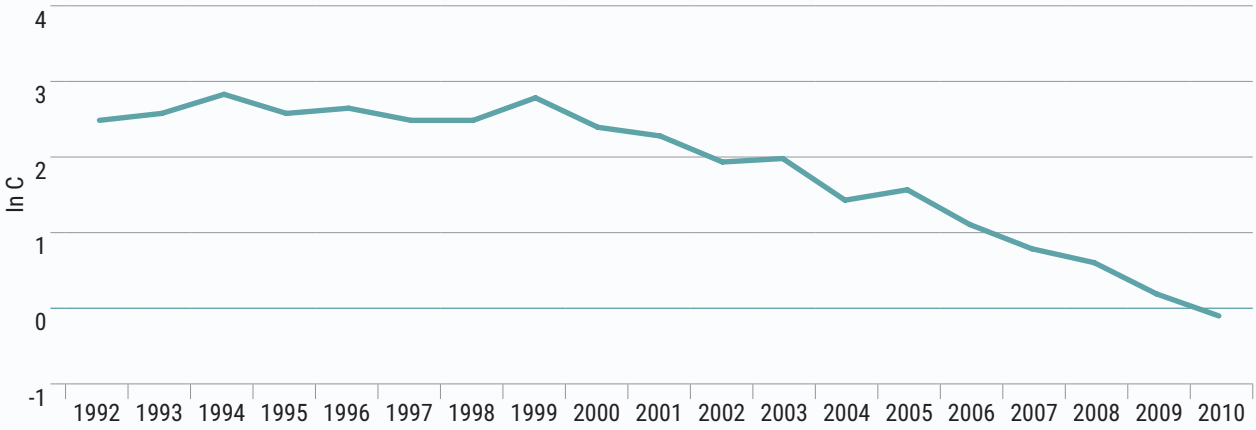
When pesticides reach higher levels in the atmosphere, they can be transported over short or long distances and deposited on land or water. This is particularly important in the case of pesticides not rapidly degraded in air. A well-known example is organochlorine pesticides used in tropical regions which have been found in the polar regions.

Atmospheric concentrations

Atmospheric contamination by pesticides is less well documented than that of other environmental compartments. There are relatively few systematic monitoring programmes for pesticides in ambient air. Since 2005 the Global Atmospheric Passive Sampling (GAPS) programme has deployed passive air samplers at over 50 locations on seven continents (Shunthirasingham *et al.* 2010). GAPS provides global atmospheric data on POPs, emerging POPs and some current use pesticides, both to domestic monitoring initiatives and to international programmes such as the Global Monitoring Plan (GMP) of the Stockholm Convention (UNEP 2017a). Historically, more attention has been paid to persistent organochlorine pesticides, but there is increasing interest in the occurrence and fate of current use pesticides in ambient air.

Many OCPs have been measured in notable concentrations in ambient air, not only near emission regions but also in distant areas. OCPs can move through the atmosphere by volatilization in relatively warmer source regions, transport in air, and subsequent deposit in colder regions. Polar regions and high mountainous areas

Figure 4.3.2-5 Time trends for γ-HCH (lindane) measured in air at the Canadian Great Lakes site, Point Petre. Atmospheric decline rates of γ-HCH have accelerated in the Arctic and the Great Lakes region following North American restriction of the pesticide lindane. UNEP (2017a).



then act as sinks (Kirchner *et al*. 2016; Wang *et al*. 2019).

The latest GMP monitoring report presents findings concerning aerial OCP concentrations at the global scale, based on information for the period 2000-2015 (UNEP 2017a). Time trends were mainly available for Asia, Europe and North America, and to a more limited extent for Africa, but were almost absent for Latin America. The GMP only covers studies of OCP concentrations conducted by institutions that are part of its network; other studies and publications are not reviewed by the programme (Annex 4.3-2).[4]

A clear decrease in ambient air concentrations of many OCPs has been observed in Asia, Europe, North America and the Caribbean, which seems to have followed their regulation in the 1980s and early 1990s (Shunthirasingham *et al*. 2010; UNEP 2017a; Rauert *et al*. 2018). For example, a reduction of lindane concentrations in the Great Lakes region of North America followed restrictions on the use of this pesticide (Figure 4.3.2-5). Similarly, reductions in emissions of α-HCH, DDT and toxaphene appear to have resulted in reductions in atmospheric concentrations in the Arctic (Li and MacDonald 2005). However, reductions in atmospheric concentrations of many other OCPs are less visible in polar and mountainous regions (Vorkamp and Rigét 2014; Kirchner *et al*. 2016; Wang *et al*. 2019).

By 2000, when the Stockholm Convention was adopted, the majority of primary sources of OCPs had been controlled. The relatively low levels currently measured in air do not show significant changes and appear to be driven by secondary sources (UNEP 2017a). Limited data available from Africa, however, do not show discernible downward trends in OCP air concentrations as yet.

OCPs such as hexachlorocyclohexane (HCH), endosulfan and DDT tend to be more prevalent in air in tropical regions (Shunthirasingham *et al*. 2010). For example, levels of DDT and its metabolites were a factor of 10-100 higher in tropical than in temperate countries (Mochungong and Zhu 2015). This was considered to be due to the continued use of DDT for vector control in Africa and Asia, as well as its higher dissipation in warmer climates.

Although concentrations of OCPs in the atmosphere are declining, they are still measured in air almost everywhere in the world, often decades after their use has been ended. This underlines the long time needed for these persistent compounds to disappear from the environment.

4 **Annex 4.3-2 is found at:** https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing

Coscollà and Yusà (2016) reviewed 34 studies on concentrations of pesticides in ambient air published between 2001 and 2014 (Annex 4.3-2). The more frequently monitored and/or detected CUPs worldwide were the fungicides chlorothalonil and folpet, as well as the insecticides chlorpyrifos, dimethoate, malathion and phosmet.

Shunthirasingham *et al*. (2010) observed declining concentrations of chlorothalonil in Europe; Vorkamp *et al*. (2014) found increasing concentrations of chlorpyrifos and trifluralin in the Arctic; and Guida *et al*. (2018) measured increasing concentrations of chlorpyrifos in Brazil. However, trends in atmospheric concentrations of CUPs have not been systematically reviewed.

Coscollà and Yusà (2016) concluded that inhalation exposure of CUPs from atmospheric concentrations in the general population does not represent a significant risk, except in some cases for chlorpyrifos. The combined risk resulting from exposure to organophosphate, pyrethroid and carbamate pesticides was also considered to be acceptable.

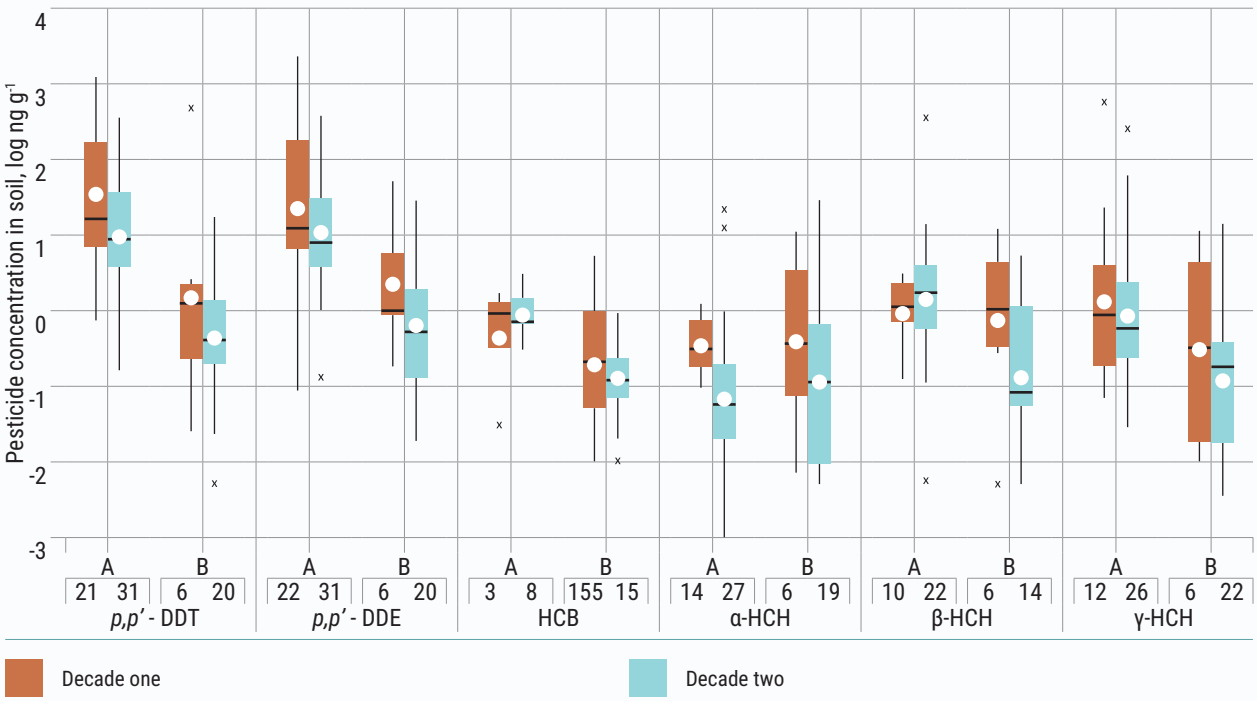
Pesticides in soil

After the target crop, agricultural soils are generally the first recipient of pesticides following their application (Hvězdová *et al*. 2018). Outdoor pesticide spray applications will almost inevitably contaminate soils through direct deposition, spray drift and wash-off from leaves. Moreover, pesticide use against soil-borne pests and diseases, as well as the use of systemic pesticides, may directly target the soil (e.g., through granular applications and seed treatments).

Further sources of soil contamination are spills and leakage from (obsolete) pesticide storage facilities, as well as spills or discharges from pesticide manufacturing and formulation plants.

The focus of this section will be on pesticide residues in soils after intended uses of

Figure 4.3.2-6 Global mean soil concentrations of most organochlorine pesticides in agricultural soils (A) and background soils (B) did not significantly decrease over time (decade one (1993-2002) to decade two (2003- 2012). Camenzuli, Scheringer and Hungerbühler (2016).



Note: Each box plot indicates the median value (–), the mean value (o), the quartile values (box) and the 10th to the 90th percentile values (vertical lines). Distribution outliers are marked by x. The number of data points in each box plot is given at the bottom.

pesticide products. A summary of the reviewed assessments is provided in Annex 4.3-3.[5]

Global reviews

The most recent global review of OCPs in soils was carried out by Camenzuli, Scheringer and Hungerbühler (2016). They reviewed all scientific publications on concentrations of DDT, DDE, HCB and HCH isomers in agricultural and background (unsprayed) soils, published between 1993 and 2012, and attempted to assess whether such concentrations had decreased over time. Statistically significant decreases were only observed for p,p'-DDT in agricultural soils and HCH isomers in background soils; concentrations of other OCPs in soil did not significantly decline during that period (Figure 4.3.2-6). This contrasts with the results of a global environmental fate and transport model, based on estimated emission data, which did suggest a decrease over time. The authors of the review therefore concluded that new emissions of DDT and HCH cannot be excluded.

Wang *et al*. (2019) recently reviewed OCP concentrations in soils in the polar regions (the Arctic, the Antarctic and the Tibetan Plateau). While currently measured concentrations were relatively low, DDTs, HCHs and HCB were found in soil in all these regions. They concluded that soils in cold regions can be considered sinks of POPs.

Few recent global reviews of current use pesticide concentrations in soils are available. Hvězdová *et al*. (2018) commented that "although agricultural soil is a primary sink and key reservoir of pesticides, large soil surveys of agricultural soils for current use pesticides are surprisingly rare".

Komárek *et al*. (2010) reviewed worldwide data on pesticide residues in vineyard soils. They highlighted that copper, originating from the intensive application of copper-based fungicides, belongs to the most important contaminants of vineyard soils, and that soil residues exceed EU environmental standards in most evaluated cases.

Wood and Goulson (2017) compiled available information on neonicotinoid insecticides, which are under close scrutiny because of various suspected adverse environmental effects. They found that neonicotinoids can be present in soils several years after the last application date, although they do not continue to accumulate indefinitely, plateauing after two to six years of repeated application.

Regional and national reviews

The large majority of regional and national studies on pesticides in soils concern OCPs even if, in many countries, these pesticides are not used anymore. Their persistence in soil makes them pesticides of interest for monitoring.

OCP residues in soil were detected in almost all countries where they had recently been measured, although concentrations were variable (Annex 4.3-3). In most cases soil concentrations were below national environmental standards (Sun *et al*. 2018 for China; Fosu-Mensah *et al*. 2016 for Ghana; Łozowicka *et al*. 2016 for Poland and Kazakhstan). High levels of OCPs in soil which exceeded health, environmental or trade standards were encountered in India (Mishra, Sharma and Kumar 2012), the French West Indies (Levillain *et al*. 2012) and Tanzania (Elibariki and Maguta 2017).

Contrary to the global trend described by Camenzuli, Scheringer and Hungerbühler (2016) (see above), Sun *et al*. (2018) did observe a decline in concentrations of OCPs in soil in China over time. Few other studies have assessed time trends of OCPs in soil.

The sources of OCPs in soil are not always clear. In many cases they persist from historical uses; in others legal or illegal recent uses of pesticides such as DDT and lindane have been identified (Mishra, Sharma and Kumar 2012; Fosu-Mensah *et al*. 2016). DDT residues in soils may also originate from the use of dicofol, which degrades into DDT in the environment.

Irrespective of the variability of OCP residues in soil, these residues are ubiquitous around the world, even when the OCPs are not (or are hardly)

5 **Annex 4.3-3 is found at:** https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing

used anymore, underlining the "environmental legacy" of persistent pesticides.

Regional and national studies of current use pesticides in soils appear to be mainly from Europe and China (Annex 4.3-3).

Silva *et al*. (2019) sampled agricultural topsoils across 11 EU countries in six major cropping systems. They detected pesticide residues in 83 per cent of tested soils (58 per cent of samples contained multiple residues) and concluded that pesticide residues in agricultural soil are the rule rather than the exception. Glyphosate and its metabolite AMPA, and the broad-spectrum fungicides boscalid, epoxiconazole and tebuconazole, were found most frequently and at the highest concentrations. DDTs (DDT isomers) were also frequently found despite a decades-long ban on use of this pesticide in the EU. The authors concluded that pesticide concentrations in soil were generally below the respective toxic endpoints for standard in-soil organisms.

The pervasive presence of current use pesticides in soil is confirmed elsewhere. In the Czech Republic, Hvězdová *et al*. (2018) detected pesticide residues exceeding national acceptable soil limits in 81 per cent of samples across the country. Most of these residues were triazine herbicides and conazole fungicides. Humann-Guilleminot *et al*. (2019) conducted a country-wide survey of neonicotinoid residues in soil in Switzerland. They found that all soil in conventional fields contained neonicotinoid residues, as did 93 per cent of organic soils which were supposed to be free of pesticides, highlighting the importance of diffuse soil contamination from surrounding uses. Pelosi *et al*. (2021), in a recent study in France, found at least one current use pesticide in all the 180 soils sampled both in treated cereal fields and non-treated habitats such as hedgerows, grasslands and organic cereal fields. In 83 per cent of samples, five or more CUPs were detected. In addition, Pan *et al*. (2018) detected organophosphate pesticides in 93 per cent of soils sampled along the Yangtze River Delta.

Pesticides in groundwater and drinking water

Pesticides applied to agricultural fields may be vertically displaced downwards from the topsoil through the soil profile and the unsaturated zone to groundwater, a process called leaching (Figure 4.3.2-1). Since groundwater is an important source of drinking water, both environmental and human health effects may be caused by groundwater pesticide pollution. The extent to which pesticides leach to groundwater depends on a large number of factors related to soil and pesticide properties, site conditions and management practices (Figure 4.3.2-7).

Pesticide properties also play an important role. The higher the water solubility of a pesticide, the greater its potential to dissolve in water infiltrating the soil. Pesticides for which there is only a short time to detect a 50 per cent decrease in pesticide concentration (detection time 50 per cent; DT50) may be degraded before reaching groundwater levels. Similarly, pesticides with a high soil adsorption coefficient (Koc) are expected to be retained in topsoil layers. It should be noted, however, that pesticides with low water solubility and high Koc (and DT50) values have greater potential for particle-bound transport, i.e., adsorption to particles in infiltrating water (Goss and Wauchope 1990).

Besides soil and pesticide properties, site characteristics (e.g., depth to groundwater, weather and climate) and management practices (e.g., application dosage) affect pesticide leaching to groundwater. Ultimately, the combination of factors shown in Figure 4.3.2-8 will determine whether, and to what extent, pesticides will leach to groundwater under field conditions.

The difficult accessibility of groundwater ecosystems hampers monitoring of pesticides, which is often restricted by the availability of superficial sampling spots. In addition, although some countries have extensive national monitoring programmes, efforts in other parts of the world are meagre or non existent (Pirsaheb *et al*. 2017).

Figure 4.3.2-7 **Factors influencing pesticide leaching to groundwater.** Adapted from Pérez-Lucas *et al.* (2018).

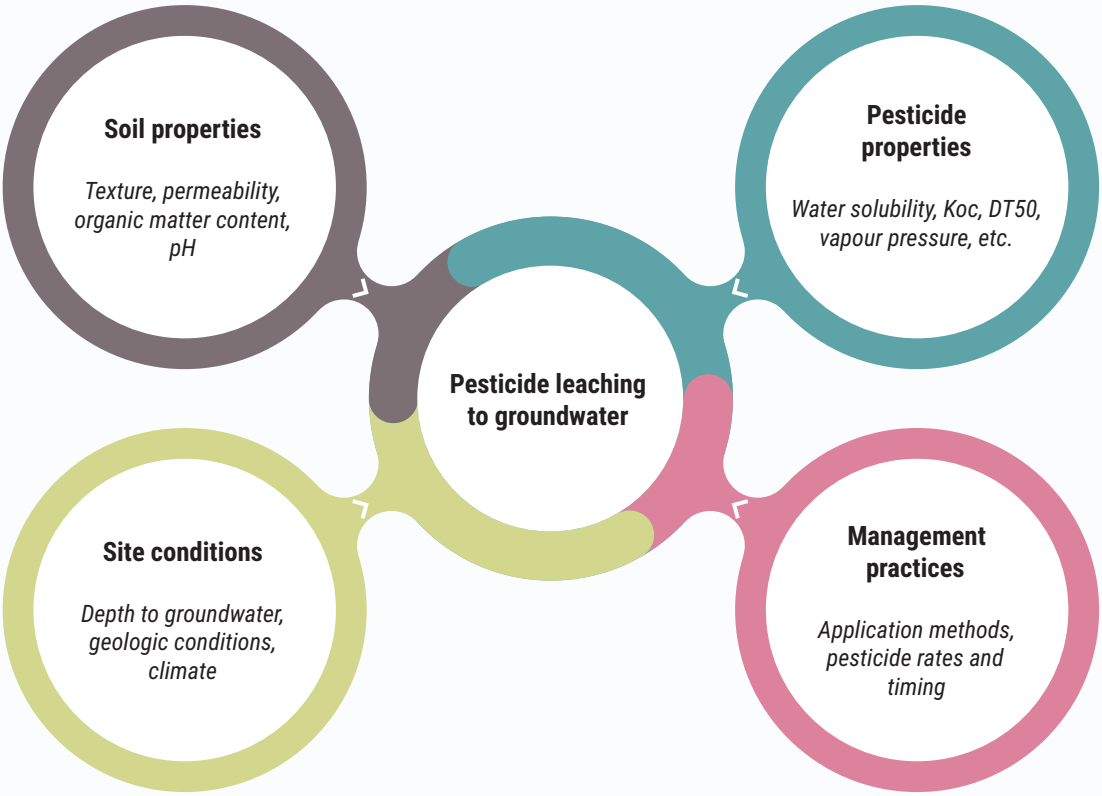
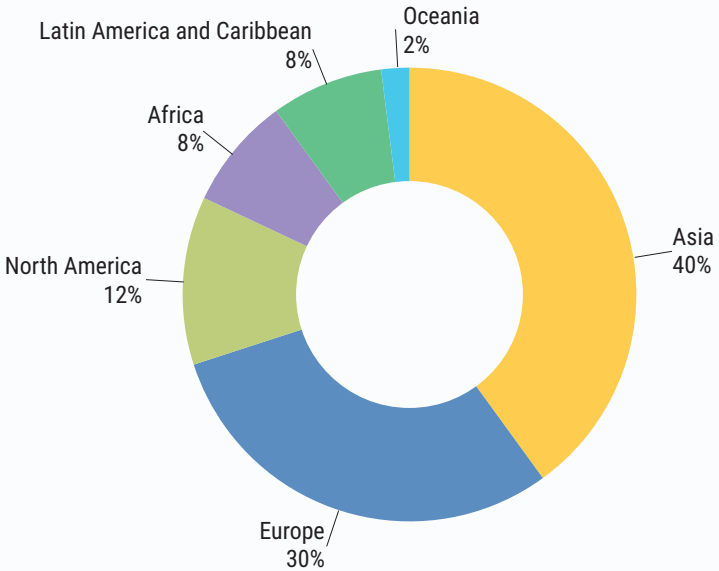
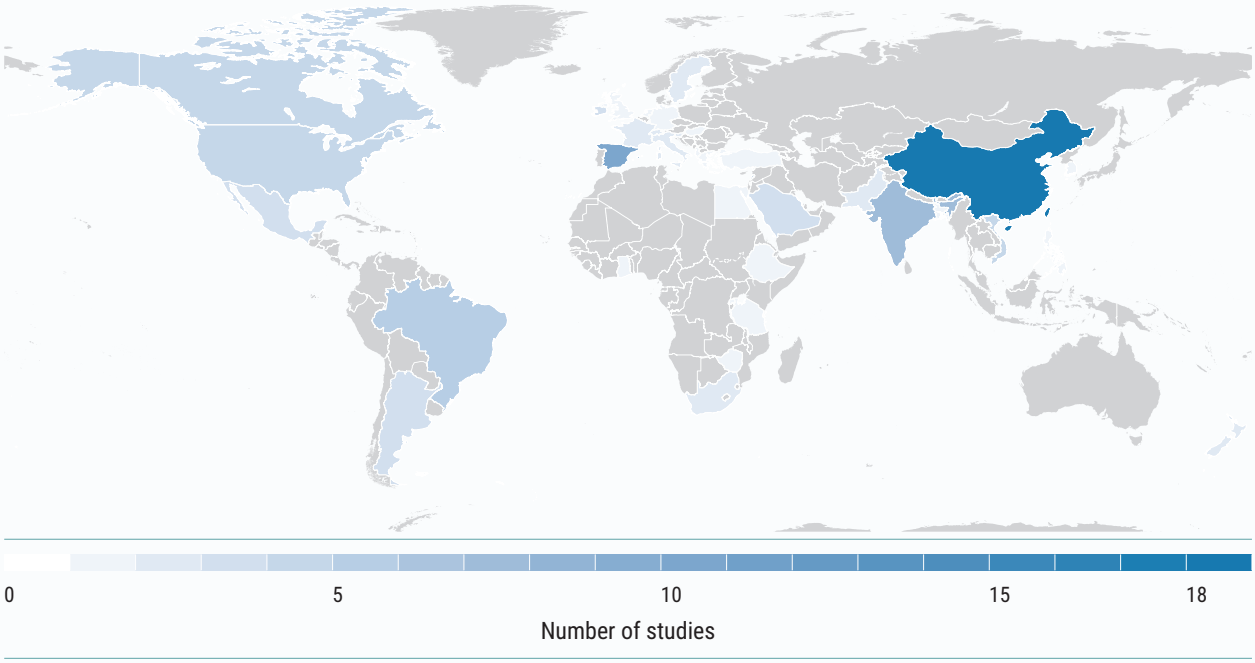


Figure 4.3.2-8 **Relative availability of local and national groundwater monitoring/review studies per continent (total = 103 publications between 2010 and 2020).**



Based on studies of pesticides in groundwater and drinking water published since 2010 (listed in Annex 4.3-4), the geographical distribution per country and continent of country-wide and local studies combined are shown in Figures 4.3.2-8 and 4.3.2-9. The main outcomes of the country-wide

Figure 4.3.2-9 Geographical distribution of available local and national groundwater monitoring/review studies (total = 103 publications between 2010 and 2020).



monitoring or review studies are presented in Annex 4.3-4.[6]

The focus of this section is on pesticides in groundwater, as a source of drinking water. However, it should be emphasized that in some parts of the world, especially in Africa, East and Southeast Asia, surface water is (also) an important source of drinking water. Pesticides are commonly detected in surface water (Chapter 4.3.2) and the analysis of risk from pesticides through drinking water should not be limited to groundwater only.

The largest number of studies are available for Asia, followed by Europe and North America, while relatively few studies are available for the other continents (Figure 4.3.2-8). Within continents studies are not evenly distributed, with a few countries making up the bulk of available studies and many countries poorly or even not represented (Figure 4.3.2-9). For example, China and India account for 61 per cent of Asian studies, and in South America all available studies are from Brazil and Argentina. For seven African countries a single study was found. There were two studies for South Africa. In Europe studies were available for 15 countries, with Spain having the highest number.

Global reviews

Only three recent global reviews of pesticides in groundwater were identified (Annex 4.3-4).

Pirsaheb *et al*. (2017) was the only available global review of the occurrence of OCPs in groundwater. Consistent with Figure 4.3.2-9, the authors noted that most studies had been conducted in Asian countries. OCP diversity and concentrations were generally lower than those of organophosphorus pesticides (OPPs) and concentrations in groundwater were lower than in surface waters. At the same time, many OCP concentrations above legal national limits were identified, especially in Asian countries.

Pesticides from the first and second EU Watch Lists were reviewed by Pietrzak *et al*. (2019). Neonicotinoids, especially imidacloprid, acetamiprid and thiamethoxam, were the most commonly monitored compounds. They were

6 **Annex 4.3-4 is found at:** https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing

found on almost every continent more than once at concentrations above EU legal drinking water limits. Only one study was available for pesticides from the Watch Lists other than neonicotinoids, which indicated an exceedance of the EU limit of 0.1 microgram/litre (µg/L) (8 µg oxadiazon/L in Italy).

A review by Postigo and Barceló (2015) concluded that pesticide transformation products (TPs) are not commonly included in monitoring studies. It noted that TPs from pesticides that are no longer locally registered, such as atrazine and terbuthylazine, were frequently detected. This was attributed to their long residence time in groundwater and/or the slow release of their precursors from the soil. Occasionally, pesticide TPs appear to be more ubiquitous and abundant than their parent molecules and to exceed EU limits.

Regional and national reviews

The regional review for Africa by K'oreje *et al*. (2020) confirms the low data availability for the continent (Figures 4.3.2-8 and 4.3.2-9). Although use of most organochlorine compounds (e.g., aldrin, DDT, dieldrin, endosulfan, endrin, heptachlor and lindane) has been discontinued in many African countries at different times since 1976, their recent high detections indicate their environmental persistence, their continuous introduction into the environment, or weak enforcement of the ban. Moreover, in some countries (e.g., Kenya, South Africa) restricted use of DDT for malaria mosquito control is still allowed (K'oreje *et al*. 2020).

Several national reviews for India reveal high concentrations of HCHs and endosulfan isomers, as well as DDT metabolites, in groundwater (Yadav *et al*. 2015; Malyan *et al*. 2019; Sackaria and Elango 2019). Almost all reviewed studies indicate high groundwater concentrations (often to always > 0.1 µg/L) throughout the country, especially during or shortly after the monsoon season. Since pesticide use in India is expected to increase, groundwater pollution is in that country is also likely to increase in the future (Malyan *et al*. 2019).

In other parts of the world where OCPs have been banned, they frequently continue to be detected at low but often stable or only slowly decreasing levels (Annex 4.3-4). In the United States, Toccalino *et al*. (2014) reported increasing benchmark exceedances for dieldrin despite its having been banned for over 25 years, as well as increasing and frequent detections of pesticide TPs. Cattan *et al*. (2019) found that groundwater concentrations of the OCP chlordecone decreased or remained stable between 2009 and 2015, but that its metabolite showed an increasing, although rather erratic, evolution overall (Figure 4.3.2-10).

Figure 4.3.2-10 Temporal trend in (A) chlordecone (CLD) and (B) its metabolite chlordecone-5b-hydro (5bCLD) between 2009 and 2015 in Chalvet (Martinique). Cattan *et al*. (2019).

Besides slow release from soil, in which OCPs are very stable, high OCP groundwater levels have been attributed to dumping, illegal use and inappropriate storage (Eqani *et al*. 2012; Hasan, Shahriar and Jim 2019).

Three regional reviews of current use pesticides were identified. As in the case of POPs, the regional review for Africa by K'oreje *et al*. (2020) indicated that information for current use pesticides is scarce. The need for more information from the continent is underlined by the fact that groundwater concentrations of pesticides (other than OCPs) in Africa occur in the same order of magnitude as in Western countries (K'oreje *et al*. 2020). A regional review for French overseas departments (Guadeloupe, Martinique, Réunion, Mayotte and French Guiana) indicated that several pesticide concentrations exceeded the EU groundwater quality standard (Vulliet *et al*. 2014). Leusch *et al*. (2018) measured pesticide concentrations in drinking water in six countries in Europe, Africa and Oceania (France, Germany, the Netherlands, Spain, South Africa and Australia). None of the pesticides were detected in Germany, South Africa or Australia; individual pesticides were detected in other countries, but at concentrations below the EU drinking water standard of 0.1 µg/L.

In line with Leusch *et al*. (2018), a national study in Lebanon (Kouzayha *et al*. 2013) indicated pesticide concentrations in drinking water in the low nanogram (ng)/L range (maximum 20 ng/L). Routine monitoring data in the Netherlands collated by Sjerps *et al*. (2019), however, indicated that 15 of 24 recently authorized pesticides were detected, including neonicotinoids. In one-third of the abstraction areas pesticide and/or metabolite concentrations exceeded water quality standards according to the EU Water Framework Directive (EU 2000). In addition, while Leusch *et al*. (2018) did not detect any of these pesticides at the sites evaluated in South Africa, Odendaal *et al*. (2015) encountered atrazine and terbuthylazine in drinking water at concentrations > 0.1 µg/L in seven South African cities although WHO drinking water limits were not exceeded. Another local study in three catchments of South Africa (Machete and Shadung 2019) also detected several pesticides in drinking water. While most of these pesticides were detected at low concentrations, they included (potentially) endocrine disrupting chemicals such as atrazine, alachlor and simazine.

Jurado *et al*. (2012) reported that Spanish groundwater was considerably less contaminated than other water bodies (e.g., rivers), although a wide array of compounds exceeded the EU limit of 0.1 µg/L. Two recent country-wide studies in China (Sun *et al*. 2020; Wang *et al*. 2020) also indicated higher herbicide concentrations in surface water than in groundwater. Concentrations in surface water derived tap water were higher than those detected in groundwater. However, there does not appear to be a clear consistent trend of higher concentrations in surface water than in groundwater. For example, Karki *et al*. (2020) found higher concentrations of the herbicides atrazine and bentazon in Swedish groundwater than in surface waters. Ultimately, the factors in Figure 4.3.2-7, in addition to treatment of received water in the case of drinking water, will determine pesticide concentrations in different water resources. Depending on these factors, pesticide detections range from undetected (i.e., < LOQ) to values that are clearly higher than the EU limit (Annex 4.3-4).

Overall, herbicides have generally been reported as the pesticide type with the most frequent detections, highest concentrations, and most frequent exceedances of regulatory limits (Close and Skinner 2012; McManus *et al*. 2014; Lopez *et al*. 2015). Close and Skinner (2012) indicated in their review for New Zealand that 17 of the 22 pesticides detected were herbicides. In France, Lopez *et al*. (2015) reported that detected pesticides in their country-wide monitoring campaigns were dominated by herbicides (68 per cent of sites), followed by fungicides (7.5 per cent) and insecticides (1.4 per cent). That might at least partly be due to the fact that herbicides appear to be included more frequently in monitoring studies (Annex 4.3-4). The 103 pesticides monitored in the study by Lopez *et al.* (2015) included 48 herbicides, 29 fungicides and 26 insecticides. The influence of research efforts may also be reflected in the detection of other pesticides. Jurado, Walther and Diaz-Cruz (2019) reported that imidacloprid was the only neonicotinoid detected in Spanish groundwater, but that other neonicotinoids were

the subject of only one or two studies. In their global review, Pietrzak *et al.* (2019) also indicated that neonicotinoids other than imidacloprid have hardly been studied in groundwater.

Temporal evaluations of groundwater pesticide concentrations have indicated stable (Close and Skinner 2012), decreasing (Köck-Schulmeyer *et al*. 2014) and increasing (Di Lorenzo *et al*. 2018) trends. Toccalino *et al*. (2014) reported a decrease in atrazine concentrations in the United States, but an increase in their TPs. Persistent pesticides such as atrazine, and especially their TPs, have consistently been detected in groundwater decades after their registration was discontinued due to their long residence time in the subsurface and/or the slow release of their precursors from the soil (Lopez *et al*. 2015 and references therein; Postigo and Barceló 2015; Karki *et al*. 2020). Differences in study designs (e.g., sampling and chemical analysis methods), locations and sampling dates make it difficult to establish national, let alone global, trends in groundwater pollution. Manamsa *et al*. (2016) concluded that the large monitoring dataset available for England and Wales (data for 2,650 sites from 2003 onwards) did not provide sufficient data for any compound for a single site to determine a trend.

Exceedances of drinking water standards

Annex 4.3-4 includes several studies that evaluated pesticide concentrations in drinking water at regional (Leusch *et al*. 2018; K'oreje *et al*. 2020) or national (Kouzayha *et al*. 2013; Odendaal *et al*. 2015; Karki *et al*. 2020; Sun *et al*. 2020, Wang *et al*. 2020) scales, covering studies conducted around the world (Australia, France, Germany, Lebanon, South Africa, Spain, Sweden, the Netherlands). Pesticide concentrations in many of these studies were reported in the low nanograms per litre (ng/L) range, with no human health risks expected at these exposure levels (Kouzayha *et al*. 2013; Leusch *et al*. 2018; Karki *et al*. 2020).

In their review of African studies, K'oreje *et al*. (2020) noted that in many African countries groundwater is generally used as drinking water without any treatment. They indicated that pesticides have been detected in groundwater in Ethiopia, Ghana, Kenya, Nigeria, South Africa and Zambia at concentrations between 0.1 ng/L and 18.4 micrograms per litre (µg/L). Pesticides were also detected in treated drinking water in Algeria, Ethiopia, Nigeria and South Africa, with concentrations ranging from 0.02 ng/L to 34 µg/L (K'oreje *et al*. 2020).

Odendaal *et al*. (2015) conducted a national survey over four seasons of potential compounds of emerging concern, including pesticides, in the drinking water of major South African cities. The herbicides atrazine and terbuthylazine were most frequently detected, although concentrations were at least one order of magnitude lower than the guideline values set by the United States Environmental Protection Agency (US EPA) and the WHO. For example, maximum atrazine concentrations were between 150 and 200 ng/L, whereas the proposed WHO guideline value is 100 µg/L and the maximum contaminant level stipulated by US EPA is 3 µg/L (Odendaal *et al*. 2015). The European Drinking Water Directive, however, establishes a drinking water standard of 0.1 µg/L for each individual pesticide and its toxicologically relevant metabolites, with the exception of aldrin, dieldrin, heptachlor and heptachlor epoxide (for which a quality standard of 0.03 µg/L applies) (European Union 1998; European Union 2020a). Thus the maximum atrazine concentrations encountered by Odendaal *et al*. (2015) exceed the EU quality standards, but not those of US EPA and WHO.

Wang *et al*. (2020) discuss a similar situation in China: while maximum atrazine drinking water concentrations were well below the country's national quality standard and those set by Health Canada, US EPA and WHO, they exceeded the European standard of 0.1 µg/L. The same conclusion was reached by Sun *et al*. (2020) in their country-wide study on drinking water concentrations of the herbicides 2,4-D and MCPA in China. This wide difference between the EU standard and others (e.g., those of US EPA and WHO) is related to the different approaches followed in setting these standards. The EU drinking water standard, which was set in the 1980s, followed the precautionary principle and corresponded to the contemporary detection limit for pesticides. On the other hand, the US EPA and WHO standards are health-based and calculated based on toxicity data.

4.3.3 Sustainability of agricultural production and vector control

Various adverse environmental effects of pesticide use may have a direct or indirect impact on the sustainability of agricultural production or of disease vector control. Rather than increasing production or reducing disease vector populations, pesticide use can have the opposite effect, with mid- and long-term effects on agriculture and public health. Such adverse effects include:

- development of resistance of pest organisms or disease vectors against pesticides;
- impact on bees and other pollinators and possible reduction of crop pollination;
- impact on natural enemies of pests, and possible pest resurgence or appearance of secondary pests;
- effects on soil organisms and possible reduction of soil fertility.

These unintended effects of pesticides on resistance development, pollinators and natural enemies are discussed in more detail below. Effects of pesticides on soil organisms are reviewed in Chapter 4.3.4.

Resistance

The development of resistance against pesticides in arthropods, diseases, weeds and rodents is an important agronomic, economic, ecological and public health problem throughout the world. Resistance development occurs when the genetic makeup of a pest population changes in response to selection by pesticides. This, in turn, may lead to repeated failure of a pesticide to achieve the expected level of control when used according to the label recommendation for that pest species.

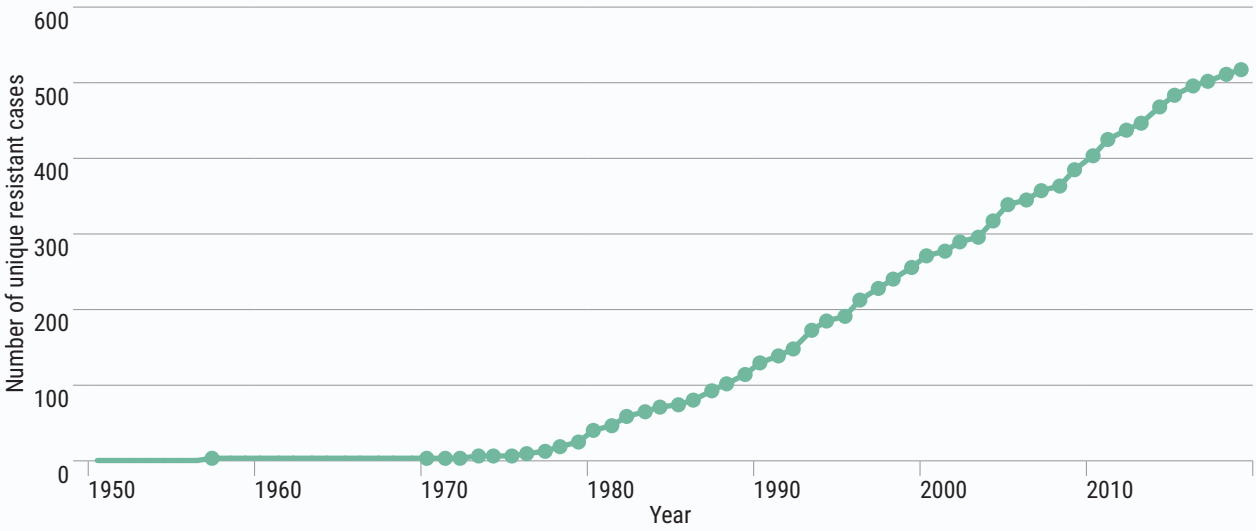
As a response to the development of resistance, farmers often increase the dose and/or frequency of pesticide applications, which in turn may lead to greater resistance in the pest populations, encouraging greater pesticide use. Consequently, agricultural production becomes more expensive or even economically unviable; adverse health effects increase; pesticide residues in crops exceed acceptable standards; and environmental impact expands. In terms of public health (e.g., combatting malaria), resistance development may lead to a breakdown in vector control and an increase in the prevalence of a disease. Alternatively, to break resistance farmers and public health entities will need to change the type of pesticide used to one with a different mode of action or move completely away from chemical pesticides, replacing them with biological control. This may require a change in crop production or vector control methods and may increase costs (FAO 2012).

Herbicide resistance

By mid-2019 there were 502 unique cases of herbicide resistant weeds globally, comprising 258 different weed species (Heap 2019). Weeds have evolved resistance to 23 of the 26 known herbicide modes of action and to 167 different herbicides. Herbicide resistant weeds have been reported in 93 crops from 70 countries. The number of cases of herbicide resistance has been growing steadily since the early 1970s (Figure 4.3.3-1). Globally, during the last 30 years on average 13 new cases have been reported every year, irrespective of the introduction of new herbicides or herbicide resistance management programmes.

Herbicide resistance leads to very serious agronomic, economic and environmental challenges. A well-known case is the failure of ryegrass control as a result of herbicide resistance in Australia. For more than 20 years Australia was known for having the most serious cases of herbicide resistance in the world. Certain herbicides considered essential for farmers' cropping systems can no longer be used due to resistance, and in major cropping areas all ryegrass has become resistant to selective herbicides (Pannell *et al.* 2016). Alternative herbicides have had only a limited effective life, as ryegrass has been able to rapidly develop resistance to these chemicals as well. Alternative weed management options have had to be developed, such as new methods to capture and destroy weed seeds at harvest time, although at greatly increased costs (Stokstad 2013; Pannell *et al.* 2016).

Figure 4.3.3-1 Global increase of unique cases of herbicide resistance. A unique case is a weed species combined with a herbicide mode of action. Heap (2019).



Similar problems in managing herbicide resistant weeds exist in all countries where chemical herbicides are used on a large scale. While farmers have often relied on applying new herbicides with new modes of action, no such new types of herbicide have been commercially developed in decades and there do not seem to be any "silver bullets" on the way (Pannell et al. 2016).

Insecticide resistance

The first case of insecticide resistance occurred more than 100 years ago, when in 1913 the San Jose scale was found to resist sprays of sulphur in fruit trees in the United States (Whalon 2008). Since the 1960s the use of organic insecticides has taken off and arthropod resistance has been increasing steadily, with no sign of tapering off (Figure 4.3.3-2).

By late 2019 almost 17,000 cases of arthropod resistance to 345 different pesticides had been reported globally, representing more than 600 species of insects and mites. Of these, almost 14,000 cases of resistance against 587 species were field-evolved, i.e., the result of pesticide use under actual field conditions rather than in the laboratory (Mota-Sanchez and Wise 2019; Mota-Sanchez, R., personal communication).

The speed of development of arthropod resistance mainly depends on the species and is less influenced by the pesticide's mode of action (Brevik *et al*. 2018). However, more resistant species do not develop resistance faster than less resistant ones. On average, in 20 studied species of insects and mites, the median duration between the introduction of an insecticide and the first report of resistance was 66 generations or about 14 years. Importantly, the year of first introduction of a pesticide does not seem to influence the speed with which arthropods become resistant (Brevik *et al*. 2018) (Figure 4.3.3-3). This suggests that insecticide and acaricide resistance management programmes, which have been developed and implemented over several decades, do not appear to have significantly reduced the speed of insecticide resistance development.

Insecticide resistance is of particular concern for the control of disease vectors such as mosquitos or sandflies. In a recent evaluation, WHO (2020a) reported that pyrethroid resistance had been confirmed in major malaria vectors in 77 per cent of countries that conducted monitoring. In 37 countries at least one malaria vector species was resistant to three or four different classes of insecticides used in indoor residual spraying (Figure 4.3.3-4). Pyrethroid resistance in malaria mosquitos increased significantly between 2010 and 2016. Since pyrethroid insecticides are the only class of insecticides available to treat bed nets in many

Figure 4.3.3-2 **The evolution of pesticide resistance in arthropods shows a steady increase in the number of species, pesticide compounds and cases from 1914 to 2019.** Mota-Sanchez and Wise (2019), Mota-Sanchez pers. comm.

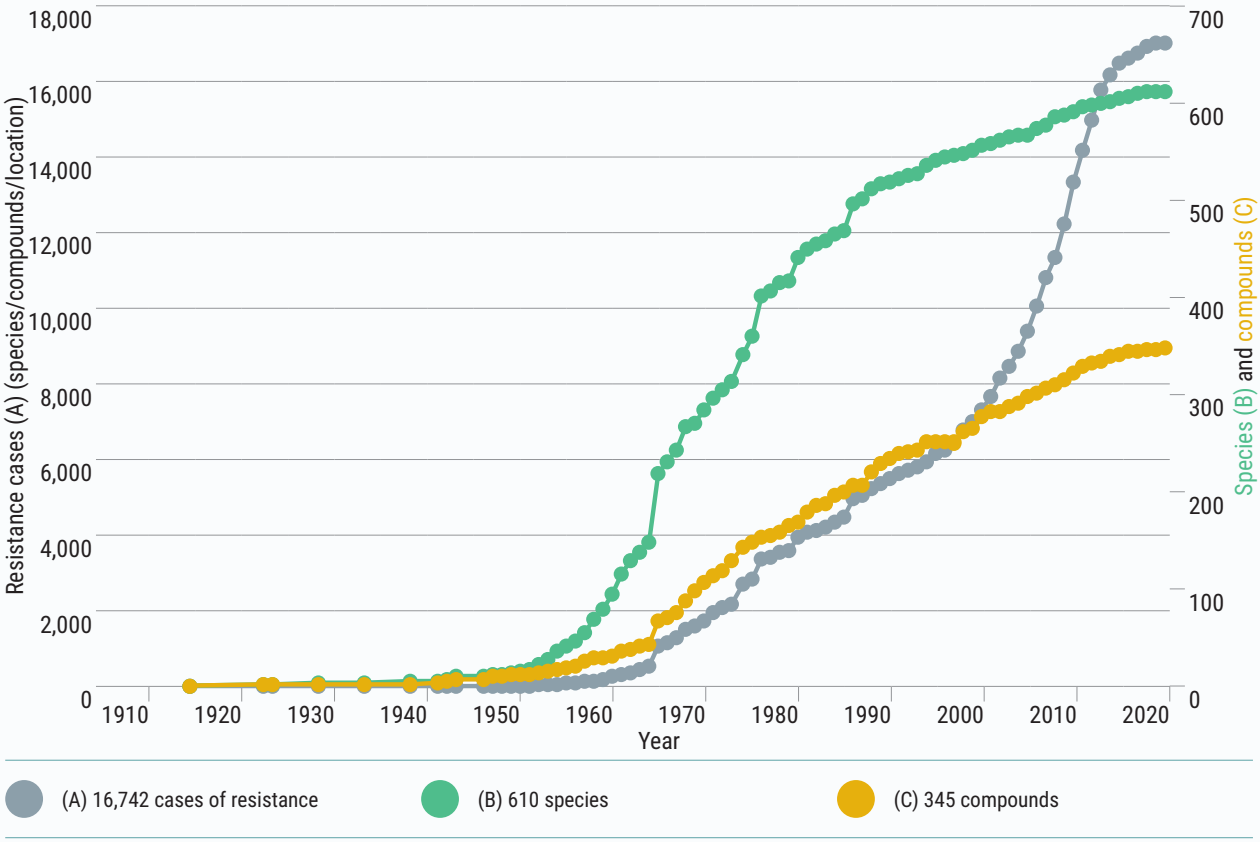


Figure 4.3.3-3 **Arthropods were becoming resistant to a pesticide as rapidly in 2010 as in the 1960s, indicating that there had not been a change in the "durability" of a pesticide over time. No significant relation was observed between the number of generations it takes for an arthropod to become resistant and the year of introduction of a pesticide.** Brevik *et al.* (2018).

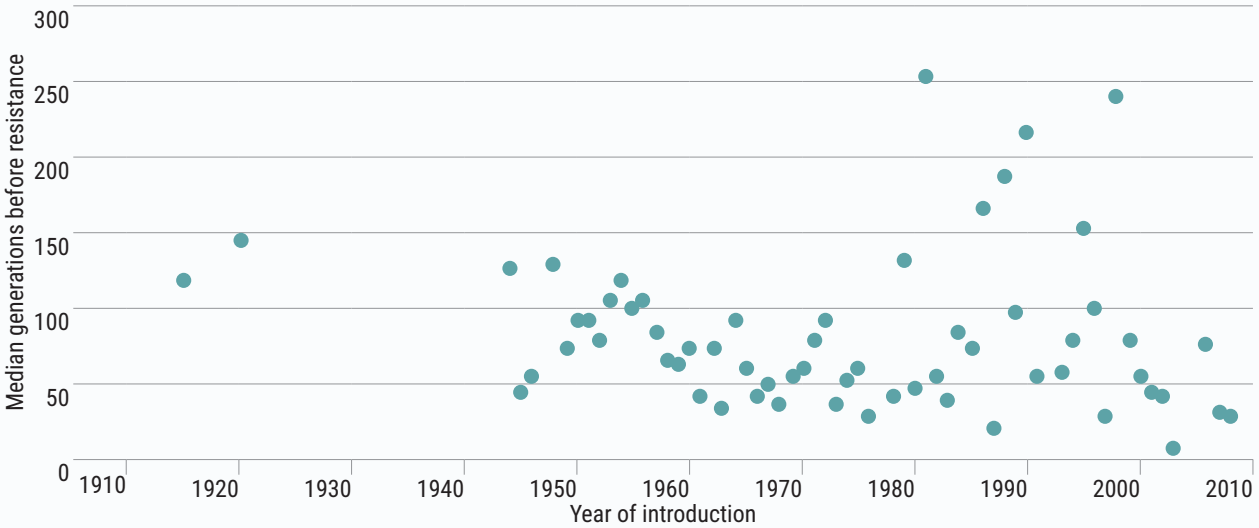
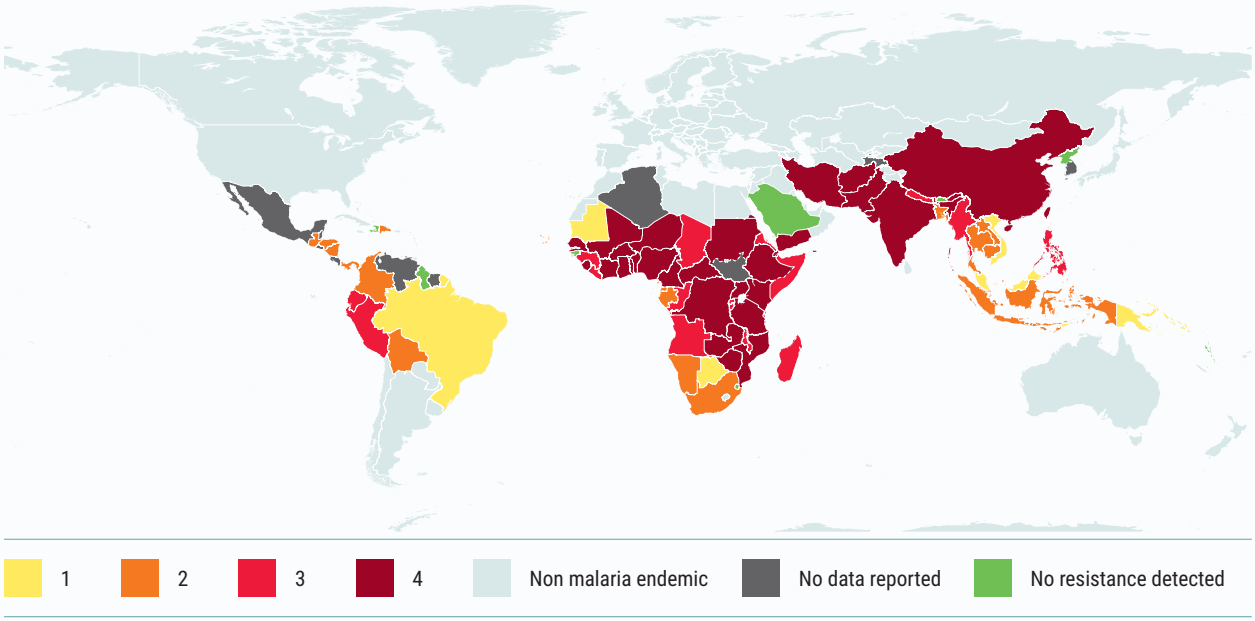


Figure 4.3.3-4 Number of classes to which resistance was confirmed in at least one malaria vector in at least one monitoring site, 2010–2019. WHO (2020).



countries, widespread pyrethroid resistance, particularly in Africa, is a serious problem for effective malaria vector control (Ranson and Lissenden 2016).

Vector resistance to insecticides is not limited to malaria mosquitos. It is also found in other vectors of human disease. Dhiman and Yadav (2016) reviewed resistance in phlebotomine sandflies, which are vectors of visceral leishmaniasis (kala-azar). They found widespread resistance against DDT and local resistance against pyrethroids in areas of the Indian subcontinent where the disease is prevalent. A recent survey in North America also showed pyrethroid resistance to be widespread in Florida (United States) in Aedes aegypti, the mosquito vector of the Zika virus (Estep *et al*. 2018). Guedes *et al*. (2020) found a high prevalence of insecticide resistance of the same species in Latin America and the Caribbean.

Fungicide resistance

Fungicides have been used in agriculture for well over a century, but the earliest documented cases of fungicide resistance date from the 1960s. Overall, instances of confirmed resistance to fungicides remained rare until the 1970s, when novel classes of antifungal pesticides were introduced and became widely used (Lucas, Hawkins and Fraaije 2015). Since then there has been an ever-increasing incidence of reported resistance cases in a wide range of plant pathogenic fungi (Table 4.3.3-1).

Genetically modified crops

Resistance development in genetically modified crops has focused on herbicide tolerant and insect resistant crops. In the former, weed resistance develops against herbicides applied to a crop; in the latter, insects become resistant to a pesticide incorporated in a crop.

The most successful insect resistant transgenic crops produce insecticidal proteins from the bacteria *Bacillus thuringiensis* (Bt). Increasingly, more than one toxin is built into the crop ("Bt crop pyramids") to increase efficacy and delay resistance. Tabashnik and Carrière (2017; 2019) recently reviewed 44 studies, from 12 countries on six continents, on the development of field-evolved insect pest resistance to transgenic crops. In 19 cases no decrease in susceptibility to Bt toxins was observed, in six cases there were early warning signs of the development of resistance, and in 19 cases practical resistance had evolved. The efficacy of the Bt crop in controlling insects

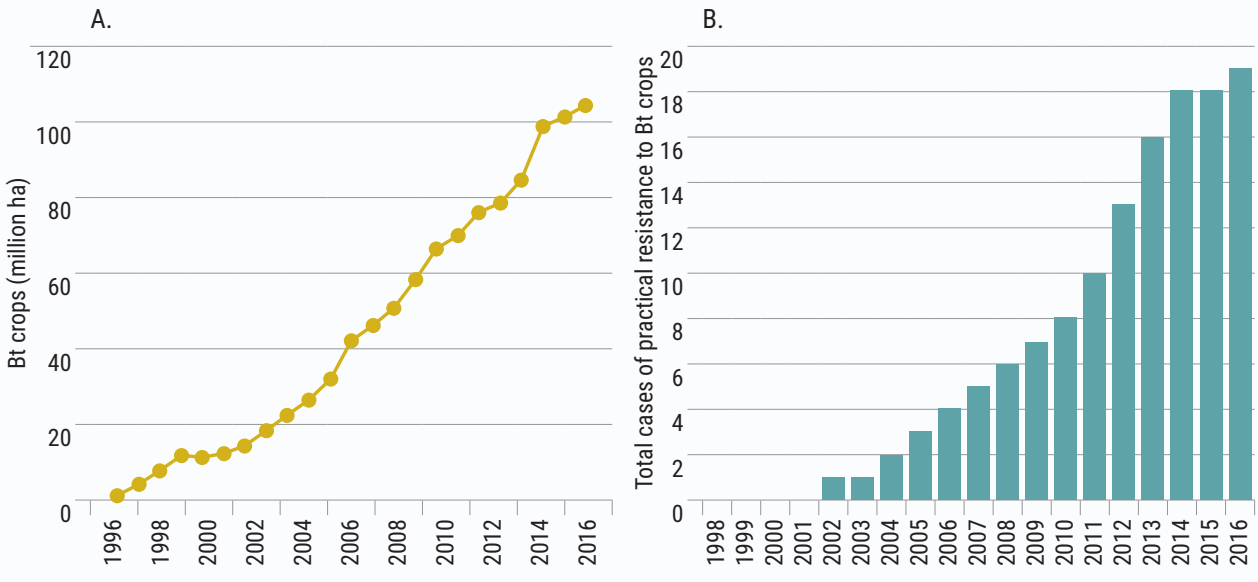
Table 4.3.3-1 Time line of fungicide resistance in crop diseases for some major fungicide classes. Lucas, Hawkins and Fraaije (2015).

Period of introduction	Fungicide class	Approximate time to resistance (years)	Disease example
1960	Aromatic hydrocarbons	20	Citrus storage rots (Penicillium spp.)
1964	Organomercury compounds	40	Cereal leaf spot (Pyrenophora spp.)
1969	Dodine (guanidine) compounds	10	Apple scab (Venturia inaequalis)
1970	Benzimidazoles	2	Many pathogens
1971	2-Aminopyrimidines	2	Powdery mildews
1980	Phenylamines	2	Potato late blight, grape downy mildew
1982	Demethylation inhibitors	7	Cereal powdery mildew and other diseases
1998	Quinone outside inhibitors	2	Cereal powdery mildew
2007	Succinate dehydrogenase inhibitors	4–5	Alternaria alternata (in nuts), potato early blight (Alternaria solani)

was therefore reduced in 43 per cent of reports. In some cases (e.g., the Western corn rootworm in maize in the United States and the pink bollworm in cotton in India) effective alternative Bt transgenic crops were no longer available, while for the fall armyworm, which is spreading rapidly across the world, only half of Bt maize crop varieties in Brazil were still effective (Tabashnik and Carrière 2017). Although there has been an initial delay in the development of resistance following the introduction of Bt crops, there has been a surge in new cases since the early 2000s (Figure 4.3.3-5).

Strict resistance management in Bt crops has been implemented, and is even mandatory, in some countries. The most widely applied approach is the high-dose refuge strategy, which combines high doses of the Bt toxins in the crop with growing a non-Bt crop on a certain fraction of the area (the "refuge"). This approach has been successful in delaying resistance

Figure 4.3.3-5 Global trend in pest resistance to Bt crops: (A) hectares planted with Bt crops (B) cumulative cases of field-evolved practical resistance to Bt crops. Tabashnik and Carrière (2017).



development. Many cases of pest resistance breaking down appear to have occurred where either refuge areas or toxin concentrations were insufficient (Huang, Andow and Buschman 2011; Tabashnik and Carrière 2017; Tabashnik and Carrière 2019). However, a recent review of resistance development of the fall armyworm (Spodoptera frugiperda) in Bt corn in the Americas suggests that current resistance management has been insufficient to avoid widespread development of resistance in this global pest (Huang 2020).

In contrast to the relatively regulated use of insect resistance, herbicide resistance has greatly increased with the adoption of herbicide resistant crops. The continuous use of glyphosate on glyphosate-tolerant crops in the United States quickly led to selection for weeds with evolved resistance to glyphosate (National Academies of Science, Engineering and Medicine 2016). More than 40 weed species are now reported to be resistant against glyphosate; many of them evolved in fields where herbicide resistant crops were grown (Heap 2019). To respond to this problem, companies have re-engineered crops to be tolerant to other herbicides, such as 2,4-D and dicamba, which have different modes of action to glyphosate. These herbicides were first commercialized in 1945 and 1967, respectively. It has been argued that there is a lack of knowledge about how to use these herbicides while avoiding the emergence of resistance (Gould, Brown and Kuzma 2018). The expected heavy use of 2,4-D and dicamba in herbicide resistant crops is likely to result in the development of further widespread weed resistance.

Only a few disease-resistant transgenic plant varieties have been commercially released (International Service for the Acquisition of Agri-Biotech Applications 2019). Limited experience therefore exists with diseases breaking the resistance of transgenic crops, and maintaining the durability of such disease resistance is likely to be a challenge (Nelson *et al*. 2018).

The impact of resistance development

Pesticide resistance in arthropods has repeatedly led to severe pest management problems, large increases in insecticide use, and even the total collapse of certain cultures. Examples from the past of pesticide resistance inhibiting crop growing include cotton production in Mexico (tobacco budworm) and Sudan (tobacco whitefly), rice in Southeast Asia (various pests) and vegetables in Asia and Africa (diamondback moth).

Glyphosate resistant weeds cost corn, cotton and soybean growers in the United States more than USD 1 billion per year (Frisvold, Bagavathiannan and Norsworthy 2017). Direct economic losses due to all types of resistance development have been estimated at USD 10 billion in that country alone (Gould, Brown and Kuzma 2018). It has also been calculated that pyrethroid resistance in mosquito vectors of malaria would imply about 26 million additional clinical cases of malaria per year in Sub-Saharan Africa (Gould, Brown and Kuzma 2018).

Nevertheless, the direct and indirect costs of resistance development, as well as its prevention, are rarely estimated in a systematic way when new pesticides are introduced or authorized.

The effects of pesticides on pollinators

Pollination is an ecosystem function that is fundamental to plant reproduction, agricultural production, and the maintenance of terrestrial biodiversity. Most of the world's wild flowering plants (about 88 per cent) are pollinated by insects and other animals. It is estimated that about one-third of the global food volume produced benefits from animal pollination (Eardley *et al*. 2016). The global annual market value of additional crop production directly linked to pollination services is estimated to be USD 235-577 billion (Gallai *et al*. 2016).

Pollinators can be found in different groups of animals, but are dominated by insects. Although pollinators are mainly bees, they also include some species of flies, wasps, butterflies, moths, beetles, weevils, thrips, ants, midges, bats, birds, primates, marsupials, rodents and reptiles. Nearly all bee species are pollinators, while a smaller (and variable) proportion of species within the other taxa are pollinators (Eardley *et al*. 2016).

By far most of the information about the adverse effects of pesticides on pollinators – resulting from controlled laboratory and (semi-) field experiments, as well as from monitoring – address honeybees. Increasingly, data are also being generated on other managed bees and on wild bees. Adverse pesticide effects on the other pollinators (e.g., flies, butterflies or bats) are not much studied, but information may be available from more general monitoring of such groups. The chapter below focuses on bees.

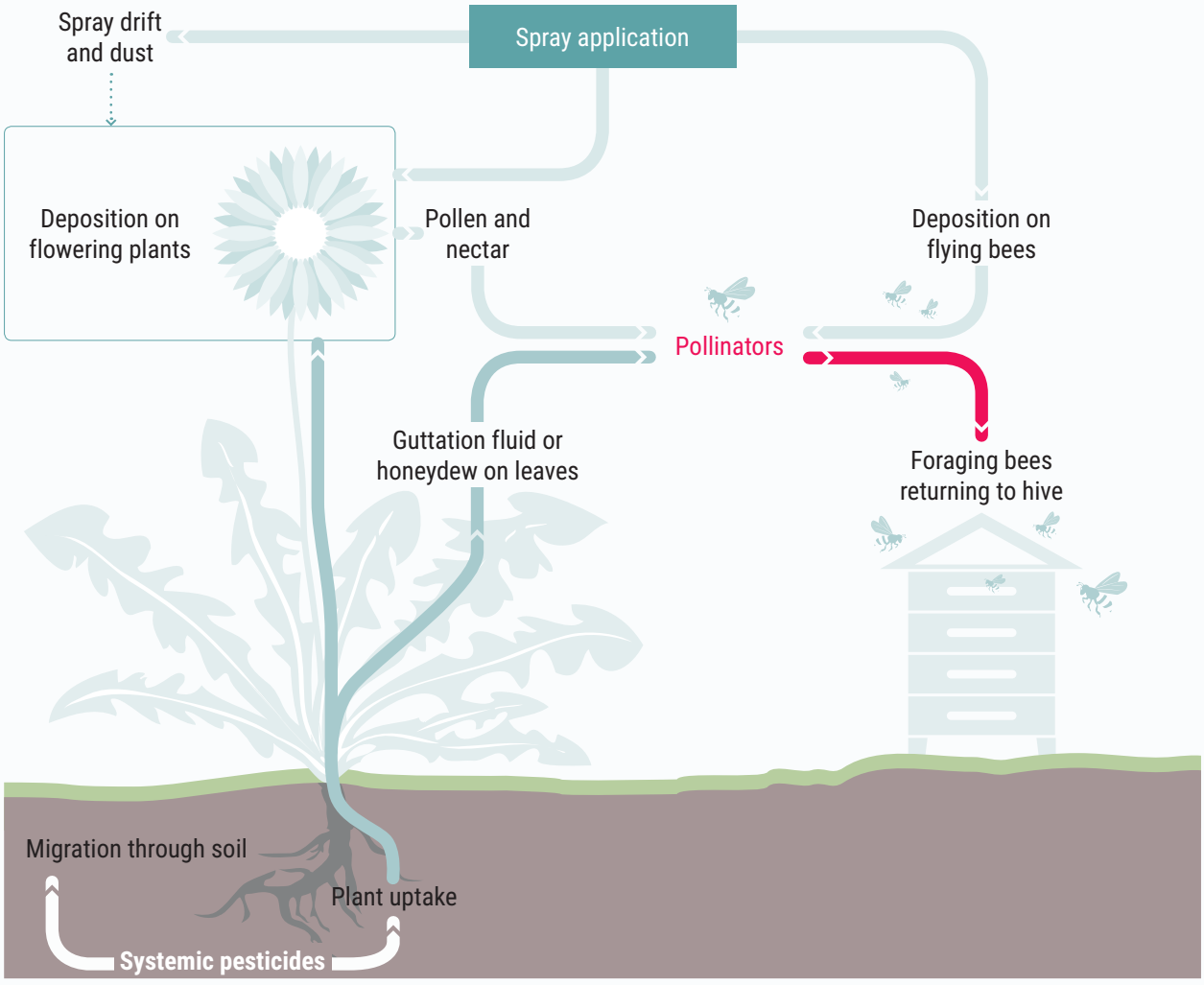
Susceptibility of bees to pesticides

Pesticides vary widely in their toxicity to bees. Some cause high acute mortality at very low doses, while others can be applied to crops without much adverse effect. Most insecticides are moderately or highly toxic to bees. However, other groups of pesticides (e.g., fungicides and herbicides) may also exert negative effects on bee health.

Bees can be exposed to pesticides in various ways, including overspraying, ingestion of contaminated pollen, nectar or honeydew, and contact with residues on foliage or flowers (Figure 4.3.3-6).

The direct adverse effects of pesticides on bees can be lethal (e.g., acute mortality immediately after applying a pesticides) or sublethal (e.g., reduced reproduction, increased susceptibility to pathogens, effects on learning or foraging by bees). Indirect effects of pesticides can occur if, for example, the use of herbicides

Figure 4.3.3-6 Main routes of exposure of honeybees to pesticides; similar routes of exposure are likely for other bees. Kovács-Hostyánszki *et al*. (2016).



eliminates weeds in a crop which is important for bee foraging (Kovács-Hostyánszki *et al*. 2016).

Adverse effects of pesticide use on bees

The most recent comprehensive assessment of the adverse effects of pesticides on pollinators in general, and bees in particular, was conducted under the auspices of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (Potts, Imperatriz-Fonseca and Ngo 2016). The conclusions of the IPBES assessment report are summarized below, complemented by more recent information when available.

Pesticides, notably insecticides, have shown a broad range of lethal and sublethal effects on bees in controlled experimental conditions (Kovács-Hostyánszki *et al*. 2016). The extent to which such adverse effects result in impairment of honeybee and bumblebee vitality or wild bee populations has been the object of much research, more recently for neonicotinoid insecticides in particular.

There is no doubt that the operational use of certain insecticides has caused acute bee mortality. Most data on acute pesticide effects under field conditions come from incident monitoring schemes in a number of European countries, Canada, the United States, Australia and Japan. They are almost entirely limited to the mortality of honeybees. Few incident reports exist for bumblebees or wild bees. The number of pesticide-induced incidents in Germany, the Netherlands and the United Kingdom, where such schemes are long-running, dropped by more than 75 per cent between the 1980s and the mid-2000s (Kovács-Hostyánszki *et al*. 2016). Reporting from Canada on the period 2007-2012 suggests that moderate and major bee poisoning incidents are also relatively rare (Cutler, Scott-Dupree and Drexler 2014). An exception is cases of dust drift during the drilling of seeds coated with neonicotinoid insecticides, which caused bee kills in Europe and North America in the late 2000s (Nuyttens *et al.* 2013).

The majority of incidents reported in Europe appear to have occurred when farmers did not use insecticides according to the instructions on the label (e.g., mistakenly applying the product on flowering crops, or while flowering weeds were present in the field). Better information, regulation and enforcement in these countries would probably have reduced the number of bee kill incidents. That suggests that in countries that do not have incident monitoring systems, or where there are no effective regulations or enforcement of mitigation measures, incidents of insecticide-associated honeybee mortality are likely to occur more regularly (Kovács-Hostyánszki *et al.* 2016). However, data to support this are lacking.

The IPBES assessment report evaluated nine published reviews of sublethal effects of pesticides on bees, three of which addressed pesticides in general and six the effects of neonicotinoid insecticides (the most recent reviews in the latter group were Godfray *et al*. 2015 and Pisa *et al*. 2015). Although the nine reviews overlapped in terms of the research included, their conclusions varied. There was agreement on the significant evidence for the adverse sublethal effects of neonicotinoids under controlled conditions. However, there were divergent views on the effects of pesticides under real-use field conditions, focusing on what constitute field-realistic doses of these insecticides that would lead to adverse effects on bees.

Kovács-Hostyánszki *et al*. (2016) therefore concluded in the IPBES assessment that how sublethal effects of pesticide exposure recorded for individual insects affect colonies and populations of managed bees and wild pollinators, especially over the longer term, was currently unresolved. The few available field studies assessing the effects of field realistic exposure provide conflicting evidence of effects, based on the species studied and pesticide used.

After the IPBES assessment report was published in 2016, Wood and Goulson (2017) reviewed evidence for environmental risks of neonicotinoids published between 2013 and 2016 (partly overlapping with the IPBES review). Their conclusions largely correspond to those of the IPBES assessment. However, the authors stated that new studies have provided stronger

indications of adverse effects of field-realistic doses of neonicotinoids on foraging behaviour, reproductive output and colony growth of bumblebees, as well as effects on reproduction of solitary bees.

More recently, the results of a number of field studies on the effects of neonicotinoids on bees were published (Table 4.3.3-2). These studies do not seem to fundamentally change the conclusions of Kovács-Hostyánszki *et al*. (2016) and Wood and Goulson (2017). Honeybee colony viability has been shown to be affected in some cases, but not in others. Field studies have shown varying effects of neonicotinoids on population parameters of wild bees. It remains ambiguous under which circumstances neonicotinoid insecticides adversely affect honeybee or wild bee populations in the field.

Table 4.3.3-2 Large-scale field studies on the effects on bees of seed treatments with neonicotinoids, not included in the review by Wood and Goulson (2017).

Country	Insecticide	Bee species	Crop	Observed effects	Reference
Germany	clothianidin + beta-cyfluthrin	Mason bee (Osmia bicornis)	Oilseed rape	No detrimental effects on the development or reproduction of mason bees was observed.	Peters, Gao and Zumkier (2016)
Hungary, Germany, United Kingdom	clothianidin, thiamethoxam	Honeybee (Apis mellifera) Bumble bee (Bombus terrestris) Mason bee (Osmia bicornis)	Oilseed rape	Honeybees: Both negative (Hungary and the United Kingdom) and positive (Germany) effects on colony viability were observed. In Hungary negative effects on honeybees (associated with clothianidin) persisted over winter and resulted in smaller colonies in the following spring (24 per cent declines). Bumble bees and mason bees: Reproduction was negatively correlated with neonicotinoid residues.	Woodcock *et al*. (2017)
France, Germany	thiamethoxam	Mason bee (Osmia bicornis)	Oilseed rape	Field and tunnel experiments. There were no significant effects from exposure to oilseed rape grown from thiamethoxam-treated seed from nest establishment through cell production to emergence under tunnel or field conditions. Oilseed rape contributed only 4 to 31 per cent of pollen provisions in the present study.	Ruddle *et al*. (2018)
Sweden	clothianidin	Honeybee (Apis mellifera)	Oilseed rape	Large fluctuations between and within years were found in honeybee colony development, (attempted) swarming/ supersedure, colony mortality, microbial composition and Varroa infestation. However, no negative effects of placement at the clothianidin-treated fields on these parameters were identified.	Osterman *et al*. (2019)
United States	acetamiprid, clothianidin, dinotefuran, imidacloprid, thiacloprid, thiamethoxam	Wild bees in field margins	Maize, soybean	Bee abundance was not affected by neonicotinoid insecticide concentrations in field margin soils. Neonicotinoid concentrations in margin soils were negatively associated with native bee richness.	Main *et al*. (2020)

The European Food Safety Authority (EFSA 2018b; EFSA 2018c; EFSA 2018d) conducted extensive regulatory reviews of the risks to honeybees, bumblebees and wild bees of seed and soil treatments with the neonicotinoid insecticides clothianidin, imidacloprid and thiamethoxam. They took into account laboratory studies, semi-field and field experiments, as well as monitoring results. EFSA concluded, for all three insecticides, that high risks exist to most or all groups of bees, and that low risks could not be excluded. Low risks were generally only concluded to exist for honeybees in certain specific crops. These risk assessments were an important reason the EU subsequently prohibited all uses of these insecticides except in greenhouses.

Weeds provide important, often exclusive, foraging resources for pollinators in agricultural landscapes. Their removal by physical means (e.g., tillage, mechanical weeding) or chemical herbicides can cause declines of native pollinators in agroecosystems (Kovács-Hostyánszki et al. 2016).

Pollinator declines and pesticides

There is no clear evidence that pesticides, particularly the neonicotinoid insecticides, have directly contributed to longer-term colony losses in the EU or the United States. Some studies have highlighted fungicides as affecting honeybee health adversely, but their role in colony loss has not yet been demonstrated. Colony loss appears to be a multifactorial issue, with different drivers varying in space and time (Kovács-Hostyánszki *et al*. 2016).

Few studies provide clear links between pesticide use and pollinator declines. Woodcock *et al*. (2016) conducted a correlational study in the United Kingdom which showed a positive association between use of neonicotinoid seed treatments in oilseed rape and reduced persistence of wild bee populations. This negative correlation was not observed for the application of foliar insecticides.

The IPBES assessment concluded that changes in land use or climate, intensive agricultural management and pesticide use, invasive alien species and pathogens can all affect pollinator health, abundance, diversity and pollination. Moreover, these multiple direct drivers have the potential to combine, synergistically or additively, leading to an overall increase in pressure on pollinators and pollination. It is rarely possible to rule out a single cause, such as pesticides, for changes in pollinator populations. A complex interplay of factors is likely to affect pollinator biodiversity and pollination, but the exact combination of factors will vary in space, time and across pollinator species (Kovács-Hostyánszki *et al*. 2016).

Natural enemies of pests

Natural enemies of crop pests or disease vectors include arthropod predators and parasitoids, as well as vertebrate predators such as insectivorous birds and reptiles. Pathogens (e.g., bacteria, fungi) can also attack pest populations. Natural enemies will often keep pest populations below levels that cause crop losses.

The greatest amount of information is available on the effects of pesticides on arthropod natural enemies. A pesticide treatment may control the pest population, but can also kill, repel, irritate or otherwise deter the natural enemies of that pest (Croft 1990; Cloyd 2012; Roubos, Rodriguez-Saona and Isaacs 2014). If the residual activity of the insecticide then expires, the same pest population will be able to increase more rapidly and to a higher abundance than when natural enemies were present. This is called target pest resurgence (Dutcher 2007).

A crop may contain other potentially injurious organisms which are normally harmless because they are kept in check by natural enemies. Secondary pest development can occur when a pesticide treatment causes the destruction of natural enemies of such organisms, which was regulated below an economic injury level by the natural enemies. The secondary pest will then be elevated to primary pest status (Dutcher 2007).

Resurgence and secondary pest development may not always be caused solely by the removal of natural enemies. Pesticide treatments can cause changes in a pests' behaviour, dispersal, development and fecundity, leading to an increase in pest populations (Dutcher 2007; Wu et al. 2020).

Resurgence and secondary pest development have various adverse consequences, including an increase in crop damage and potential yield losses; greater disease vector populations, resulting in possible rise of disease transmission and prevalence; disruption of biological control programmes; higher costs of pest management for additional or more expensive alternative chemical control or for alternative biological control; and an increase in pest abundance that carries over to the next growing season (Dutcher 2007).

Toxicity of pesticides to natural enemies

Until the mid-1960s observations of the effects of pesticides on natural enemies of crop pests tended to be incidental and generally part of pesticide efficacy trials. Studies on natural enemy responses to pesticides became more specific and were greatly expanded in the 1970s and 1980s, coinciding with the development of integrated pest management (IPM) as the recommended pest management approach in many countries (Croft 1990). Research on both the lethal and sub-lethal effects of pesticides on beneficial arthropods, as well as on the influence pesticides may have on pest and natural enemy interactions and population dynamics, have been conducted ever since. Regulatory testing of pesticides on natural enemies has also become part of the pesticide registration process, although this is limited mainly to Europe (Alix *et al*. 2012).

Given the importance of natural enemies for pest control and their susceptibility to many types of pesticides, the toxicity and selectivity of these products has been evaluated for a long time. In the mid-1980s Oregon State University (United States) established the SELCTV Database which, at the time, contained more than 12,000 records of pesticide effects on natural enemies (Croft and Theiling 1990). A decade earlier the Working Group on Pesticides and Beneficial Arthropods of the International Organization for Biological and Integrated Control (IOBC) started a joint testing programme in Europe on pesticides and beneficial arthropods (Franz et al. 1980). That work resulted in an on-line database on the selectivity of pesticides which includes both regulatory and academic information (Jansen 2013; International Organization for Biological and Integrated Control – West Palearctic Regional Section [IOBC] 2019). The IOBC database demonstrates that insecticides and acaricides are the most hazardous to arthropod natural enemies, but other groups of pesticides are not without risk. While some of the newer groups of insecticides are more target-specific and less hazardous for many arthropod natural enemies (e.g., diacylhydrazines, sulfloximines), this is not the case for all (e.g., neonicotinoids, diamides).

Biocontrol affected by pesticides

Pesticides have been widely studied and shown to have negative effects on natural enemy populations in many circumstances (Rusch *et al*. 2010). Many individual cases of pesticide-induced declines in biocontrol continue to be reported, as illustrated by the examples below.

Geiger *et al*. (2010) conducted a study in eight European countries on 270 cereal farms. They investigated the effects of 13 components of agricultural intensification on biodiversity, as well as on the potential for biological pest control of aphids. The use of insecticides adversely affected the potential for biocontrol of aphids in a consistent manner. Agro-environmental schemes using smaller amounts of pesticides had positive effects on biocontrol, but (somewhat surprisingly) this was not the case for organic farms.

Krauss, Gallenberger and Steffan-Dewenter (2011) compared aphid and predator densities in conventional versus organic triticale fields. They found that the abundance of cereal aphids was five times lower in organic fields, while their predator abundances were three times higher and predator-prey ratios were 20 times higher in organic fields, indicating a significantly higher potential for biological pest control in organic fields. Preventative insecticide application in conventional fields had only short-term effects on aphid densities, but long-term negative effects on biological pest control.

Ricci *et al*. (2019) conducted a large-scale study in three consecutive years in 80 fields of perennial and annual crops located in four French regions. They found that pesticide use intensity at local field level had clear negative effects on the

predation rates of pest insects. They suggested that the predation activity of a wide range of natural enemies (feeding on pests on the soil surface and on crop plants) was reduced when pesticide pressure was high.

Dainese *et al.* (2019) carried out a global synthesis of studies which measured richness and abundance of pollinators, pest natural enemies, and associated ecosystem services. They found clear evidence that the species richness of these organisms, as well as their abundance, positively influenced the delivery of ecosystem services such as pollination and pest control. They also found a positive correlation between natural enemy richness and crop production in areas not sprayed with insecticides. This link was not observed in sprayed areas, indicating that insecticide use undermines the full potential of natural pest control.

Resurgence and secondary pest development due to pesticides

In a first review on the topic, Ripper (1956) found that pesticide-induced resurgences in the field had been recorded in temperate, sub-tropical and tropical climates for over 50 species of phytophagous arthropod pests almost immediately after the introduction of broad-spectrum synthetic pesticides.

Examples of target pest resurgence and secondary pest development which have caused major agronomic and economic problems include: early season applications of pyrethroid and organophosphate insecticides, leading to resurgence of bollworm in the United States (Dutcher 2007); application of insecticides to rice, leading to outbreaks of brown plant hopper in Asia (Wu *et al*. 2020); and applications of non-selective insecticides such as pyrethroids, which greatly increased populations of diamondback moth, a major pest of cabbages and other brassica crops

Table 4.3.3-3 Examples of pest resurgence or secondary pest development induced by the use of pesticides.

Crop	Resurgent or secondary pest	Description	Reference
Cotton (United States)	Cotton bollworm (Helicoverpa armigera)	Early season applications with pyrethroids led to a resurgence of bollworm.	Dutcher (2007)
Cotton (United States)	Aphids, mites, armyworms	Early season application against Lygus plant bugs resulted in a 25 per cent increase in late season pesticide costs to control secondary pests.	Gross and Rosenheim (2011)
Cabbage and other brassicas (Africa, Asia)	Diamondback moth (Plutella xylostella)	Early application of non-selective insecticides is an important initiating factor in subsequent diamondback moth outbreaks because it reduces, among others, parasitoid populations.	Grzywacz *et al*. (2010)
Apple (North America, Europe)	Two-spotted spider mite (Tetranychus urticae)	Use of insecticides and fungicides in apple orchards kills predatory mites, leading to a resurgence of spider mites.	Hardman *et al*. (2006)
Cereals (West Africa)	Grasshoppers (Acrididae)	Control with broad-spectrum insecticides of early season grasshoppers augmented populations of late hatching grasshopper species.	van der Valk, Niassy and Bèye (1999)
Various arable crops in rotation (Australia)	Slugs (Milax gagates, Deroceras reticulatum)	Application of organophosphate insecticides against mites and aphids resulted in an increase of slug populations and associated yield reduction in canola.	Hill, MacFadyen and Nash (2017)
Rice (Asia)	Brown planthopper (Nilaparvata lugens)	Insecticide applications, especially in early season, killed generalist predators, resulting in an increase of planthopper populations.	Settle *et al*. (1996) Wu *et al*. (2020)

in the tropics and subtropics (Grzywacz *et al*. 2010) (Table 4.3.3-3)

4.3.4 The terrestrial environment

Soil organisms

Soil organisms provide essential contributions to many soil functions, such as nutrient cycling, soil formation, regulation of pests and diseases, and food, fibre and water supply, among others (Adhikari and Hartemink 2016; Food and Agriculture Organization of the United Nations and Intergovernmental Technical Panel on Soils [FAO and ITPS] 2017). Examples of major functions performed by soil organisms are shown in Box 4.3.4-1.

The Intergovernmental Technical Panel on Soils (ITPS) conducted a global review of the impact of plant protection products on soil functions and soil ecosystems (FAO and ITPS 2017). Other pesticides, such as biocides or domestic use products, were not considered. In its assessment the ITPS assumed that plant production products are applied on crops at locally relevant application rates. Point-source contamination through spills or leakage was not assessed. The conclusions of this review are provided below, along with the results of additional reviews and studies.

Soil microorganisms

Soil microorganisms, such as bacteria and fungi, can be important sources of plant disease that damage crops and reduce yields. However, they also play an essential role in organic matter decomposition and nutrient cycling, suppression of pathogens and pests, plant growth promotion and maintenance of soil structure (Box 4.3.4-1).

The ITPS (FAO and ITPS 2017) concluded that the use of plant protection products consistently resulted in measurable and statistically significant

Box 4.3.4-1 Selected examples of major functions performed by soil organisms. A more complete list is provided in FAO and ITPS (2017).

Function	Significance	Main soil organisms involved
Carbon and nutrient cycling		
Nitrogen fixation	Conversion of atmospheric nitrogen to a form available to plants	Bacteria
Nitrification	Conversion of immobile ammonium to mobile nitrate; production of nitrogen oxides	Bacteria
Decomposition of organic materials	Decomposition and degradation of plant residues	Bacteria, fungi, termites
Soil organic matter formation	Creation of stable forms of organic matter (humus)	Bacteria, fungi
Pesticide decomposition	Limits transfer of pesticides from farms to the broader environment	Bacteria, fungi
Creation and maintenance of soil structure		
Physical structure development	Creation of soil aggregates and porosity	Earthworms, potworms, soil arthropods
Transfer of nutrients and energy through the food web		
Nutrient release	Predation of bacteria, fungi, nematodes and consumption of readily decomposable organic matter	Protists, nematodes, mites, springtails, potworms, isopods
Food source for birds and other animals		Earthworms, insects, ants, termites, spiders
Pest control		
Pest control	Control of soil pests such as root-rot fungi, nematodes, grubs	Fungi, bacteria, actinomycetes

effects on soil microorganisms. These effects have led to both significant decreases and increases in attributes of soil microorganisms such as biomass, enzyme activity, respiration and species composition.

In a major review of published studies, Puglisi (2012) found that most often no significant difference was observed in microbial activity, abundance or biomass following the application of a pesticide (in a range of 25-55 per cent of cases, depending on the type of pesticide). In about one-third of cases, fungicides and herbicides resulted in a reduction of soil microbial activity or biomass; insecticides showed less adverse effects. The remaining studies demonstrated an increase in microbial activity or biomass caused by pesticides.

The structure of terrestrial microbial communities (e.g., species richness), on the other hand, was significantly affected in 80-95 per cent of cases, in the case of fungicides, herbicides or insecticides (Puglisi 2012). However, changes in structure do not necessarily imply a reduction in biodiversity or an adverse effect on soil functions owing to the very large number of microbial species present and the functional redundancy present in soil ecosystems (FAO and ITPS 2017).

There is only limited evidence that the observed effects of pesticides on soil microorganisms have led to significant and long-lasting decreases in soil functions. However, our understanding of the links between observed effects of pesticides on soil microorganisms on the one hand, and resulting changes in soil processes and functions on the other, is still inadequate (Imfeld and Vuilleumier 2012; FAO and ITPS 2017).

Earthworms and potworms

Earthworms and potworms (Oligochaeta, Annelida) are important soil fauna because they represent a large fraction of soil living biomass in many temperate and tropical ecosystems and play an important role in soil functioning. They actively participate in soil aeration, water infiltration and mixture of soil horizons, and influence organic matter decomposition and soil structure (Box 4.3.4-1). They are also an important source of food for many organisms like birds or moles (Pelosi *et al*. 2014). Earthworms have become a standard test and monitoring organism globally for the study of the environmental impact of pesticides and other contaminants.

There is considerable evidence for significant harmful effects of pesticides on earthworms (FAO and ITPS 2017). Based on their review, Pelosi *et al*. (2014) concluded that earthworms are impacted by pesticides at all organizational levels: they disrupt enzymatic activities, increase individual mortality, decrease fecundity and growth, change individual behaviour such as feeding rate, and decrease the overall community biomass and density, among others.

Soil fumigants generally have the greatest effects on earthworms, followed by fungicides, insecticides and herbicides in that order (FAO and ITPS 2017). Based on laboratory studies, Pelosi *et al*. (2014) concluded that the most harmful pesticides appeared to belong to the neonicotinoids, strobilurins, sulfonylureas, triazols, carbamates and organophosphates. The negative effects of copper-based fungicides on earthworms are also well established.

Vašíčková *et al*. (2019) evaluated the risks posed by residues of CUPs measured in 75 arable soils in the Czech Republic. They applied the EU risk assessment methodology for earthworms, potworms, springtails and soil mites as far as toxicity data were available. Their assessment shows that pesticide residues posed a risk to soil organisms at 35 per cent of the sites. Pelosi *et al*. (2021) also measured CUP residues in cereal fields, hedgerows and grasslands in France which were treated, untreated, or under organic farming practices (Annex 4.3.3). A high risk of chronic toxicity to earthworms was found in 46 per cent of samples, both in treated cereals and non-treated habitats considered to be refuges.

Römbke, Schmelz and Pélosi (2017) reviewed the effects of synthetic pesticides on enchytraeids (potworms) in agroecosystems. They found that very few pesticides have been studied intensively in both the laboratory and field, mainly the fungicide carbendazim. Most available data refer to organochlorine and organophosphate pesticides

which are rarely used while very few data are available on CUPs.

The effects of agricultural management practices in China and Europe on various soil quality parameters, including earthworm abundance, was recently evaluated by Bai *et al*. (2018). Most of the studies reviewed were long-term experiments. Their comparisons of no tillage and organic production systems with conventional production are of relevance to pesticide use. No-tillage refers to land cultivation with little or no soil surface disturbance, although herbicides are often used for weed control. No-tillage practices enhanced earthworm populations by a factor 1.7 unless herbicides or insecticides were applied. In organic agriculture few if any synthetic pesticides are used, although copper-based fungicides may be applied. Earthworm abundance almost doubled under organic agriculture when compared to conventional practices. Both comparisons illustrate the long-term adverse effects of pesticides on earthworms under realistic pesticide application scenarios.

No review was available of post-registration monitoring of earthworm populations following real pesticide applications at recommended rates. Pelosi *et al*. (2014) noted that studies based on realistic conditions in terms of soil, pesticide dose and experimental duration were lacking. The extent to which CUPs affect earthworm populations is therefore not well established, but the few studies from Europe cited above suggest that the risks may be higher than previously estimated.

Soil arthropods

Many arthropods inhabit soils, including springtails (or collembolans), mites and ants, and perform important functions (Box 4.3.4-1). Soil (micro) arthropods affect soil organic matter directly by fragmenting detritus, and indirectly by influencing microbial activity. Furthermore, soil arthropods can impact soil and plant health directly by feeding on pest organisms or serving as alternate prey for larger predatory arthropods (Neher and Barbercheck 2019).

Pesticide effects on soil arthropods are less well studied than on earthworms (FAO and ITPS 2017). A number of comparisons have been made of the sensitivity of soil arthropods to pesticides when compared to earthworms (Daam *et al*. 2011; Huguier *et al*. 2015; Kohlschmid and Ruf 2016). Partially different data sets were used, consisting mainly of laboratory toxicity studies. Certain variations were observed depending on pesticide, species and test conditions.

For insecticides and herbicides, no systematic differences in sensitivity between earthworms and soil arthropods were reported. Earthworms were generally found to be more sensitive than soil arthropods to fungicides. In comparison with other soil invertebrates (either earthworms or other arthropods), soil mites showed lower to similar sensitivities to chemicals (Huguier *et al*. 2015). Arachnids and isopods were more sensitive to insecticides, and nematodes to fungicides, as compared to earthworms.

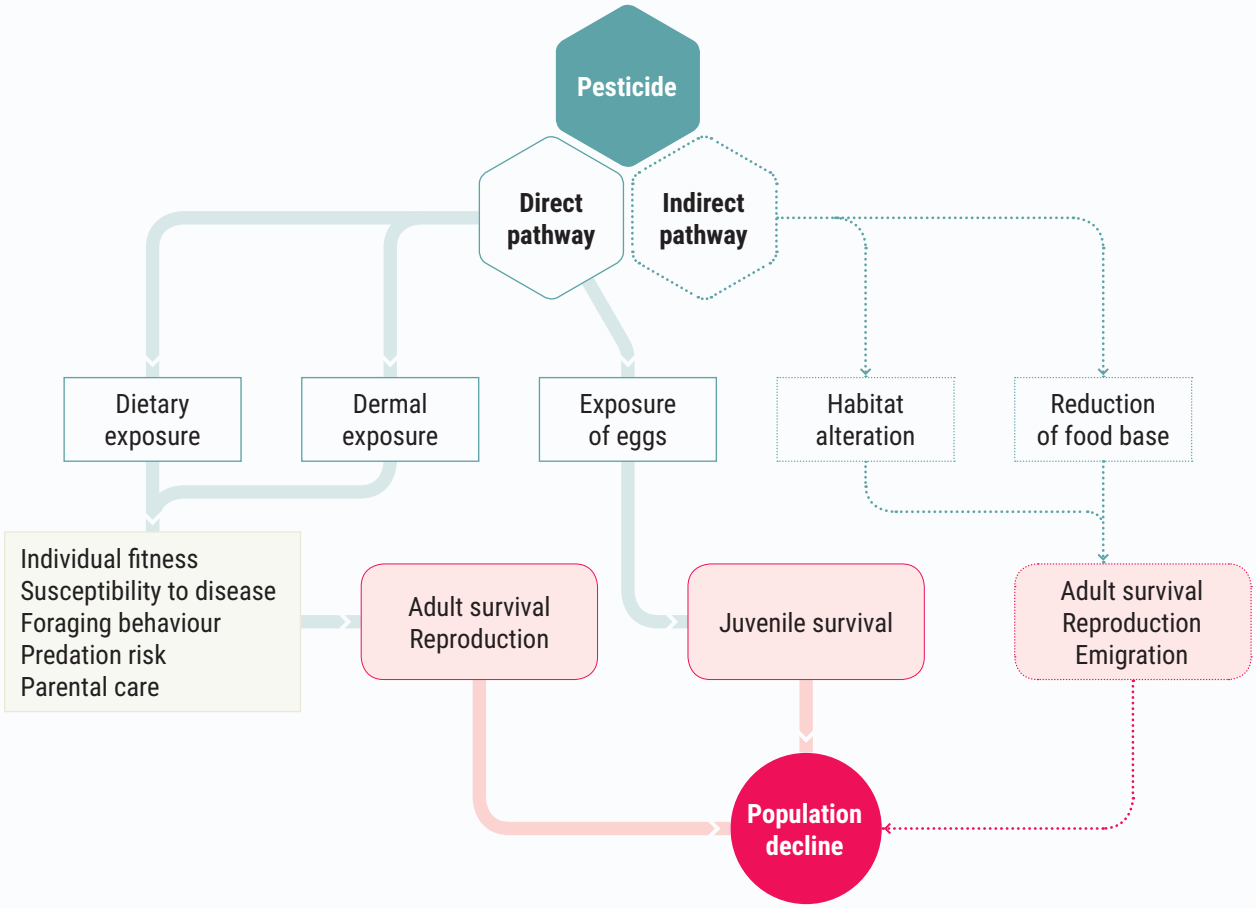
Since Jänsch *et al*. (2006), no review of semi-field and field studies of the effects of pesticides on soil arthropods seems to have been published. No global or regional evaluations were found of the monitoring of pesticide effects on soil arthropods.

Terrestrial vertebrates

Terrestrial vertebrates, such as birds, mammals, amphibians and reptiles, can be exposed to pesticides through different routes (Figure 4.3.4-1). Contact exposure occurs if animals are exposed to spray drift, or come into contact with pesticides on soil and other surfaces or in water (e.g., for the aquatic phase of amphibians). Pesticides can also be absorbed by the eggs of amphibians and reptiles in water or soil. Dietary exposure arises through consumption of contaminated food items (such as seeds, vegetation, invertebrate and vertebrate prey) or pesticide granules and soil, or through preening and grooming. Such exposure may cause mortality or sublethal adverse effects (e.g., on reproduction, immunity systems and behaviour) and ultimately result in population declines (EFSA 2009; Amaral *et al.* 2012a; Mineau 2013a; EFSA 2018a; Stanton, Morrissey and Clark 2018).

Pesticide use can also indirectly affect terrestrial vertebrate populations by reducing their food base

Figure 4.3.4-1 Pathways through which pesticide application can ultimately lead to population declines in terrestrial vertebrates. Based on Amaral *et al.* (2012a); Stanton, Morrissey and Clark (2018).



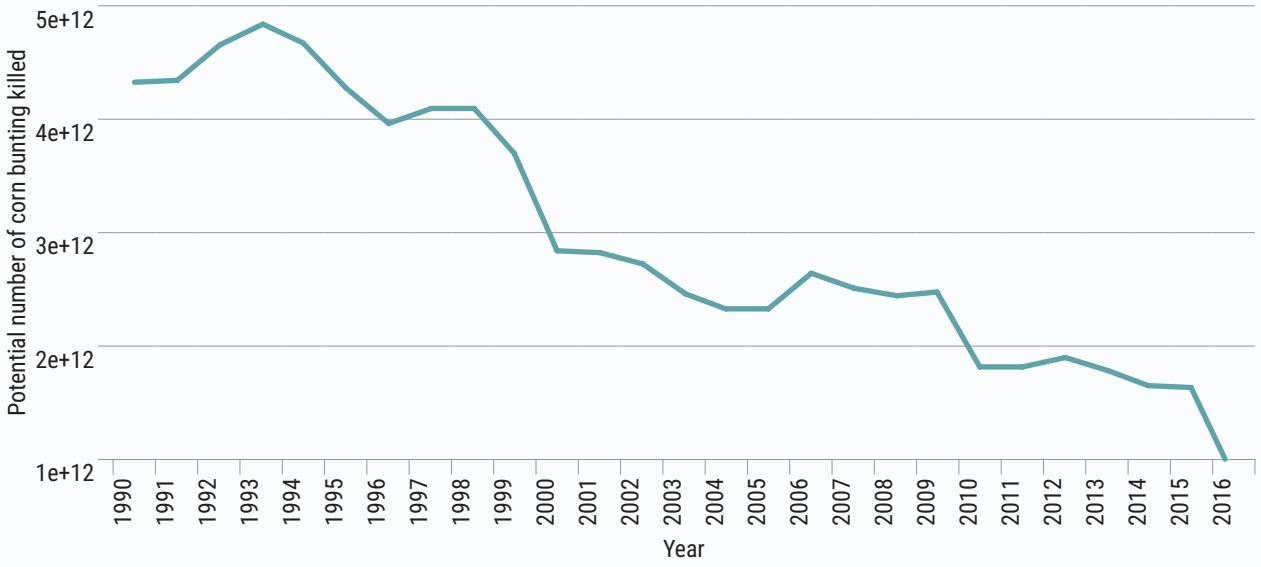
(e.g., if insecticides kill insect prey), or altering their habitat (e.g., if vegetative shelter is affected by herbicide use).

Most research into the effects of pesticides on terrestrial vertebrates published since the mid-1980s has focused on domestic and laboratory mammals and to a lesser extent on birds and amphibians. Köhler and Triebskorn (2013) indicated that less than 5 per cent of studies on terrestrial vertebrates concerned wild mammal species and reptiles. Based on the literature reviews conducted for this report, current knowledge about pesticide effects on terrestrial vertebrates appears to be skewed to a limited group of animals.

Historically, several distinct "periods" of pesticide effects on birds and other vertebrates can be recognized. Both acute lethal toxicity and effects on reproduction (e.g., eggshell thinning) were at the basis of these effects slowly (Nygård *et al.* 2019; Sonne *et al.* 2020). Between the 1950s and 1970s populations of raptors, fish-eating birds, seals and other vertebrates were severely affected, and sometimes completely collapsed, due to persistent organochlorine pesticides such as aldrin, dieldrin, chlordane, DDT and its metabolites, among others (Peakall 1993; Mineau 2013a; Matthiessen, Wheeler and Weltje 2018; Sonne et al. 2020). In many cases populations recovered after organochlorine pesticides were banned, albeit sometimes very slowly (Nygård *et al.* 2019; Sonne *et al.* 2020).

Subsequently, with the introduction of acutely toxic organophosphates and carbamates, direct mortality of, in particular, bird populations became problematic. With the stricter regulation of these groups of pesticides in certain regions of the world, effects on bird populations also appeared to

Figure 4.3.4-2 Potential kill of corn bunting due to the application of the 294 pesticides included in an analysis in Great Britain fell by about 80 per cent between 1990 and 2016. Tassin de Montaigu and Goulson (2020).



decline (Mineau and Whiteside 2006; Mineau and Whiteside 2013; Jahn *et al*. 2014).

The focus of the chapter below is on recent reviews, as well as on large-scale field studies looking at the effects of current use pesticides on populations of wild birds, amphibians and reptiles. No recent global or regional reviews were available concerning the effects of pesticides on wild mammal populations.

Birds

Recent reviews of pesticide effects on populations of birds, as well reports of large-scale monitoring of such effects are summarized in Annex 4.3-5.[7]

During the last two decades there does not appear to be much evidence of significant population effects arising from direct effects of pesticides on farmland birds (Jahn *et al*. 2014). Time trends between 1991 and 2003 show a decreasing risk of pesticide lethality to birds in several major crops in the United States (Mineau and Whiteside 2006). Similarly, Tassin de Montaigu and Goulson (2020), using acute toxicity data for the corn bunting as a model, estimated that the total toxic load for birds of pesticides applied in the United Kingdom fell by about 80 per cent between 1990 and 2016 (Figure 4.3.4-2). In both studies the decrease in risks was largely due to a reduction in the use of acutely toxic organophosphate and carbamate insecticides and their replacement by pesticides with lower vertebrate toxicity. The situation may be different in countries that have not yet removed pesticides with high bird toxicity from the market.

This does not mean that current use pesticides do not cause acute mortality in birds. In Canada pesticides were estimated to be the sixth source of bird mortality (out of 28 sources identified), resulting in 0.9-4.3. million deaths per year (Calvert *et al*. 2013; Mineau 2013b). Parsons, Mineau and Renfrew (2010), when reviewing the effects of rice cultivation on aquatic birds, reported direct mortality of many avian species as a result of pesticide applications, though mostly with "older" organochlorines, organophosphates and carbamates. There are also indications that certain neonicotinoid insecticides can cause acute mortality in granivorous birds following consumption of treated seeds at field-realistic doses (Gibbons, Morrissey and Mineau 2015; Wood and Goulson 2017).

7 **Annex 4.3-5 is found at:** https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing

Box 4.3.4-2 Effects of anticoagulent rodenticides on vertebrate wildlife.

Photo: © Martha de Jong-Lantink,Little owl - Steenuil (Athene noctua) CC BY-NC-ND 2.0

Rodents can be vectors of human or animal disease, damage crops, consume or foul stored food, and damage power supplies and electrical infrastructure. Rodenticides have been used for nearly a century to control rodent populations. They are also the most frequently used method to eradicate rodents from islands and fenced areas for the purpose of preserving or reintroducing native biodiversity (Lohr and Davis 2018; van den Brink *et al*. 2018a).

Most currently used rodenticides are anticoagulants, which prevent blood from clotting. First generation anticoagulants (e.g. warfarin, chlorophacinone) need to be consumed several times by rodents to cause mortality. Second generation anticoagulant rodenticides (e.g. brodifacoum, bromadiolone, difenacoum) have higher acute toxicities and are lethal after a single feed. They are also considerably more persistent in the animal's body, increasing the risk of bioaccumulation (Lohr and Davis 2018; Nakayama *et al*. 2019).

Recently, van den Brink *et al*. (2018b) brought together the available information on the environmental risks associated with rodent control with anticoagulent rodenticides (ARs). Primary exposure to ARs, mainly through ingestion of baits containing the rodenticide, has been widely reported for a range of non-target birds and mammals and, to a lesser extent, reptiles and insects. Primary exposure is thought to be more likely to cause acute mortality than secondary exposure. Effects of primary exposure to rodenticides on non-target populations have generally been found to be transient and populations recovered relatively rapidly. Exceptions, however, are situations in which baiting is permanent (e.g. in some urban settings or around farm buildings) or in areas where immigration is limited such as on islands (Shore and Coeurdassier 2018).

Secondary exposure to, and poisoning of, predators is caused by consumption of prey containing AR residues. i.e. of other animals that have been exposed due to bait ingestion (López-Perea and Mateo 2018). Secondary exposure of predators, particularly raptors, has been found to be widespread. Both López-Perea and Mateo (2018) and Nakayama *et al*. (2019) conducted global reviews of field monitoring studies and reported that AR residues were present on average in about 60 per cent of sampled predatory birds and mammals. Similar detection rates, ranging from 45 per cent to 90 per cent, were found in reptiles (Lettoof *et al*. 2020). In isolated environments such as islands, AR residues were detected in more than 80 per cent of sampled animals (Pitt *et al*. 2015). These data suggest that ARs are consistently and widely present in food webs of the ecosystems in which they are applied.

Many reports exist from around the world about mortality of non-target wildlife following AR exposure. Most commonly involved are bromadiolone and brodifacoum, both second generation ARs which are bioaccumulative and highly toxic. Sublethal effects of ARs are less well researched, but several studies have shown evidence of adverse effects of AR treatments on the breeding success of predators (López-Perea and Mateo 2018). Clear gaps exist, however, in knowledge about the long-term effects of exposure to ARs in populations of non-target organisms (Quinn 2019). Furthermore, despite the evidence of wide distribution of ARs in the environment as well as human consumption of wildlife, there is a distinct lack of information about the presence of AR residues in humans that is unrelated to pharmacological use (López-Perea and Mateo 2018).

Given the known risks of ARs to wildlife, their use has been regulated more or less strictly in many countries. Mitigation measures include restrictions on who can purchase and use these rodenticides, restrictions on where they can be used, requirements for protected bait stations, and removal of dead rodents and left-over baits. Such risk reduction measures are essential to limit primary and secondary exposure of non-target organisms to the rodenticides (Buckle and Prescott 2018). However, it has become clear that ARs travel beyond the point of application, through various means of primary and secondary exposure, and that these processes are largely outside of the control of the applicator. It has therefore been argued that mitigation measures based on the assumption that professional applicators will apply rodenticides more safely are likely to be limited in their effectiveness. A combination of measures will need to be developed and imposed for specific rodenticide use situations (Buckle and Prescott 2018; Quinn 2019).

Another group of current use pesticides causing bird mortality are second generation anticoagulant rodenticides, which are often implicated in wildlife poisoning, particularly of raptors. While there is much evidence that this group of rodenticides may kill large numbers of predatory birds, the extent to which they can cause long-term population declines is not established (Box 4.3.4-2).

With the acute risks of pesticides declining with the progressive introduction of pesticides with lower vertebrate toxicity, attention has shifted to sublethal effects, e.g., on bird physiological condition and behaviour (Eng *et al*. 2019), as well as indirect effects, e.g., through a reduction of food or shelter (Jahn *et al*. 2014).

Various recent studies and reviews have linked population declines of birds with the application of current use pesticides. A number of authors consider pesticides to be one of several factors that drive reductions in bird populations, e.g., Spiller and Dettmers (2019) for aerial insectivore birds in North America, Brain and Anderson (2019) for birds in North America, and Coates *et al*. (2017) for ring-necked pheasants in California (Annex 4.3-5).

Direct associations between pesticide use and bird population declines, or community composition have been found in several studies in both North America and Europe. Bouvier et al. (2011) observed that birds were significantly more abundant in French organic apple orchards than in IPM orchards, which in turn contained more birds than conventional orchards. They linked this abundance to levels of insecticide use. Chiron *et al*. (2014) found that herbicide use in cereal fields on France resulted in a shift in bird community composition from herbivore specialist to generalists. Mineau and Whiteside (2013) analysed surveys of grassland birds in the United States between 1980 and 2003 and found that the best predictors of species declines was lethal risk from insecticide use. Large-scale monitoring of breeding birds in cereal fields in eight European countries by Emmerson *et al*. (2016) indicated that applications of fungicides and insecticides were significantly associated with a lower total abundance of all breeding birds surveyed. In addition, a recent systematic review of drivers of farmland bird declines in North America found that pesticides were the predominant factor (42 per cent of all studies) negatively affecting bird populations (Stanton, Morrissey and Clark 2018) (Annex 4.3-5).

In recent years the use of neonicotinoid insecticides has received increasing attention. In their review of environmental effects of neonicotinoids, Gibbons, Morrissey and Mineau (2014) included only one field population study, which reported an effect of imidacloprid on three insectivore bird species in forests in the United States. However, more recently several studies have indicated adverse effects of neonicotinoid use on bird populations. Hallmann *et al*. (2014) found that higher concentrations of imidacloprid in surface water in the Netherlands were consistently associated with lower or negative population growth rates of passerine insectivorous birds. Ertl *et al*. (2018) assessed bobwhite quail population data from Texas (United States) for the period 1978-2012 and found that of the six predictor variables tested, the strongest negative association was between bobwhite abundance and neonicotinoid use. Li, Miao and Khanna (2020) estimated that the use of neonicotinoids in the United States resulted in annual decreases of 4 per cent and 3 per cent in population abundance of grassland birds and insectivorous birds, respectively, between 2008 and 2014 (Annex 4.3-5).

Amphibians

Amphibians are the most endangered group of vertebrate animals, with 41 per cent of currently existing species estimated to be threatened (International Union for Conservation of Nature 2020). Many causes for this decline have been cited, including habitat loss, emerging infectious diseases, invasive species, climate change and chemical pollution (Araújo *et al*. 2014; Whitfield, Lips and Donnelly 2016). The use of pesticides also figures explicitly as one of the drivers for observed declines in amphibian abundance and diversity (Hayes *et al*. 2010; Bishop *et al*. 2012; Brühl *et al*. 2013). Amphibians are particularly numerous in tropical and subtropical regions (Whitfield, Lips and Donnelly 2016). Some amphibians and reptiles inhabit agricultural

landscapes, either as residents or migrating through, and can thus be exposed to agricultural pesticides. Amphibians may also breed in water bodies adjacent to agricultural fields.

The state of the science on pesticide risk assessment for amphibians and reptiles was recently reviewed by the European Food Safety Authority (EFSA 2018a). Based on a limited comparison of the acute and chronic sensitivity of amphibians with the standard fish test species, they suggested that the sensitivity of amphibians may be covered by fish in some, but not all, cases. However, an additional extrapolation factor may be warranted when using fish toxicity data in amphibian risk assessments. EFSA (2018a) further concluded that recent evidence from both field and laboratory studies indicates that the use of plant protection products poses a risk to the reproduction and survival of amphibian field populations. In particular, studies on terrestrial stages of amphibians have shown that the use of currently registered pesticides at authorized rates can cause mortality in frogs and toads. They noted that existing risk assessment procedures for pesticides likely do not protect amphibians to a sufficient degree and further research is needed, especially taking into account the current endangered status of many of these animals (EFSA 2018a).

Most studies on the effects of pesticides on amphibians are experimental and have been conducted in the laboratory. Unlike studies on birds and aquatic fauna, for example, few studies have explored the effects of pesticides on amphibians under field conditions (Annex 4.3-5). Abnormalities and effects on biomarkers have most often been reported, but unequivocal links to pesticide field exposures have been difficult to make as other stressors might have caused similar effects (Agostini *et al*. 2020). Therefore, the strongest indications of impact of pesticides on amphibian population health currently come from reviews which have linked laboratory studies of pesticide effects to realistic concentrations of such compounds in the field (Annex 4.3-5).

Egea-Serrano *et al*. (2012) and Baker, Bancroft and Garcia (2013) conducted meta-analyses of laboratory studies of the effects of pesticides on survival and physiological parameters of amphibians at concentrations expected to be encountered in the environment after spray events. Both concluded that pesticides adversely affected amphibian survival at environmentally realistic pesticide exposure levels. When European common frogs were oversprayed with various pesticides at recommended application rates, high acute mortality was observed even though these products were registered in one or more EU countries (Brühl *et al*. 2013).

Baker, Bancroft and Garcia (2013) identified carbamate, organophosphate and neonicotinoid insecticides, as well as triazine and phosphonoglycine herbicides, as negatively affecting amphibian survival. Organophosphate insecticides and phosphonoglycine herbicides also reduced amphibian growth. On the other hand, developmental time and frequencies of abnormalities were not found to be significantly affected by pesticides (Egea-Serrano *et al*. 2012).

Some of these effects were confirmed in a recently published field study from Argentina, in which Agostini *et al*. (2020) monitored the effects of five pesticides, applied according to farmer practices, on amphibian populations in 91 ponds adjacent to agricultural fields. They observed almost complete mortality of amphibians in the ponds after use of the insecticides cypermethrin, chlorpyrifos and endosulfan, as well as effects on amphibian mobility from the herbicides glyphosate and 2,4-D.

Schiesari and Corrêa (2016) evaluated the consequences of Brazilian sugarcane production for freshwater biodiversity including amphibians. To this end, they conducted field surveys across a gradient in land-use intensity ranging from seasonal Atlantic Forest and the Cerrado to pastures to sugarcane plantations in Southeast Brazil. They observed a reduction in amphibian diversity on sugarcane plantations. This was associated with habitat loss, intensification, and use of agrochemicals. Schiesari and Corrêa (2016) asserted that tadpole dieoffs in ponds adjacent to sugarcane fields could plausibly be explained by pesticide applications. This was confirmed in laboratory toxicity tests evaluating realistic exposure levels of the main pesticides used in the sugarcane fields (Moutinho *et al*. 2020).

Of particular concern have been the effects of endocrine disrupting chemicals (EDCs) on amphibians, which exert adverse effects through perturbations of hormonal systems (Box 4.3.4-3). In a review of the effects of EDCs on amphibian reproduction and development, Orton and Tyler (2015) concluded that many pesticides have the ability to alter hormone systems and affect reproductive development and function in frogs at relatively high concentrations. The herbicide atrazine was also found to exert such adverse effects at concentrations measured in some aquatic environments. They noted that there is now a substantial and growing body of evidence from field studies indicating that agriculture, herbicides and/or pesticides in general are associated with increased intersex in anurans. Caution was expressed, however, that the lack of analysis of the associated concentrations of these chemicals in anuran breeding habitats and/or body tissues makes assigning cause-effects relationships to individual pesticides difficult.

Pesticides may also affect amphibians in conjunction with other stressors. Emerging infectious diseases, such as chytrid fungi and ranaviruses, have been identified as important drivers of amphibian population declines (Bishop *et al*. 2012; North *et al*. 2015). Pesticides have been shown in the laboratory to increase the susceptibility of amphibians to diseases and indirectly to increase infection rates, leading to a reduction in survival or increased mortality due to predation (North *et al*. 2015, Bienentreu and Lesbarrères 2020; Campbell, Pawlik and Harrison 2020). This relationship appears to be reciprocal and the presence of a ranavirus has been shown to exacerbate the lethality of pesticides (Campbell, Pawlik and Harrison 2020).

Fewer studies have looked at the relationship between exposure to specific pesticides and pathogen infections in amphibians. North *et al*. (2015) found that Ranavirus prevalence in common frogs increased with the use of herbicides and slug pellets in British gardens. Another study conducted across the United States (Battaglin *et al*. 2016) showed a positive correlation between the prevalence of the fungus Batrachochytrium dendrobatidis in amphibian hosts and total fungicide concentrations in the environment.

Reptiles

About one-third of currently existing chameleons, crocodiles and alligators, marine turtles and sea snakes are considered to be threatened with extinction (IUCN 2020). Several factors contribute to local and regional declines in reptile populations, including habitat loss, unsustainable removal (e.g., for skins and traditional medicines), climate change, invasive species (e.g., exotic predators or competitors for resources), diseases and parasites, as well as pollutants including pesticides (Todd *et al*. 2010).

Reptiles exhibit various ecological and life history characteristics that make them particularly vulnerable to pesticides. With the exception of a few lizard and turtle species, they are carnivorous and many occupy high trophic positions within food webs. As a result, they are at risk of high pesticide exposures from biomagnification through the food chain. Many reptiles are also long-lived and have small habitats, making them susceptible to long-term pesticide exposure (Todd *et al*. 2010). According to Köhler and Triebskorn (2013), however, less than 1 per cent of published pesticide effect studies concerned reptiles, making them one of the least studied groups of non-target organisms. This lack of data holds true for laboratory toxicity studies, but even more so for evaluations of causal relationships between pesticide use and reptile population declines (Wagner *et al*. 2015).

While studies regarding the ecotoxicological effects of pesticides on reptiles are scarce, lethal as well as sub-lethal effects have been observed. These include hormonal changes and enzymatic responses, oxidative stress, neurotoxic implications and immunosuppression, fertility, development and locomotor performance impairments, and hermaphroditism (Mingo, Lötters and Wagner 2017).

No recent comprehensive review has been published on pesticide toxicity to reptiles or pesticide effects on reptile populations in the field. In their review of the state of the science on pesticide risk assessment for amphibians and reptiles, EFSA (2018a) concluded that there is sufficient evidence from both field and laboratory

Box 4.3.4-3 Endocrine disruption and wildlife.

An endocrine disrupting chemical (EDC) has been defined as "an exogenous substance or mixture that alters function(s) of the endocrine system and consequently causes adverse health effects in an intact organism, or its progeny, or (sub) populations" (UNEP and WHO 2012). The endocrine system is a complex network of glands that release hormones into the blood stream. They control growth, development, reproduction, metabolism and immune responses, among other functions. EDCs can mimic or antagonize natural hormones and thus interact with hormone receptors, which can disrupt the body's normal functions. Some of the observed health effects associated with EDCs include cancers as well as reproductive, developmental, immunological and neurological disorders (UNEP and WHO 2012; UNEP 2017b).

Much research has been conducted into the mechanisms of endocrine disruption, identification of EDCs, and linkages between EDCs and the presence of adverse effects in human and animal populations. However, despite substantial advances in scientific understanding of EDCs, important uncertainties and knowledge gaps still exist. Even after several decades of research on EDCs, there is no consensus among scientists, regulators, the chemical industry and civil society groups about the outcomes and implications of such research (Encarnação *et al*. 2019; European Parliamentary Research Service 2019; La Merrill *et al*. 2020).

The latest global reviews of the effects of EDCs on human health and the environment were published almost a decade ago by UNEP and WHO (UNEP and WHO 2012; Bergman et al. 2013) and by the European Environment Agency (EEA 2012). Both reviews express serious concerns about the possible effects of EDCs on human health and wildlife: Many chemicals are known or suspected to be capable of interfering with hormone receptors, hormone synthesis or hormone conversion; human and wildlife populations are exposed to EDCs globally; and chemically induced endocrine disruption likely affects human and wildlife endocrine health the world over (UNEP and WHO 2012; EEA 2012).

Identifying which chemicals are likely to affect endocrine systems has been the subject of substantial research and screening. This is partly due to the scientific complexity of endocrine disruption. Consequently, different scientific and regulatory bodies apply different approaches and criteria (Slama *et al*. 2016; UNEP 2017b; European Chemical Agency and European Food Safety Authority 2018; La Merrill *et al*. 2020). UNEP (2017b) listed 28 initiatives to identify EDCs which could aid in overcoming this problem.

Pesticides as EDCs

There is currently no agreed global list of pesticides identified as EDCs or potential EDCs. Lists, established by regulatory bodies, tend to be variable. The International Panel on Chemical Pollution (IPCP) lists six pesticides (metam-sodium, zineb, ziram, thiram, tebuconazole and pentachlorophenol [PCP]) that were identified as EDCs following a publicly accessible, thorough scientific assessment using the World Health Organization and International Programme on Chemical Safety (WHO and IPCS) 2002 definition of EDCs and with multi-stakeholder involvement (UNEP 2017b). In 2016 the European Commission identified between 15 and 50 pesticide active ingredients as potential EDCs, out of a total of about 600 authorized compounds, depending on which criteria were followed (European Parliamentary Research Service 2019). Other regulators have identified a larger number of chemicals or chemical groups as potential EDCs or included them in current screening programmes (UNEP 2017b). It is very likely that with further screening and increased scientific understanding, more pesticides will be identified that with high likelihood can disrupt the endocrine system and affect wildlife populations (UNEP and WHO 2012).

Newer pesticides are less well understood in terms of their endocrine disrupting potential (Warner *et al*. 2020). There is ample evidence of mechanisms by which current use pesticides can disrupt endocrine systems in animals such as mammals (Pelch *et al*. 2011; Warner *et al*. 2020), fish (Martyniuk, Mehinto and Denslow 2020) or amphibians (Orton and Tyler 2015; Trudeau *et al*. 2020). However, there is less evidence of such pesticides causing adverse effects on wildlife due to endocrine disruption in the field or at environmentally realistic concentrations. This is partly due to the complexity of measuring such effects under realistic field conditions.

Endocrine disrupting effects of current use pesticides tend to be highly debated, in part due to the large economic value attributed to the use of some of them. For instance, reviews of the endocrine disrupting effects of atrazine on, in particular, amphibians have resulted in contrasting conclusions. Rohr and McCoy (2010), Hayes *et al*. (2010), Hayes *et al*. (2011) and Orton and Tyler (2015) concluded that atrazine does cause endocrine disruption in certain amphibians at environmentally realistic concentrations, but others have opposed that view (Matthiessen, Wheeler and Weltje 2018, Hanson et al.2019). Gendron (2013) noted that these contrasting conclusions to a large extent positioned pesticide industry against certain research groups.

However, it is generally accepted that organochlorine and organotin pesticides have endocrine disrupting properties and that they have caused adverse effects on the growth and reproduction of wildlife populations (e.g., raptors, seals, fish, molluscs). The evidence that such pesticides cause wildlife population declines is reinforced by the recovery of some of these populations following bans or restrictions on their use (UNEP and WHO 2012; EEA 2012; Matthiessen, Wheeler and Weltje 2018; Encarnação et al. 2019; Plaza, Martínez-López and Lambertucci 2019; Martyniuk, Mehinto and Denslow 2020).

Box 4.3.4-4 **Effects observed on reptiles as a result of insecticides applied for locust control in Africa and Australia are highly dependent on the type of insecticide and the way it is applied.**

© Wildlife Wanderer,Western Sahara Fringe-toed Lizard(Acanthodactylus aureus), Western Sahara

Country (type of study)	Insecticide	Reptile	Observed effects	Reference
Mauritania (laboratory)	*Metarhizium anisopliae* var. *acridum* fipronil	*Acanthodactylus* spp. (lacertid lizard)	*M. anisopliae* var. *acridum* was considered to pose a low risk under field conditions. Acute risk of exposure to fipronil was considered high.	Peveling and Demba 2003
Madagascar (field)	fipronil (as full cover treatment) deltamethrin triflumuron	*Chalarodon madagascariensis* *Mabuya elegans* (lizards)	Significant declines in abundance of the two lizard species were observed in fipronil treated zones but not after deltamethrin or triflumuron treatments. Fipronil induced food shortages were considered the principal cause of the decline in lizards.	Peveling *et al*. 2003
Australia (laboratory)	fenitrothion	*Pogona vitticeps*	Oral doses expected to occur after locust control did not cause mortality and only minor poisoning symptoms. Except for a reduction in plasma cholinesterase, no sublethal effects were observed.	Bain *et al*. 2004
Australia (field)	fipronil (as barrier treatment) *Metarhizium acridum*	22 species	Neither reptile abundance nor community composition were significantly affected in the short term by the treatments.	Maute *et al*. 2015
Niger (field)	chlorpyrifos fenitrothion	*Acanthodactylus* spp. (lacertid lizard)	Significant reductions were found in the abundance of lizards in the case of both insecticides until about 30 days after treatment. Moribund and dead lizards were observed between nine and 21 days after treatment.	Abdou 2015

studies indicating that the use of plant protection products poses a risk to reproduction and survival of reptile populations. Selected studies published during the last decade are listed in Annex 4.3-5.

Wagner *et al*. (2015) and Mingo, Lötters and Wagner (2016) assessed the risk of pesticide exposure to reptiles in Europe, based on the occurrence of reptiles in agricultural areas where pesticides are used, reptile physiology, and their life

history. They concluded that about one-third of all European reptile species are at high risk of current exposure to pesticides, especially in Southern Europe. Many of these species are considered threatened and/or are protected in the region.

The few recent field studies concerning pesticide effects on reptile health or abundance in Europe and Oceania do not show short-term effects on population sizes (Annex 4.3-5). However, physiological and endocrinological effects have been observed following the application of herbicides, in some cases also resulting in adverse effects on the fitness of the organisms (Amaral *et al*. 2012a; Amaral *et al*. 2012b; Bicho *et al*. 2013; Mingo, Lötters and Wagner 2017).

The effects of insecticides on reptiles have received particular attention in locust control, which is often carried out in arid and semi-arid ecosystems where reptiles are an important part of vertebrate fauna (Box 4.3.4-4). While biological control with the entomopathogen Metarhizium acridum did not cause unintended effects under field conditions, treatments with organophosphate insecticides did. Furthermore, locust control with fipronil, when applied as full cover sprays, resulted in indirect effects on lizard populations through insect food shortage. However, when it was applied as a (partial) barrier treatment, such effects were not observed.

Deliberate wildlife poisoning

The use of poisons to kill wildlife has a long history the world over. Highly toxic synthetic pesticides have become the preferred tool for deliberate wildlife poisoning, as they are silent, cheap, easy to obtain and use, and effective (Ogada 2014). Most information about wildlife poisoning is available from Africa, Europe and to a lesser extent Asia and the Americas (Plaza, Martínez-López and Lambertucci 2019).

In parts of Europe the deliberate use of pesticides to kill wildlife is a common practice, mostly associated with human-wildlife conflicts with predators, presumed pest species, feral dogs and birds of prey that compete with hunters and poachers (Guitart *et al*. 2010). Common reasons for wildlife poisoning in Africa are control of damage-causing animals, harvesting fish and bushmeat for human consumption, and harvesting animals for traditional medicine. Pesticides are also increasingly used in poaching elephants for ivory, rhinos for horn and carnivores for fur, as well as in killing wildlife sentinels (e.g., vultures because their aerial circling alerts authorities to poachers' activities) (Ogada 2014; Ogada, Botha and Shaw 2016; Aziz *et al*. 2017).

Deliberate poisoning incidents with pesticides across Africa have been on the increase since the 1990s, resulting in an unsustainable number and diversity of African wildlife being killed. There is substantial evidence of corresponding population declines in lions, raptors, large mammals, vultures and hyenas (Ogada 2014; Richards *et al*. 2018). In Europe wildlife poisoning may have considerable implications for endangered species (Guitart *et al*. 2010; Grilo *et al*. 2021).

Of particular concern are vulture and condor species, of which about 70 per cent are threatened by human activities. Plaza, Martínez-López and Lambertucci (2019) concluded that the most important threat currently affecting vultures and condors worldwide is probably exposure to pesticides, both accidentally and through deliberate abuse. Ogada *et al*. (2016) came to a similar conclusion in regard to African vulture populations. When reviewing vulture mortality cases in 26 African countries, they found that poisoning, mainly by pesticides, was the cause of death in 60 per cent of cases. Vultures were deliberately poisoned or were unintentional victims when they consumed carcasses baited with highly toxic pesticides to kill carnivores such as lions, hyenas and jackals. It has been argued that if poisoning with pesticides is not stopped, this threatened avian group could become extinct very soon (Plaza, Martínez-López and Lambertucci 2019).

The pesticides most widely used to poison wildlife are organophosphates and carbamates with high acute toxicity, such as aldicarb, carbofuran, methomyl, monocrotophos, diazinon, parathion and fenthion. In recent years the pesticide most commonly used to deliberately kill wildlife has been carbofuran (Guitart 2010; Ogada 2014; Plaza, Martínez-López and Lambertucci 2019)

It should be emphasized that using pesticides to poison wildlife is illegal in most countries. The wildlife laws in 38 of 46 African countries specifically mention that it is illegal to use poison, poison bait, or poisoned weapons for the purpose of hunting wildlife (Ogada 2014). However, in many countries, especially low and middle income countries (LMICs), there is inadequate enforcement of such legislation and highly toxic pesticides are easily available even if their use is legally restricted. Proposed measures to mitigate this abuse of pesticides include banning the pesticides most used for wildlife poisoning (preferably by groups of neighbouring countries in order to minimize smuggling), stricter regulation, control of distribution and enforcement, higher penalties, and targeted educational programmes (Ogada 2014; Plaza, Martínez-López and Lambertucci 2019).

Terrestrial arthropods

Pesticides can have an adverse impact on populations of terrestrial arthropods such as insects, mites and spiders. Pesticide exposure may result in lethal and sublethal effects in arthropods, but can also indirectly modify their food base or their environment. Insecticides and acaricides are most often implicated due to their modes of action, but fungicides and herbicides can also perturb terrestrial arthropod populations.

The effects of pesticides on various specific groups of terrestrial arthropods are discussed elsewhere in this report: pollinators in Chapter 4.3.3; natural enemies of pests in Chapter 4.3.3; and soil arthropods in Chapter 4.3.4. This section covers the effects of pesticides on the population abundance or biomass of arthropods in general, as observed in the field. Pesticides' impact on arthropod biodiversity (i.e., changes in number of taxa, functional groups, genetic composition, or other diversity parameters) are reviewed in Chapter 4.3.6. While changes in the abundance or biomass of specific arthropod taxa may correlate with changes in their taxonomic diversity, this is not necessarily always the case.

Reports of significant declines in arthropod populations in different parts of the world have recently received much attention. Most of them focus on insects (Leather 2018; Sánchez-Bayo and Wyckhuys 2019; Cardoso *et al*. 2020; Montgomery *et al*. 2020; Wagner 2020). It is not always clear whether the insect declines discussed refer to population abundance, biomass or species diversity. Sometimes these terms are used interchangeably.

The most recent worldwide meta-analysis of trends in insect abundance and biomass was conducted by van Klink *et al*. (2020), who evaluated 166 long-term insect surveys across 1,676 sites

Table 4.3.4-1 Recent studies in the United Kingdom associating pesticide use with trends in terrestrial arthropod population abundance or biomass.

Scope (study period)	Study method	Pesticide	Arthropods	Findings	Reference
Sussex (1970-2004)	Monitoring of ~100 cereal fields/year	Insecticides, fungicides, herbicides	26 taxonomic groups	Abundance of Coleoptera and Araneae declined with increasing total pesticide use. Trends in abundance of other groups of arthropods were more associated with weather parameters.	Ewald *et al*. (2015)
National (1985-2012)	Population abundance indices from the Butterfly Monitoring Scheme	Neonicotinoid insecticides	17 butterfly species	Area of farmland treated the previous year with neonicotinoids was negatively associated with butterfly abundance indices Abundance Indices for 15 of 17 species show negative associations with neonicotinoid usage.	Gilburn *et al*. (2015)

in 41 countries. They found strong evidence for a decline in terrestrial insect abundance (estimated to be almost 11 per cent per decade), particularly in North America and Europe. Insect abundance trends were negatively associated with urbanization and positively with degree of crop cover. No evaluation of the effects of agricultural intensification or pesticide use was made.

Wagner (2020), in his review of studies on insect declines, did find that these declines have been linked to agricultural intensification and to insecticide use. However, the number of studies that explicitly assess the effects of current insecticide use on the abundance or biomass of arthropods – other than pollinators or natural enemies of pests – is very small. A few studies in the United Kingdom have associated the use of pesticides with decreasing abundances of beetles, butterflies and spiders (Table 4.3.4-1).

Table 4.3.4-1 Recent studies in the United Kingdom associating pesticide use with trends in terrestrial arthropod population abundance or biomass.

It may seem clear that widespread pesticide, particularly insecticide, use would result in a decline in the abundance or biomass of terrestrial arthropod populations beyond the boundary of treated fields. However, very little field evidence appears to be available to confirm or disprove such a hypothesis. On the other hand, more evidence has been published indicating a decline in the diversity (e.g., number of species) of terrestrial arthropods which was associated with the use of pesticides (Chapter 4.3.6).

4.3.5 The aquatic environment

As discussed in Chapter 4.3.2, pesticides applied in agricultural fields may reach edge-of-field water bodies via spray drift and run-off, among other entry routes. Subsequently, pollution of these water bodies may result in toxic effects on aquatic biota and ecosystem functioning (Schäfer, van den Brink and Liess 2011).

Current pesticide toxicity assessments largely rely on laboratory bioassays and semi-field experiments in model ecosystems (i.e., microcosms and mesocosms) (Schäfer 2019). Field studies evaluating the effects of pesticides on aquatic ecosystem structure and functioning are not frequently conducted. The high labour intensity, costs, spatial-temporal variation and difficulty of establishing causal dose-effect relationships are the most common reasons indicated for this low research effort using field testing and monitoring (Schäfer 2019; Zubrod *et al*. 2019; Rosic *et al*. 2020; Schepker *et al*. 2020).

Risk as a function of aquatic pesticide concentrations

Given the difficulty of conducting field studies and monitoring real-time pesticide use, an often applied method for evaluating the risks of pesticides to aquatic organisms is to compare field-measured pesticide concentrations with environmental or regulatory threshold values. These threshold values are usually set by using toxicity data derived in the above mentioned laboratory bioassays and/or semi-field studies.

Global and regional assessments of pesticide concentrations in freshwater ecosystems have been reviewed in Chapter 4.3.2. Such monitoring is heavily biased towards North America and Europe, with relatively little information available from other parts of the world. Whenever data have been available, however, measured pesticide concentrations have often exceeded environmental or regulatory threshold values. This implies that actual applications of pesticide products, which generally have been approved for use, frequently have a significant likelihood of causing adverse effects on aquatic organisms.

Field effects of pesticides in general

Schäfer (2019) reviewed studies concerning the effects of pesticides (mainly insecticides and fungicides) on field communities of freshwater macroinvertebrates. Of the 13 reviewed field studies from different world regions conducted during the last 15 years, nine found a clear or likely relationship between pesticide toxicity or concentrations on the one hand and biotic responses (e.g., community composition or different community indices) on the other (Table 4.3.5-1). These field studies indicate widespread effects from agricultural

Table 4.3.5-1 **Studies, covering at least five agricultural water bodies, which involved pesticide residue sampling and macroinvertebrate community sampling. Effects from urban and industrial land use were excluded. Studies reanalysing the same data are not considered independently.** Adapted from Schäfer (2019).

Region	Number and types of sites	Type of biological response	Relationship between pesticide exposure and biological response
Western France and southern Finland	29 streams	Species at risk of pesticides (SPEAR) index	Clear (R2 = 0.64)
Central Germany	19 streams	SPEAR index	Clear (R2 = 0.59)
Denmark	14 streams	SPEAR index	Clear (multiple time points, R2 between 0.4 and 0.68)
Central Argentina	22 streams	SPEAR index	Clear (multiple time points, R2 between 0.35 and 0.42)
South-eastern Australia	24 streams	SPEAR index	Clear (R2 = 0.68)
Midwestern United States	98 streams	Multimetric index	Likely (R2 = 0.22)
Central Argentina	6 channels	Species richness and total abundance	Likely (r = 0.59 and 0.61)
Netherlands	14 ponds and ditches	Community composition	Clear (5.4 per cent of community variance explained)
Portugal	6 ditches	Community composition	Clear (23.7 per cent of community variance explained)
Northern Germany	9 streams	SPEAR index	Ambiguous (group-based comparison)
Eastern Argentina	8 streams	Community composition	Ambiguous (group-based comparison)
Southern Brazil and Paraguay	18 and 17 streams	27 indices including SPEAR	Ambiguous to none (most metrics exhibit no correlation including SPEAR)
Western Germany	23 streams	8 indices including SPEAR	Ambiguous (correlation with SPEAR and per cent EPT (ephemeroptera, plecoptera and trichoptera) between 0.26 and 0.4)

pesticide use despite the existence of pesticide regulation. Further field studies are required in order to understand the mechanisms underlying pesticide toxicity; provide the data to develop and critically evaluate effect models; and provide information about adaptation processes in aquatic communities and drivers of community composition in the face of global environmental change (Schäfer 2019).

Further effects of pesticides on the biological diversity of communities of aquatic organisms have been observed in the global review by Sánchez-Bayo and Wyckhuys (2019) and studies by Beketov *et al*. (2013) in Germany, France and Australia and Ito *et al*. (2020) in Japan (Chapter 4.3.6). Pesticides were one of several drivers of aquatic biodiversity losses, albeit an important one.

Macroinvertebrates play an important role in the decomposition process such as leaf litter breakdown. Subsequently, a loss in aquatic macroinvertebrate abundance and diversity may exert impacts on ecosystem functioning. Schäfer *et al*. (2012) compiled data from eight field studies in Europe, Siberia and Australia to derive thresholds for the effects of pesticides on macroinvertebrate communities and leaf breakdown. Dose-response models for the relationship of pesticide toxicity and the abundance of sensitive macroinvertebrate taxa showed significant differences with reference sites at 1/1,000 to 1/10,000 of the median

acute effect concentration (EC_{50}) for *Daphnia magna*. The invertebrate leaf breakdown rate was positively related to the abundance of pesticide-sensitive macroinvertebrate species in the communities (Schäfer *et al*. 2012).

In line with these findings, Peters, Bundschuh and Schäfer (2013) reported that more than one-third of the reviewed studies indicated reductions in ecosystem functions at pesticide concentrations that are assumed to be protective by regulatory standards. In addition to direct toxic effects on macroinvertebrates, negative effects on ecosystem functioning (e.g., decomposition and nutrient recycling) have been reported as a result of pesticide-induced toxicity to microorganisms at environmentally realistic concentrations (Staley, Harwood and Rohr 2015).

Field effects of insecticides

In line with the previous chapter, the effects of insecticides on both water-dwelling (Brander et al. 2016) and benthic (Li, H. *et al*. 2017) invertebrates have been reported on a global scale at environmental realistic concentrations. Huang, Cui and Duan (2020) also demonstrated that measured environmental concentrations of the organophosphate insecticide chlorpyrifos were often above allowable environmental limits. These studies recommended the development of accurate sediment quality criteria and more effective ecological risk assessment methods. In the EU, the effects assessment of pesticides with regard to sediment-dwelling organisms in edge-of-field surface water has received increasing attention in recent years (EFSA 2015). In addition, a case study with chlorpyrifos has shown the potential of using post-registration monitoring data in environmental pesticide risk assessments (Rico *et al*. 2020).

Neonicotinoid insecticides have received increasing attention in surface water pesticide risk assessment, as the standard test species ***Daphnia magna*** is highly tolerant to this insecticide group (Sánchez-Bayo and Tennekes 2020). Apart from laboratory and mesocosm studies, however, limited research has been directed towards the role neonicotinoids may have in structuring aquatic invertebrate communities in field settings (Schepker *et al.* 2020).

Two large-scale wetland field studies in Nebraska (United States) (Schepker *et al*. 2020) and Saskatchewan (Canada) (Cavallaro *et al.* 2019) revealed that neonicotinoid concentrations were significant factors in shaping invertebrate field communities. Schepker *et al*. (2020) further reported that the significant negative effects on aquatic invertebrate biomass observed across all wetlands studied occurred at neonicotinoid concentrations below benchmark concentrations proposed by government regulations.

Another issue that has been raised in regard to the risks of neonicotinoids to aquatic life is that their toxicity increases with exposure time, as much as with the dose, and has therefore been described as time-cumulative toxicity. Regulatory assessments for this class of compounds should not be based solely on exposure doses, but also need to consider the time factor (Sánchez-Bayo and Tennekes 2020).

Although OCPs were taken off the market decades ago in most parts of the world, they remain present in low concentrations due to their high persistence. Martyniuk, Mehinto and Denslow (2020) compiled the available data on molecular and apical endpoints and concluded that prolonged exposure to these low environmental OCP levels (in low to mid ng/L range) are likely to disrupt sex hormone production in male fish, including a reduction of testosterone. At the same time, there is an increase in vitellogenin, suggesting agonism of the oestrogen receptor. These changes lead to impaired sperm cell maturation and release and reduced fecundity (Martyniuk, Mehinto and Denslow 2020).

Field effects of fungicides

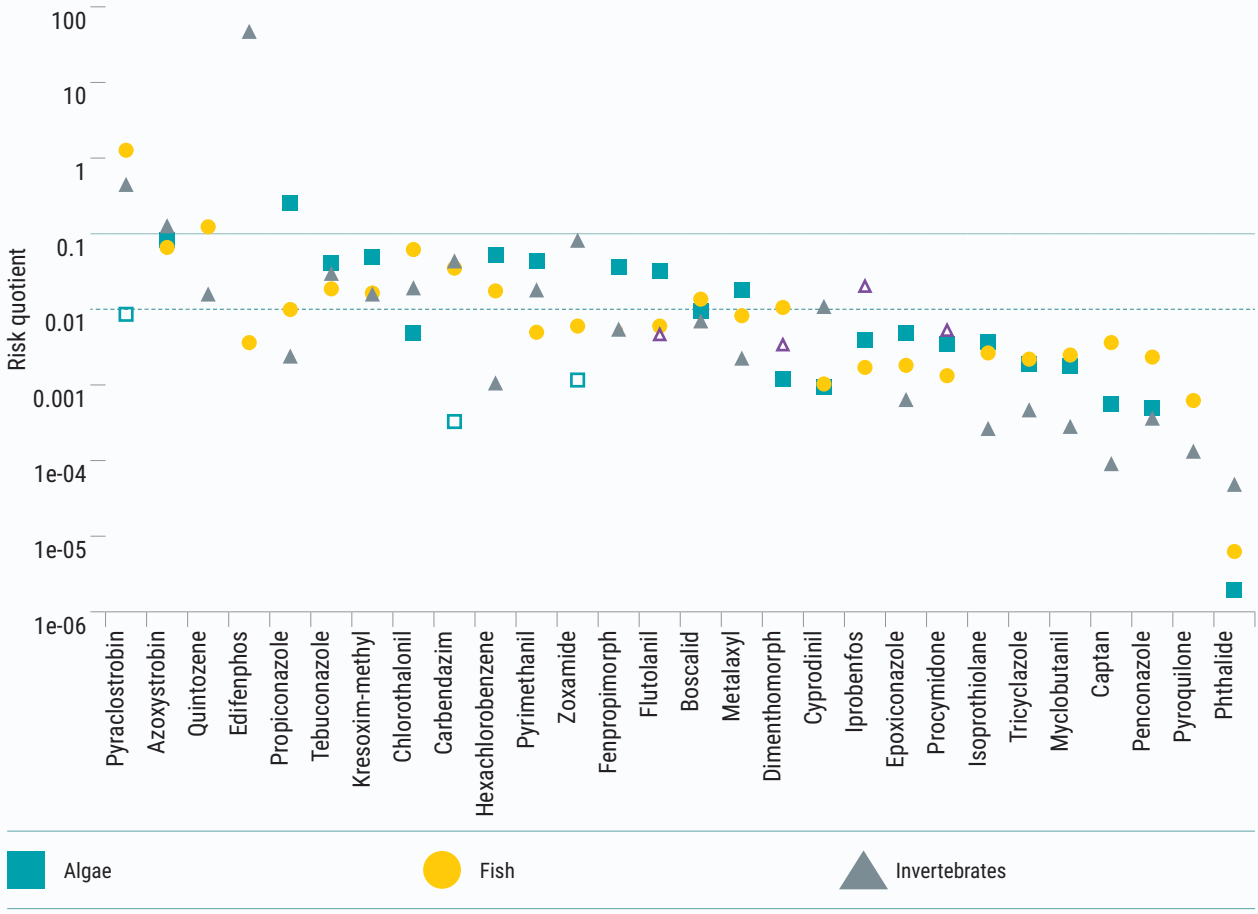
In comparison to insecticides, exposure to and the effects of fungicides and herbicides in aquatic ecosystems have received considerably less attention (Maltby, Brock and Van Den Brink 2009; Rico, Brock and Daam 2019; Zubrod *et al*. 2019; Rosic *et al*. 2020). Zubrod *et al*. (2019) compiled laboratory bioassay, model ecosystem and field

studies, but only reported field studies for five different fungicides. The study also showed that environmentally realistic concentrations of fungicides are likely to exert moderate to high risks to aquatic life (Figure 4.3.5-1). The authors indicated that the reasons for these identified high risks are presumably manifold, such as underestimation of field concentrations by exposure modelling, application of safety factors that are too low, and use of higher-tier risk assessment methods that can strongly increase regulatory acceptable concentrations (RACs). The non-inclusion of the most sensitive organisms (lowest effect concentrations are often for fungi and fungal-like organisms) in regulatory testing might have further contributed to this situation (Zubrod *et al*. 2019).

Rico, Brock and Daam (2019) compared lower-tier (bioassays) and higher-tier (model ecosystems) RACs for fungicides. They concluded that the current RACs for individual fungicides, with a few exceptions (e.g., tebuconazole), show a sufficient level of protection for structural and functional fungal endpoints, but that more data are needed to extend this comparison to other fungicides with different modes of action. Consistent with Zubrod *et al*. (2019), Rico, Brock and Daam (2019) also concluded that further research is needed on the impact of realistic exposure regimes of (mixtures of) fungicidal compounds on the decomposer food chain.

For insecticides (arthropods) and herbicides (primary producers), a most sensitive taxonomic group of test species can generally be distinguished. For fungicides, this appears not to be the case as it depends on the chemical group or mode of action of the fungicide of concern (e.g., see Maltby Brock and Van Den Brink 2009).

Figure 4.3.5-1 Risk quotients for algae (squares), fish (circles) and invertebrates (triangles) as ratios of maximum detected global field concentrations and acute standard toxicity data. Open symbols indicate that toxicity was provided as "greater than" values. Risk quotients > 0.01 (dashed line) and > 0.1 (solid line) indicate moderate and high risks, respectively. Zubrod *et al*. (2019).



In the study by Zubrod *et al*. (2019), however, risks were relatively similar across algae, fish, and invertebrates although high risks tended to occur more frequently in the case of invertebrates than in that of algae or fish (Figure 4.3.5-1). A comparison across continents showed most frequent risks for Europe (11 substances), followed by North America (six), Asia (four to five), Africa (three) and South America (two). This situation might not reflect only intensity of use, but also monitoring efforts (Zubrod *et al*. 2019).

Field effects of herbicides

As in the case of fungicides, the potential side-effects of environmentally realistic herbicide concentrations in agricultural edge-of-field surface water on primary producers and on ecosystem functioning have been poorly studied. Rosic *et al*. (2020) reviewed the potential risks related to realistic field concentrations of the herbicides bromoxynil, diquat and paraquat to Australian freshwater life. They concluded that at concentrations resulting from current agricultural practices, diquat exerted toxic effects on snails and bromoxynil on microalgae. The clearest and most consistent evidence of adverse effects was found for paraquat. At realistic field concentrations paraquat severely inhibited healthy bacterial growth (E. coli), distorted tropical freshwater plankton communities, and increased fish mortality (common carp) three times more than the weed (water hyacinth) it was employed to control (Rosic *et al*. 2020).

Model ecosystem studies in general, including those designed to evaluate the toxicity of herbicides, have traditionally been designed to evaluate a concentration series enabling the setting of a no observed effect concentration (NOEC), which is ultimately to be compared with a predicted environmental concentration in a prospective pesticide risk assessment (van den Brink and Daam 2014). However, those studies have little relevance to the purpose of this report. Model ecosystem studies evaluating realistic pesticide application scenarios (or concentrations) are conducted in prospective risk assessments, but their results are often confidential and/or confined to regulatory reports. An evaluation of such reports, in combination with reported pesticide field concentrations, would therefore be an interesting way forward to elucidate actual field risks of pesticides.

Some model ecosystem studies evaluating the effects of environmentally realistic herbicide concentrations on primary producers have been published in the open literature. Mohr *et al*. (2007, 2008), for example, demonstrated that adverse effects on algae and macrophytes are likely to occur at concentrations well below those detected after the registered use of the herbicide metazachlor. Model ecosystem studies designed to evaluate pesticide mixture exposure resulting from crop-based permissible applications of pesticides are also a promising way forward to elucidate (mixture) effects of a pesticide in actual potato (Arts *et al*. 2006), strawberry (Arts *et al*. 2017) or bulb (van Wijngaarden *et al*. 2004) cultures. Mixture applications of triazine herbicides, for example, are known to potentially result in synergistic effects in the field (Cedergreen 2014).

Pesticide effects on marine ecosystems

Mariculture activities and river inputs can lead to pollution of coastal seawater with micropollutants such as antibiotics and pesticides, with a seaward decreasing trend of pesticide concentrations mainly due to pesticide dilution, dispersion and degradation (Grant *et al*. 2018; Xie *et al*. 2019). On the other hand, these decreasing pesticide concentrations may not suffice to prevent side effects on (especially filter-feeding) marine organisms that may bioaccumulate pesticides even when present in very low concentrations in environmental matrices (Ojemaye *et al*. 2020 and references therein).

Pesticides used on land may therefore pose risks to marine organisms and especially to organisms in coastal areas (Xie *et al*. 2019; Ojemaye *et al*. 2020; Parsons *et al*. 2020), the more so since their sensitivity to pesticides has been demonstrated to be comparable to that of freshwater organisms (Maltby *et al*. 2005; Klok *et al*. 2012; EFSA 2013b; European Commission 2018).

No recent global reviews of the effects of pesticides on marine ecosystems were available,

however a few outlined below indicate that marine organisms may be exposed to pesticide concentrations that pose a risk to population health.

Ojemaye *et al*. (2020) detected five herbicides in seawater, sediment, seaweed and selected marine organisms, such as limpets (*Cymbula granatina*), mussels (*Mytilus galloprovincialis*), and sea urchins (*Parechinus angulosus*), which were present in the near shore environment of Camps Bay (Cape Town, South Africa). In addition to environmental risks, these herbicides were indicated to pose adverse health effects should an average sized human (70 kg) consume any of the marine species analysed in the study on a daily basis over a lifetime (Ojemaye *et al*. 2020). Deltamethrin, an anti-sea lice pesticide used in the salmonid aquaculture industry, was also found to pose a significant risk to European lobster in the Norwegian marine environment (Parsons *et al*. 2020).

Marine monitoring studies may be especially important in coastal areas with high biodiversity value such as the Great Barrier Reef. The annual report on inshore pesticide monitoring of the Great Barrier Reef Marine Park Authority indicated that the current water quality guideline values (levels to protect 99 per cent of marine species) were not exceeded at any site for any pesticide in 2016-2017 (Grant *et al*. 2018). The herbicides diuron, atrazine and hexazinone were the pesticides most frequently detected and in the highest concentrations, which was related to pesticide usage by the sugarcane industry in adjacent catchments. Despite overall low pesticide concentrations, Grant *et al*. (2018) argue that the cumulative effects of long-term exposure to the mixture of chemicals on the resilience of the reef ecosystem need to be evaluated, especially considering the multiple local, regional and global stressors already acting on this ecosystem such as cyclonic activity and the effects of global climate change (increasing sea temperatures, ocean acidification).

Recently Spilsbury *et al*. (2020) have indicated that an ecotoxicological risk exists resulting from pesticide mixtures in 38.5 per cent of samples taken from Australian rivers discharging to the Great Barrier Reef. Analysis of land use patterns in the catchment areas showed an association between the sugarcane industry and elevated risk levels, which were driven by the presence of diuron (Spilsbury *et al*. 2020).

However, the chronic effects on corals and seagrass of low-level pesticide exposure, especially combined with other local and global pressures, remain poorly understood on the Great Barrier Reef (Grant *et al*. 2018). Furthermore, Brodie and Landos (2019) have indicated that potential effects on important local species of the reef's marine environment have only been studied through laboratory single species toxicity testing. Thus there is a need to include biological monitoring of field communities (e.g., through the use of biomarkers) along with exposure assessments in future coastal and marine monitoring campaigns (Varea, Piovano and Ferreira 2020).

4.3.6 Pesticides and biodiversity

Biological diversity – or biodiversity – has been defined by the Convention on Biological Diversity (CBD) *as the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems* (Convention on Biological Diversity [CBD] 1992, Chapter 3.2.4).

At its simplest, biodiversity is the number of species present in a given geographical unit: that is, species richness. More complex indices of species richness may include relative abundances, biomasses or productivities of coexisting species. However, biodiversity can also be evaluated at a smaller scale than a species, e.g., genetic diversity within the species; or at a larger scale, such as the variety of community types or ecosystems present in a region (Begon, Harper and Townsend 1996). The CBD recognizes three main components of biodiversity which are important for its conservation and sustainable use: genomes and genes; species and communities; and ecosystems and habitats (CBD 2019).

Effects of pesticides on biodiversity

Many adverse effects of pesticides on non-target organisms and communities discussed in Chapter 4.3 above may have an impact on biodiversity. In this section studies will be discussed in which the use of pesticides has been explicitly linked to adverse effects on biodiversity.

No systematic global review of the effects of pesticides on biodiversity is currently available, however there are several global reviews and large-scale studies and partial reviews of effects on biodiversity. There are summarized in Annex 4.3-6. Included are reviews comparing organic with conventional agriculture, as the absence or strong limitation of pesticide use is an important characteristic of organic production.

The use of pesticides in agriculture, public health and elsewhere has consistently been mentioned in international reviews and reports as one of the drivers of biodiversity loss (Table 4.3.6-1). However, these reviews are generally not very specific as to the exact role of pesticides in the decline of biodiversity. With the exception of the IPBES assessment report on pollinators (Potts, Imperatriz-Fonseca and Ngo 2016), these status reports do not systematically review the effects of pesticides on biodiversity.

Relatively few studies have assessed the effects of pesticide use on parameters that describe or quantify biodiversity. More common are studies that assess the effects of agricultural intensification on biodiversity, although these studies may include consideration of factors such as fertilizer use, habitat destruction, increase in large-scale monocultures, reduction in crop rotation, besides pesticide use. The number of studies that attempt to disaggregate the importance of pesticides among other drivers of biodiversity loss is limited. In other cases, the effects of pesticides on abundance of individual taxa are assessed and subsequently cited as impact on biodiversity.

Arthropod biodiversity

The majority of available assessments concern the effects of pesticides on biodiversity of insects and other arthropods. This is not surprising given the fact that insecticides and certain fungicides, in particular, are well known to adversely affect both terrestrial and aquatic arthropods (Chapters 4.3.3, 4.3.4 and 4.3.5).

Sánchez-Bayo and Wyckhuys (2019) reviewed a large number of studies of insect declines across the globe, in both terrestrial and aquatic environments, and attempted to assess the underlying drivers. They found that 41 per cent of insect species are in decline and one-third of species are threatened with extinction. Among aquatic insects, habitat and dietary generalists, as well as pollutant-tolerant species, are replacing the large insect biodiversity losses experienced in water within agricultural and urban settings.

The main drivers of these species declines have been habitat loss and conversion to intensive agriculture and urbanization along with pollution, mainly by synthetic pesticides and fertilizers. When individual factors were assessed, the use of pesticides was associated with insect declines in 13 per cent of cases, the second most important factor identified, after intensive agriculture (in which pesticides may also be an integral factor). Although intensive agriculture and pesticides are the main factors associated with insect declines (Figure 4.3.6-1B), habitat loss appears to be the largest driver of decline, followed by pollution (including by pesticides) (Figure 4.3.6-1A).

The impact of pesticides on insect biodiversity is further substantiated by global comparisons between organic and conventional agriculture, which showed that organic production was associated with significantly higher diversity of wild bee populations (Kennedy *et al*. 2013) or insects in general (Tuck *et al*. 2013). Other large-scale studies in the United States and Europe have confirmed that the use of pesticides, particularly insecticides, is associated with declines of wild bee diversity (Brittain *et al*. 2010; Park *et al*. 2015; Mallinger, Watts and Gratton 2015), as well as diversity of bumblebees and butterflies (Brittain *et al*. 2010; Forister *et al*. 2016). Sattler *et al*. (2020) have found that pesticide use has strong negative effects on

Table 4.3.6-1 Global biodiversity reviews and status reports citing pesticide use as one of the drivers of biodiversity loss

Publication year	International organizations	Title	Mentions of pesticide use affecting biodiversity
2020	Food and Agriculture Organization of the United Nations (FAO), Intergovernmental Technical Panel on Soils (ITPS,) Global Soil Biodiversity Initiative, Secretariat of the Convention of Biological Diversity and European Commission	State of Knowledge of Soil Biodiversity – Status, Challenges and Potentialities (FAO, ITPS, Secretariat of the Convention on Biological Diversity and European Commission 2020)	Agricultural intensification, and associated greater use of external inputs such as pesticides, has resulted in decreased soil biodiversity.
2020	Organisation for Economic Co-operation and Development (OECD)	Managing the Biodiversity Impacts of Fertiliser and Pesticide Use: Overview and Insights from Trends and Policies across Selected OECD Countries (Sud 2020)	Various studies have shown that excessive use of pesticides has led to biodiversity loss and ecosystem degradation. (The report cites a number of studies.)
2019	FAO	The State of the World's Biodiversity for Food and Agriculture (FAO 2019)	There is abundant evidence that intensification of crop, livestock and aquaculture systems through excessive use of synthetic inputs adversely affects biodiversity for food and agriculture and particularly associated biodiversity. Examples are provided of effects of pesticides on pollinators, soil organisms, biological control agents and aquatic organisms.
2019	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES)	The Global Assessment Report on Biodiversity and Ecosystem Services (IPBES 2019)	Pesticides are mentioned as among the drivers of changes in biodiversity (Sections 2.1 and 2.3).
2016	IPBES	Assessment Report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on Pollinators, Pollination and Food Production (Potts, Imperatriz-Fonseca and Ngo 2016)	Changes in land use or climate, intensive agricultural management and pesticide use, invasive alien species and pathogens affect pollinator health, abundance, diversity and pollination directly.
2015	FAO and ITPS	Status of the World's Soil Resources (FAO and ITPS 2015)	The large-scale use of pesticides may have direct or indirect effects on soil biodiversity. However, studies on the effect that pesticides have on soil biodiversity have shown contradictory results.
2014-2020	Convention on Biological Diversity	Global Biodiversity Outlook (GBO), most recently the GBO 5 (2020) (Secretariat of the Convention on Biological Diversity 2020)	The use of fertilizers and pesticides has stabilized globally, though at high levels. Despite such progress, biodiversity continues to decline in landscapes used to produce food and timber; and food and agricultural production remains among the main drivers of global biodiversity loss… Pollution, including from excess nutrients, pesticides, plastics and other waste, continues to be a major driver of biodiversity loss.
2005	Millennium Ecosystem Assessment	Ecosystems and Human Well-being: Biodiversity Synthesis (Millennium Ecosystem Assessment 2005)	There have been worldwide declines in pollinator diversity… The causes of these declines are multiple, but habitat destruction and the use of pesticide are especially important.

Figure 4.3.6-1 **The main drivers (A) and factors (B) associated with insect declines. Pesticides are identified as the second most important factor associated with declines of insect species' abundance and biodiversity. Shown is (A) the number and (B) the percentage of studies in which specific drivers and factors of decline were determined.** Based on Sánchez-Bayo and Wyckhuys (2019).

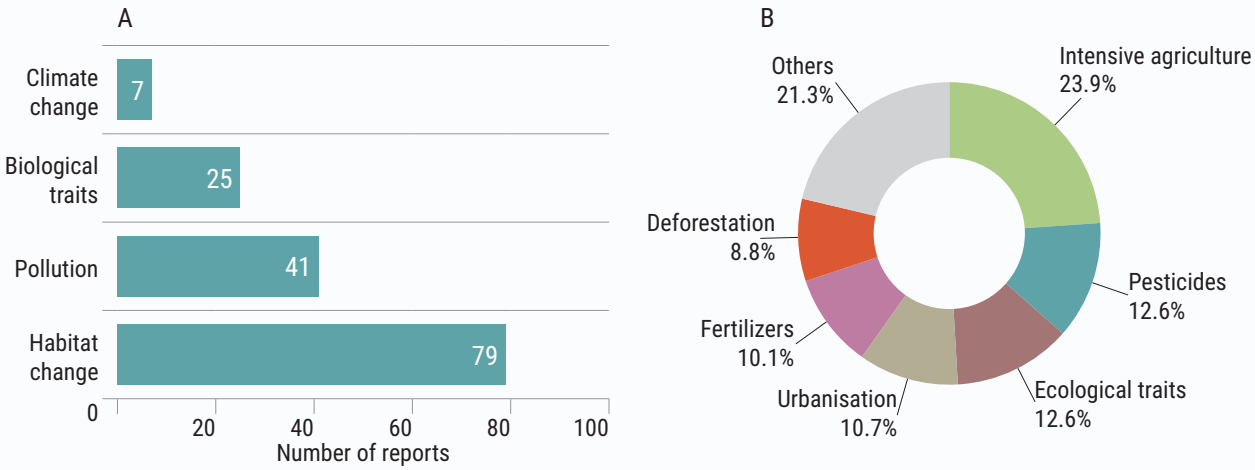
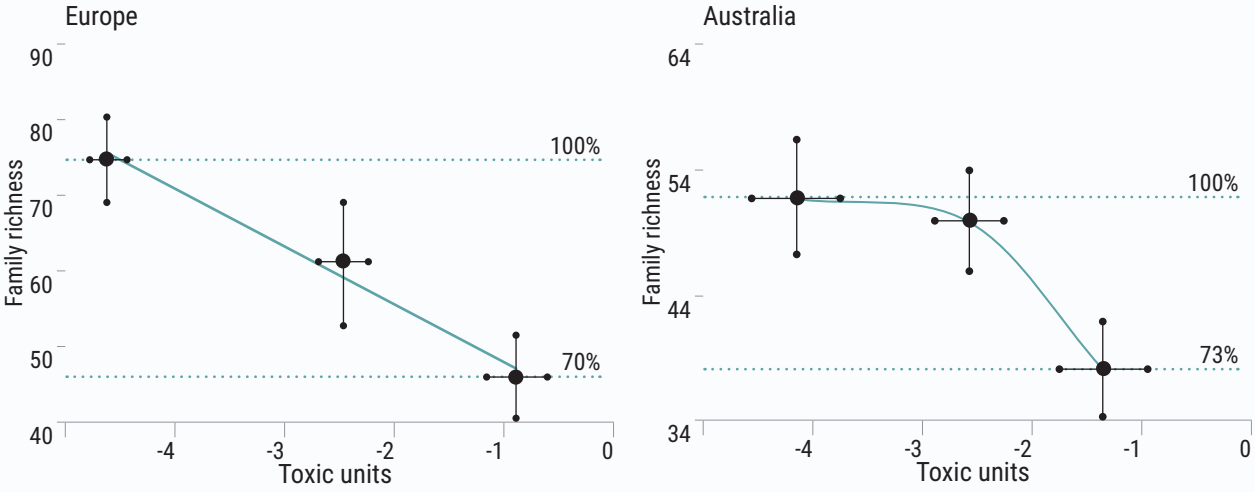


Figure 4.3.6-2 **Concentration-response dependence between the mean pesticide concentration and mean overall aquatic macroinvertebrate family richness measured in sampling sites in Europe and Australia. The dashed horizontal lines indicate maximum and minimum mean richness and are marked with the percentages compared to maximum richness. The error bars indicate 95 per cent confidence intervals.** Beketov *et al.* (2013).



taxonomic and functional diversity of terrestrial rice field arthropods.

For the aquatic environment, Beketov *et al.* (2013) evaluated the field effects of pesticides on the regional taxa richness of stream macroinvertebrates in Europe (Germany and France) and Australia (southern Victoria). By capturing episodic run-off events, it could be determined that pesticides caused statistically significant effects on both species and family richness in both regions, with losses in family richness up to 42 per cent of recorded taxonomic pools (Figure 4.3.6-2). Furthermore, the effects in Europe were detected at concentrations that current legislation considers environmentally protective (Beketov *et al.* 2013).

Ito *et al.* (2020) evaluated the impacts of pesticides and other environmental stressors (including

eutrophication, decreased macrophyte coverage, physical habitat destruction and invasive alien species) on the taxonomic richness of freshwater animals in 21 irrigation ponds in Japan. Similarly to the conclusions of Sánchez-Bayo and Wyckhuys (2019), Ito *et al*. (2020) found that the taxonomic richness of freshwater animals in Japanese irrigation ponds has been affected by multiple stressors including pesticides.

In other cases, however, pesticide effects are less clear-cut, e.g., for soil fauna biodiversity (de Graaff *et al*. 2019). Indeed, studies evaluating the effects pesticides may have on soil biodiversity have shown contradictory results. Effects appear to be dependent on a variety of factors including chemical composition, the pesticide application rate, the buffering capacity of the soil, the soil organisms considered and the time scale of the study. There have been no comprehensive studies to quantify the effects of pesticides on soil organisms at multiple trophic levels across regions (FAO and ITPS 2015).

More generally, various authors have indicated that whether effects of pesticides on biodiversity are observed in the field may depend on many parameters and complex interactions. For instance, Mallinger, Werts and Gratton (2015) and Park *et al*. (2015) observed that pesticide use in apple orchards affected the species richness of certain groups of wild bees but not of others.

Vertebrate biodiversity

Few large-scale studies and reviews of pesticide effects on vertebrate biodiversity are available, and most have focused on birds. In their global meta-analysis, Tuck *et al*. (2013) found 20 per cent greater bird species richness in organic agriculture compared to conventional production systems.

A large study was conducted on 270 cereal farms across eight European countries, in which the effects of 13 components of agricultural intensification on biodiversity were investigated across three trophic levels (diversity of wild plants, carabid beetles and breeding bird species) (Geiger *et al*. 2010; Emmerson *et al*. 2016). The use of pesticides, especially insecticides and fungicides, was reported to have consistent negative effects on species diversity at all three trophic levels. Bird species diversity, in particular, declined with increasing frequency of fungicide applications. It was noted that while finding negative effects of pesticides on biodiversity may be unsurprising, these effects were found consistently at a pan-European scale and despite decades of policy to reduce pesticide risks.

In France, Chiron *et al*. (2014) observed that, as herbicide dose increased, there was a shift in functional groups from specialist species to more generalist ones but no effect on species richness. Losses of endangered mammal, bird, amphibian and reptile species of in Canada were found to be strongly associated with the proportion of the region treated with agricultural pesticides (Gibbs, MacKey and Currie 2009). Species losses were more strongly related to herbicide use than to the use of other pesticides.

4.3.7 Knowledge gaps on environmental effects of pesticides

Availability of systematic reviews

Many scientific studies have been and are being conducted on the occurrence of pesticides in the environment, as well as on their potential impact on non-target organisms and ecosystems. Nevertheless, very few systematic reviews have been conducted of the available data, or of what their implications may be for pesticide risks and their mitigation options.

To make full use of existing research, the scientific literature should be monitored and analysed on a regular basis through systematic reviews and/or meta-analyses, particularly for topics of specific interest with regard to regulatory decision-making (Group of Chief Scientific Advisors 2018). Such topics include, but are not limited to, pesticide concentrations in the environment and their potential risks; the accuracy of fate models (and their scenarios) in predicting pesticide field concentrations; the field effects of pesticides (mixtures) on populations and communities of non-target organisms; the impact of pesticides on the sustainability of agricultural production and disease vector control; and mechanistic studies

evaluating the underlying pathways from pesticide exposure until adverse effects in organisms.

It is important that the results of such reviews and meta-analyses be interpreted with consideration given to the risks of the current authorized use of the pesticides involved, and that results be made available to risk managers and decision-makers in other parts of the world. A specific platform may need to be established for the elaboration and publication of such reviews, similar, for example, to the Cochrane Reviews of research in health care and health policy (Cochrane Library 2020). As these systematic reviews would go beyond the environmental effects of individual pesticides (i.e., include pesticide mixtures), and thus current regulatory approaches to pesticide registration, dedicated efforts by research or regulatory bodies would likely be required.

Environmental risk assessment

Substantial advances have been made in strengthening the environmental risk assessment procedures used as a basis for the registration of pesticides in many parts of the world. Toxicity data for a broader range of non-target organisms have been generated; adverse outcome pathways are increasingly being elucidated; more realistic exposure and ecological effect models have been developed; the protection of ecosystem services and biodiversity are better taken into account; and the validation of risk assessment approaches has improved.

Nevertheless, despite these significant improvements in prospective regulatory risk assessment, unexpected environmental impacts are frequently identified, typically many years after a pesticide has been placed on the market. Such impacts may be due to incorrect or unintended use of the pesticide; inadequate risk assessment models or approaches; or unforeseen adverse effects (Boyd 2018; Group of Chief Scientific Advisors 2018; Brühl and Zaller 2019; Schäfer *et al*. 2019; Topping, Aldrich and Berny 2020).

Many recommendations have been made on how best to strengthen environmental risk assessment, either through the generation of additional pesticide toxicity or fate data, the development of new or better risk assessment models, or even the introduction of new risk assessment paradigms.

Some of the principal directions for improving pesticide risk assessment include the following:

- **Pesticide mixtures and other stressors:** Many organisms will be exposed to multiple pesticides, as well as to other stressors, which may result in larger effects on their long-term survival than separate exposure to each substance individually. There is a need to further develop methodologies that quantify the risks of exposure to multiple pesticides simultaneously or sequentially, as well as to other (chemical and non-chemical) stressors (Babut *et al*. 2013; Brühl and Zaller 2019; Topping, Aldrich and Berny 2020; Martin *et al*. 2021).

- **Neglected organisms and ecosystems:** Currently pesticide risk assessment focuses on a limited set of organism groups and ecosystems and is heavily biased towards temperate climatic conditions. Dedicated risk assessment procedures for (sub-)tropical and hot (semi-) arid ecosystems, microbial and fungal communities, amphibians and reptiles, and groundwater organisms are urgently needed (Sánchez-Bayo and Hyne 2011; EFSA 2018a; Ittner, Junghans and Werner 2018; Daam *et al*. 2019; Castaño-Sánchez, Hose and Reboleira 2020).

- **Indirect and delayed effects:** Pesticide ecotoxicological testing and risk assessment primarily aims at identifying direct lethal and sublethal effects on non-target organisms. However, indirect effects of pesticides, e.g., across trophic levels or by reducing plant cover, can also affect non-target populations and communities and require attention in risk assessment (Schäfer *et al*. 2019). In addition, pesticide effects may emerge, persist or even increase after the pesticide has dissipated from the ecosystem due to delayed (latent) effects or fitness costs associated with the adaptation

to pesticide exposure (Sánchez-Bayo and Tennekes 2020; Siddique *et al*. 2020). Such delayed effects are not yet well covered in risk assessment.

- **Agricultural sustainability:** While pesticides are known to affect the sustainability of agricultural production systems, prospective risk assessments on this topic tend to be limited to assessing the toxic effects of pesticides on a few non-target arthropods. Effects on ecosystem services such as natural pest control, pollination and nutrient cycling tend to become apparent only in post-registration studies and monitoring. There is a need to more systematically integrate the potential effects of pesticides on agro-ecosystem functioning and long-term agricultural sustainability up front in the prospective decision-making process.

- **Assessment at landscape scale:** At present, pesticide risks are evaluated based on a single product-single crop assessment. However, pesticides are applied at the landscape scale, where effects on non-target organisms are ultimately influenced by sequential pesticide applications in multiple fields and crops, migration and recovery of non-target populations across the landscape, and interactions among species, among others. Risk assessment approaches need to be developed which take into account "landscape dosing" with pesticides and the spatial dynamics of non-target organisms (Boyd 2018; Schäfer *et al*. 2019; Topping, Aldrich and Berny 2020).

Monitoring

As discussed in the previous chapters, systematic monitoring of pesticide concentrations in environmental compartments such as air, surface waters, groundwater and soil is only conducted in some parts of the world; large data gaps exist in many others. Systematic monitoring of adverse effects on non-target organisms or communities is virtually absent in low and middle income countries and is rare even in high income economies.

The reasons for lack of environmental monitoring include its relatively high costs; the absence or inadequacy of pesticide residue analysis laboratories in many countries; the absence of relevant biomarkers of biological exposure; the complexity of field effects monitoring; and the difficulty of disaggregating the effects of individual pesticides from other (chemical and non-chemical) stressors on non-target organisms, communities and ecosystem functioning. Moreover, post-registration monitoring of environmental pesticide concentrations and effects is not legally required in many countries.

Due to limitations in the current prospective risk assessment procedures discussed above, and the lack of systematic post-registration monitoring, significant gaps in knowledge about adverse effects of current pesticide use on the environment persist. Recently, explicit calls for broader and more intensive post-registration monitoring or vigilance of pesticide use, concentrations and effects have therefore been made (Milner and Boyd 2017; Vijver *et al*. 2017; Group of Chief Scientific Advisors 2018; Rico *et al*. 2020; Topping, Aldrich and Berny 2020).

Post-registration environmental vigilance should ideally be based upon a combination of monitoring, modelling and experimental activities (Group of Chief Scientific Advisors 2018) and exploit existing large ecological and chemical datasets as well as currently scattered monitoring efforts (Vijver *et al*. 2017). It has been suggested that responsibility for such monitoring could be placed on pesticide manufacturers and users by applying previously registered designs for how data should be collected. New methods of precision farming also provide opportunities to facilitate data flows (Milner and Boyd 2017).

Data and knowledge obtained in this way could be used to assess or reassess risks as part of the national or regional pesticide registration system. In this way a "reality check" feedback loop would be established for the impact of pesticides on the environment which in turn can inform appropriate mitigation measures (Group of Chief Scientific Advisors 2018).

4.4 Adverse human health effects of pesticides

4.4.1 Overview

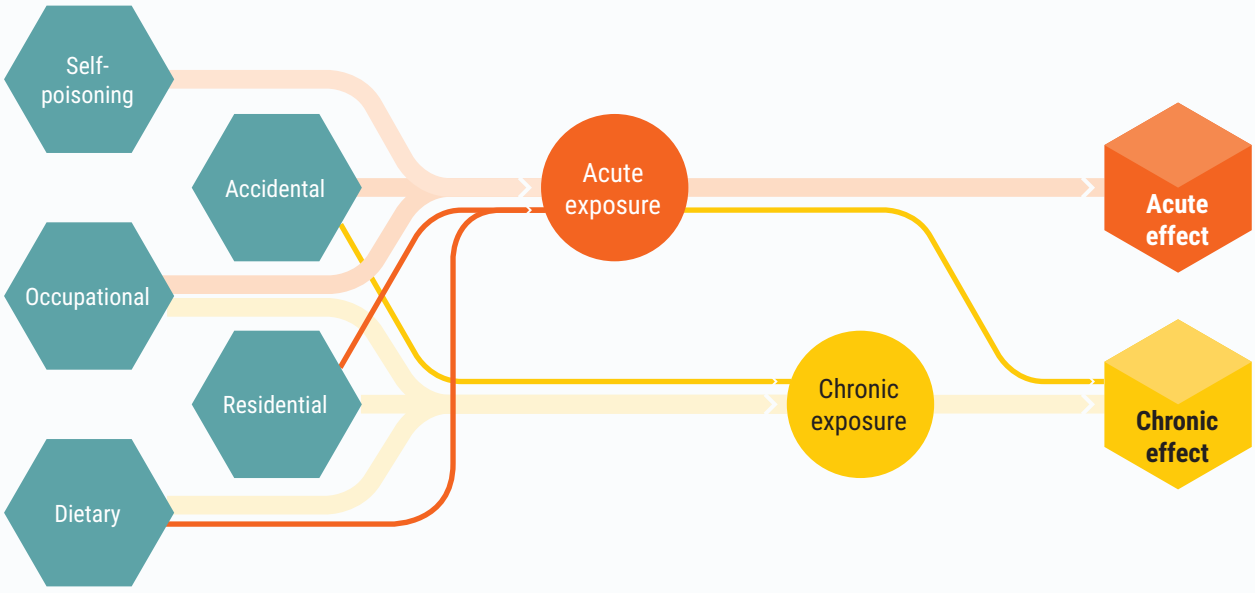
Almost all humans will be exposed in some way to pesticides, either when working with these products (e.g., farmers, gardeners, agricultural workers, professional pest control operators, workers in pesticide manufacturing, or those employed in pesticide sales, storage and disposal), through their diet (food, drinking water), via the environment (e.g., pesticides in air, dust and soil, or on treated surfaces in dwellings), or through products containing pesticides (e.g., impregnated mosquito nets or clothing, house paints).

Exposure to pesticides can be acute or chronic (Figure 4.4-1). Examples of acute exposure are accidents, occupational exposure and self-poisoning, which tend to involve relatively high levels of exposure to pesticides. Chronic exposure occurs, for example, through diet, working for longer periods with pesticides, and exposure to pesticides in the local environment.

Human health effects can also be acute or chronic. Acute pesticide poisoning occurs rapidly (within one to two days, sometimes within minutes or hours), after often high levels of acute exposure. Chronic health effects can develop over a longer period, sometimes many years. Chronic health effects may result from acute as well chronic exposure (Figure 4.4-1).

Most often, cases of acute unintentional pesticide poisoning tend to be occupational (e.g., poisoning of pesticide applicators or agricultural workers) or accidental (often in household settings). It has been estimated that in the 1980s, in developing countries alone, there were about 1 million serious unintentional poisoning cases per year, resulting in 20,000 deaths and an additional 25 million minor poisonings (WHO 1990; Jeyaratnam 1990). The only subsequent global estimate, based on a recent review of the scientific literature, suggests that about 385 million cases of unintentional acute pesticide poisoning occur every year, all severities combined, and that there are approximately 11,000 fatalities (Boedeker *et al.* 2020). This large increase in estimated acute pesticide poisoning may be partly due to growing pesticide use in many regions of the world. The Boedeker *et al.*

Figure 4.4-1 Human health effects of pesticides can show up in the short term (acute) or after a long period (chronic) and may be the result of activities leading to both acute and chronic exposure. Shown are the most common routes of poisoning (less likely options are indicated by thinner arrows).



(2020) review applied a methodology which is likely to include a much larger number of minor pesticide poisoning cases than the estimates for the 1980s. [Chapter 4.4.2]

A limited number of pesticides appear to be responsible for most cases of acute pesticide poisoning. In cases representing all severities, these pesticides include widely available groups such as pyrethroids, disinfectants, anti-coagulant rodenticides and glyphosate based-herbicides. Moderate and severe intoxications are caused mostly by organophosphates, aluminium phosphide fumigants and bipyridylium herbicides (UNEP 2020). [Chapter 4.4.2]

Chronic exposure to pesticides has been associated with a wide range of adverse health effects (American Academy of Pediatrics 2012; Bergman et al. 2013; Mostafalou and Abdollahi 2013; Blair *et al*. 2015; Hertz-Picciotto *et al*. 2018). A large number of epidemiological studies published during the last decade have examined such associations. However, based on epidemiological studies alone it is difficult to demonstrate that a specific pesticide is causing an adverse health effect. This is due to the large number of pesticide active ingredients, the variety of study designs used, the long period of time that often occurs between exposure to a pesticide and the development of a disease, the frequently inadequate characterization and quantification of exposure to specific (groups of) pesticides, and the variable presence of confounding and bias (Blair *et al*. 2015; European Food Safety Panel on Plant Protection Products and their Residues [EFSA PPR Panel] 2017). [Chapter 4.4.3]

Despite these limitations, however, there is increasing evidence of significant positive associations between (mainly) occupational or residential exposure to specific (groups of) pesticides, or pesticides in general, and various adverse health outcomes, including both adult and childhood cancers as well as neurological, immunological and reproductive endpoints (Ntzani *et al*. 2013; Ntzani *et al*. 2020; Ohlander *et al*. 2020). [Chapter 4.4.3]

Most intentional pesticide poisoning consists of self-poisoning in suicide attempts. Suicides with pesticides have been evaluated on a regular basis since the 1980s (Karunarathne *et al*. 2019). In the period 2006-2015 it was estimated that 1-2 million self-poisoning incidents related to pesticides occurred globally, resulting in about 168,000 deaths (Mew *et al*. 2017). Suicides by intentional pesticide ingestion occur primarily in rural areas of low and middle income countries in Africa, Asia and Central America (WHO 2014).

Pesticide suicides peaked in the 1990s and then started to decline (Karunarathne *et al*. 2019). Their number has fallen faster than that of overall suicides. The main drivers of this change are likely to be economic growth in some parts of the world and associated population shifts from rural to urban areas; further mechanization of agriculture; and stricter regulation of the most toxic pesticides (Mew *et al*. 2017; Gunnell *et al*. 2017; Karunarathne *et al*. 2019). [Chapter 4.4.4]

Many factors influence occupational pesticide exposure, related to the demography of the populations involved, pesticide application practices, the organization of the work environment, and workplace behaviours. A partial review of such factors shows a large variety of outcomes, indicating that whether a specific factor reduces or increases pesticide exposure and subsequent effects greatly depends on local conditions of use. [Chapter 4.4.5]

Demographic factors such as age, education and experience using pesticides have not been found to consistently increase or decrease pesticide health effects. Higher pesticide application rates and frequencies, or use of more toxic products, increase health effects, as do workplace behaviours that are in conflict with label instructions, good application practices or good occupational hygiene. Training in judicious pesticide use generally reduces pesticide exposure. If such training is conducted in isolation, however, its impact with regard to reducing health effects has been questioned (Table 4.4-1).

Long-sleeved shirts and trousers are worn by a majority of pesticide users across the world as basic protection against exposure. More specific personal protective equipment (PPE), such as gloves, masks, eye protection and chemical

Table 4.4-1 Principal lines of evidence investigated for this report in order to identify human health effects of pesticide use.

Type of evidence	Pesticide poisoning			Pesticide residues	Pesticide chronic effects	Risks factors
	Poison centre data	Published studies of poisoning	Suicide statistics	Monitoring of food and drinking water	Epidemiological studies	Various types of studies
Type of study	Mainly acute poisoning	Mainly acute poisoning	suicides	MRL exceedence, dietary risks	chronic effects	factors influencing pesticide exposure and effects
Type of outcomes	accidental, occupational, residential, self-poisoning	accidental, occupational, residential, self-poisoning	self-poisoning	dietary	occupational, dietary, residential	occupational residential

resistant footwear, is used by less than half of pesticide users globally. PPE is worn more regularly in high income than in low income countries. PPE may not be available or affordable for farmers in low income countries (Sapbamrer and Thammachai 2020). Moreover, many types of PPE are not suited to the hot and humid conditions in many of these countries (Garrigou *et al*. 2020; Sapbamrer and Thammachai 2020).

While not wearing appropriate PPE generally increases the incidence of poisoning, using this equipment does not always reduce it (Kim *et al*. 2013; Garrigou *et al*. 2020). Pesticide handling practices have a considerable influence on exposure. Furthermore, PPE does not provide complete protection against exposure. While proper use of PPE should always be promoted, in large parts of the world inadequate or partial use of PPE (together with the incomplete protection it provides) means PPE should be considered a last line of defence after other measures have been taken (Alli 2008; United States National Institute for Safety and Health 2015; European Agency for Safety and Health and Work 2018).

Residents living close to agricultural fields may be exposed to pesticides through different pathways. The most important are spray drift, volatilization of pesticides beyond the treated area, take-home exposure (e.g., when agricultural workers or farmers have pesticide residues on their clothing or shoes), pesticide use in/around a residence, and dietary ingestion (Deziel *et al*. 2015).

Residents living closer to pesticide-treated agricultural lands have been found to have higher levels of pesticide residues/metabolites in their households and/or biological samples, higher levels of oxidative stress markers, greater DNA damage, and decreased activity of cholinesterase compared with people who live farther away. In addition, the amount of pesticides applied, the acreage treated, and the time of year compared with the spray season have been positively correlated with human exposure (Dereumeaux *et al*. 2020). Associations between proximity to agricultural fields and various adverse health outcomes have been observed, although studies have not always been conclusive (Dereumeaux *et al*. 2020; Health Council of the Netherlands 2020).

Residential exposure to pesticides is likely to be widespread, especially in areas where there is intensive agriculture. Risk mitigation measures (e.g., drift-reducing technologies, buffer zones and improved work hygiene) have been recommended or required in many countries.

Pesticide residues may remain in/on food or feed following pesticide applications to the crop or post-harvest treatments of the commodity,

or because of contamination from environmental sources. Many countries set maximum residue limits (MRLs), which are legally permitted levels of pesticide residues on food and feed commodities. Pesticide residues in food are monitored on a regular basis mainly in high and upper-middle income countries; in lower income economies, generally only irregular ad hoc residue assessments are conducted if at all (Table 4.4-11). [Chapter 4.4.6]

Pesticide residue monitoring programmes in high and upper-middle income countries show that exceedances of MRLs typically range from <1 to 10 per cent of samples taken in a given year (Table 4.4-12). No reviews were available for pesticide residues in commodities in lower income economies. However, there are indications that residues in food in these countries are higher, based on ad hoc local studies as well as monitoring of food imported into high income countries (FAO/WHO Joint Meeting on Pesticide Residues, personal communication).

Exceedance of MRLs indicates that good agricultural practices have not been respected, but does not mean that dietary risks are therefore unacceptable (Box 4.4-3). Most dietary risk assessments conducted in high and upper-middle income countries indicate that consumer health risks from pesticide residues in food are low. Scarcely any dietary risk assessments, on the other hand, are available from low and lower-middle income countries (Table 4.4-13).

Antimicrobial resistance (AMR) refers to microorganisms – bacteria, fungi, viruses and protozoans – that have acquired resistance to antimicrobial substances. An increasing number of human pathogens have become resistant to one or more antibiotics, severely reducing the options for treating the diseases they cause (WHO 2015). Pesticides may select or co-select for AMR (Wellcome 2018; FAO and WHO 2019; Taylor and Reeder 2020). [Chapter 4.4.7]

Antibiotics are used in crop protection, especially to control bacterial diseases (FAO 2015). Certain of these antibiotics (e.g., streptomycin, gentamycin, oxolinic acid) are considered very important in human medicine (Berger *et al*. 2017; Wellcome 2018; WHO 2019; FAO and WHO 2019). However, they are used for crop protection much less than for veterinary purposes (FAO 2015; Taylor and Reeder 2020).

Furthermore, a rapid emergence is currently being seen of multi-drug resistant fungi for which the main treatment is antifungal medicines from the azole group (Fisher *et al*. 2018). The same group of chemicals is also among the fungicides most used in crop protection. There is increasing evidence that use of azole fungicides in agriculture is at least partly responsible for resistance development in human fungal diseases (Berger *et al*. 2017; Fisher *et al*. 2018; Wellcome 2018).

Copper fungicides, another widely used type of agricultural fungicides, although not used in human medicine, have been shown to co-select for AMR (Wellcome 2018; FAO and WHO 2019). The use of chemical disinfectants, of which a surge has been seen during the COVID-19 pandemic, may also select for AMR in antimicrobial human drugs, although evidence is still limited (FAO and WHO 2019).

Overall, the use of pesticides which have been shown (or have the potential) to select for AMR in human diseases, and possible subsequent failure of medical treatments, is receiving increasing attention.

Much knowledge has been amassed during the last few decades about the human health effects of pesticides. However, there is scope for further improvement in the scientific data that underpin human health risk assessments of pesticides and the methods by which such data are analysed (Science Advice for Policy by European Academies [SAPEA] 2018). [Chapter 4.4.8]

Knowledge gaps have been identified in the toxicological studies that generate data for risk assessments, the epidemiology of health effects of pesticides, in estimates of human exposure to pesticides and in post-marketing surveillance of approved pesticides (Milner and Boyd 2017; European Food Safety Authority Scientific Committee 2019; Liu *et al*. 2019).

Different lines of evidence regarding the effects of pesticide use on human health are evaluated in this report (Table 4.4-1). The focus of this chapter is on observed health effects of pesticides under real conditions of use. The chapter does not cover all possible human health risks of pesticide use, nor is it a detailed discussion of health risk assessment methodology. The dietary risk assessments discussed below are only included when they are based on pesticide residue levels following actual use.

4.4.2 Acute unintentional pesticide poisoning

Acute unintentional pesticide poisoning tends to occur soon after exposure to relatively high levels of a pesticide. The main activities leading to acute health effects of pesticides are work-related (e.g., applying pesticides, harvesting and packing treated crops, manufacturing pesticides), accidental exposure, or self-poisoning. In some cases, bystanders and residents living close to areas treated with pesticides may be acutely poisoned (e.g., through pesticide drift), as may be consumers of food contaminated with high levels of a pesticide. However, the latter cases are less common and would often be classified as accidents.

Pesticide poisoning can have different levels of severity, ranging from minor (with only mild, transient symptoms) to severe (with life-threating symptoms) or even death (Table 4.4-2). While mild pesticide poisoning symptoms are often transient, this is not always the case. (e.g., Kamel *et al*. 2007; Baldi *et al*. 2011). Epidemiological studies on such effects are relatively scarce.

Historical estimates of pesticide poisoning

The earliest published global estimates of acute unintentional pesticide poisoning date to the early 1970s, when the WHO Expert Committee on the Safe Use of Pesticides, using a model based on poisoning statistics from 19 countries, arrived at a first estimate of about 500,000 accidental/occupational poisoning cases per year (Copplestone 1977). Subsequently, Levine (1986) updated these estimates using data until the mid-1980s from a larger number of countries; these estimates of annual acute unintentional poisoning ranged from 800,000 to 1.5 million cases, double to triple those made a decade earlier.

WHO (1990) and Jeyaratnam (1990), using the data from Levine (1986) and additional poisoning statistics mainly from Asia, subsequently estimated that there were 1 million serious unintentional poisoning cases per year, of which 70 per cent were occupational and the rest accidental. Furthermore, 25 million minor poisonings per year were estimated to occur in developing countries only.

Litchfield (2005) discussed the shortcomings of the previous estimates, suggesting that they

Table 4.4-2 Poisoning severity score, as applied by the World Health Organization (WHO) and many national poison centres. Persson *et al*. (1998); Roberts and Reigart (2013); WHO (2020b).

Severity grade			Examples of symptoms or signs of pesticide poisoning
None	0	No symptoms or signs related to poisoning	
Minor	1	Mild, transient and spontaneously resolving symptoms	Irritation (respiratory, skin, eye), vertigo, headache, nausea, mild muscular pains
Moderate	2	Pronounced or prolonged symptoms	Pronounced or prolonged vomiting or diarrhoea, prolonged coughing, burns in the gastro-intestinal tract, confusion, agitation, unconsciousness, pronounced cholinergic/anticholinergic symptoms, infrequent, generalized or local seizures
Severe	3	Severe or life-threatening symptoms	Permanent eye damage, manifest respiratory insufficiency, pulmonary oedema, pulmonary fibrosis, frequent and generalized seizures, deep coma
Fatal	4	Death	

overestimated pesticide poisoning, but did not provide a comprehensive alternative. Others have argued that a large percentage of unintentional poisoning cases are not reported, suggesting that official data underestimate the real situation (see the chapter on poison centre statistics below).

Thereafter, no global estimates of unintentional pesticide poisoning were made until the effort by Boedeker *et al*. (2020) (see below). As a result, even recent reviews and discussions of acute pesticide poisoning (APP) (Jørs, Neupane and London 2018; Tyrell *et al*. 2019) cite estimates dating back 30 years which were made on the basis of even older poisoning statistics.

Poison centre statistics

Poison centres are an important source of information about pesticide poisoning. In addition to providing advice on, and assistance with, the prevention, diagnosis and management of poisoning, they often maintain databases on cases of poisoning. Nevertheless, in early 2021 only 47 per cent of WHO member states had one or more poison centres, with a lack of coverage especially in low income countries (WHO 2020c) (Figure 4.4-2).

No recent global or regional reviews of pesticide poisoning statistics compiled by poison centres have been published. However, a number of poison centres, or associations of poison centres, regularly publish their own data. In some countries specific pesticide poisoning monitoring programmes exist or have existed, such as the SENSOR (Sentinel Event Notification System for Occupational Risks)-Pesticides programme of the National Institute for Occupational Safety and Health (NIOSH) in the United States (Calvert *et al*. 2016) and the Pesticides Surveillance Study of the United Kingdom's National Poisons Information Service (NPIS) (Perry *et al*. 2014).

Poison centres use a variety of sources to compile statistics on pesticide poisoning (Figure 4.4-3). They include direct enquiries to a centre about possible poisoning cases (by health professionals or the public), cases recorded by medical facilities and subsequently reported to the centre (such notifications are mandatory in some countries), poisoning surveys carried out by the centre or other specialized institutions and researchers, and poisoning cases reported in the press.

Statistics reported by poison centres can therefore be quite variable or incomplete. If pesticide poisoning is relatively mild, cases will not usually be reported to medical practitioners at all. On the other hand, if severe pesticide poisoning leads to death before a medical facility is visited, a death certificate may not include pesticide poisoning as

Figure 4.4-2 Poison centres are primarily concentrated in high and middle income countries. They are much rarer in some low income parts of the world. WHO (2020c).

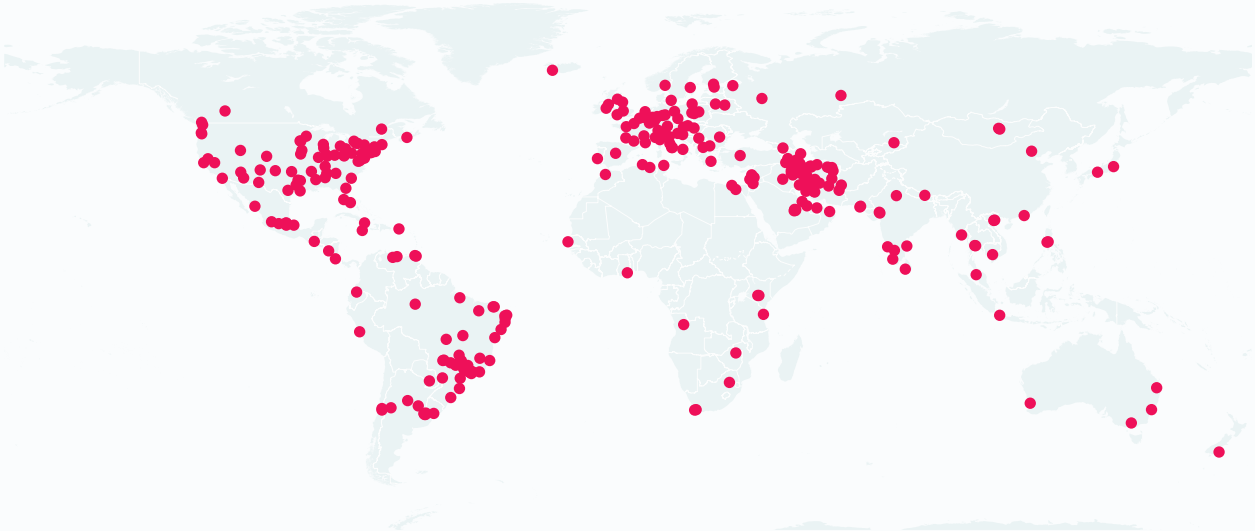
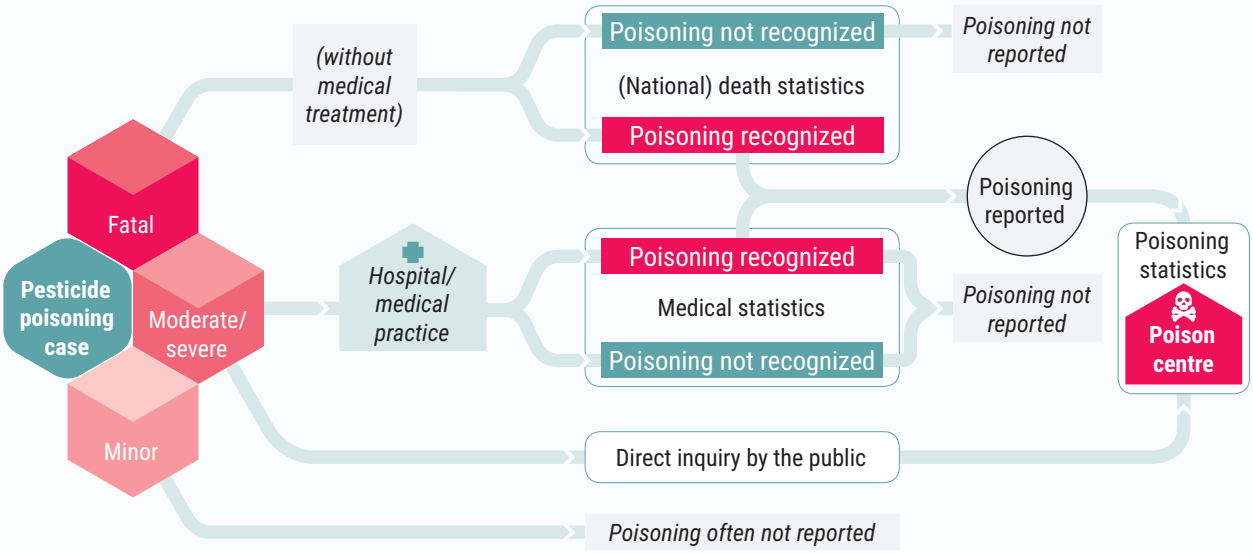


Figure 4.4-3 **Simplified representation of how pesticide poisoning information may, or may not, reach a poison centre database. Poison centres may use other sources to complete their poisoning statistics, such as surveillance reports, dedicated monitoring studies or media releases.** UNEP (2020).



the cause. Many countries and rural regions where pesticide poisoning occur do not have an adequate civil registration and vital statistics systems with which to record all deaths.

Even if a patient visits a doctor or hospital, the symptoms of pesticide poisoning may be non-specific and medical staff may therefore confuse them with symptoms of other health problems. Making diagnoses from non-specific signs of illness is often difficult, and patients will usually get better following acute poisoning, so that further investigations may not be carried out in busy health care systems. Moreover, decentralized medical facilities may not fully register cases of pesticide poisoning in a central database, for example because pesticide poisoning is not a notifiable event that requires mandatory registration

Pesticide poisoning statistics from poison centres have therefore been considered to be imprecise and variable over time, and generally to underestimate the real number of poisoning cases that occur (Calvert *et al*. 2016). Jeyaratnam (1990) suggested that, at least in developing countries, for every poisoning case reported to a hospital 25 milder cases were not reported. More recent studies suggest that 1-20 per cent (median value 6 per cent) of acute pesticide poisoning cases are reported in national poisoning statistics (Table 4.4-3). This is only slightly higher than the estimates by Jeyarathnam (1990) up to three decades earlier, indicating that under-reporting of pesticide poisoning remains an important problem.

There can be many reasons why people acutely affected by pesticides do not seek medical care or report to public health facilities. For example, they may not feel ill enough to justify the expense, or they may have become habituated to poisoning symptoms as "normal" farming practice; medical facilities may be too far away, especially in rural areas; or farm workers may not be covered by health insurance or may be afraid to lose their jobs if they report pesticide exposure (Roberts and Reigart 2013; Lekei, Ngowi and London 2016). Under-reporting at the level of medical facilities may be due to incorrect or incomplete diagnosis (see above), inadequate disease registration systems, or lack of communication within the public health system. While high income countries may have better poisoning registration systems than countries with fewer resources, under-reporting appears to occur to some degree everywhere (Table 4.4-3) (Lekei, Ngowi and London 2016; Prado *et al*. 2017).

WHO has developed tools to facilitate the collection of internationally harmonized data

Table 4.4-3 Levels of under-reporting of acute pesticide poisoning (APP) cases, as quantified in different parts of the world.

Country	People who experience APP seeking medical care (per cent)	Cases of medical care for APP reported in poisoning statistics (per cent)	Overall degree of reporting of APP in poisoning statistics (per cent)	Source
South Africa		5-20		London and Bailie (2001)
Costa Rice, El Salvador, Guatemala, Nicaraga, Panama			8 (range 1-20)	Henao and Arbelaez (2002)
Honduras	--	80	--	
Nicaragua		4.5		Corriols *et al*. (2008)
Tanzania (occupational poisoning only)	21	21	4.5	Lekei, Ngowi and London (2016)
United States (farm workers only)			12	Prado *et al*. (2017)
Morocco		15		Rhalem *et al*. (2012); Rhalem, N., personal communication (2019)
Federal District of Brazil (fatal poisoning only)			9	Magalhães and Caldas (2018)
United Kingdom		50 (estimate)		National Poisons Information Service (United Kingdom) (2019)

on poisoning, including a common vocabulary for describing poisoning cases and a poisoning severity score (a standardized scale for grading the severity of acute poisoning) (Persson *et al*. 1998; WHO 2020b) (Table 4.4-2). Thundiyil *et al*. (2008) collaborated with WHO staff to further develop an international classification tool for acute pesticide poisoning. The European Union (EU) has also started to harmonize rules pertaining to data collection and reporting of pesticide poisoning cases (Settimi *et al*. 2016).

The UNEP, in collaboration with WHO, recently compiled poison centre reports and conducted a call-in of pesticide poisoning data from the WHO informal network of poison centres (UNEP 2020). Data from 17 countries referring to the period 2009-2019 were assessed, the large majority of which concerned upper-middle and high income economies.

Average annual pesticide poisoning incidences were highly variable, ranging from 0.22 to 24 cases per 100,000 inhabitants (median value 4.9) (Table 4.4-4). There was no significant correlation between country income group and poisoning incidence. However, the fraction of total poisoning cases in a country attributable to pesticides increased with the percentage rural population in that country, suggesting a link with pesticide use in agriculture (Figure 4.4-4).

In about half of countries the majority of reported pesticide poisoning cases were unintentional (either accidental or occupational) (Table 4.4-5). There was not a clear regional predominance in regard to the different circumstances of exposure.

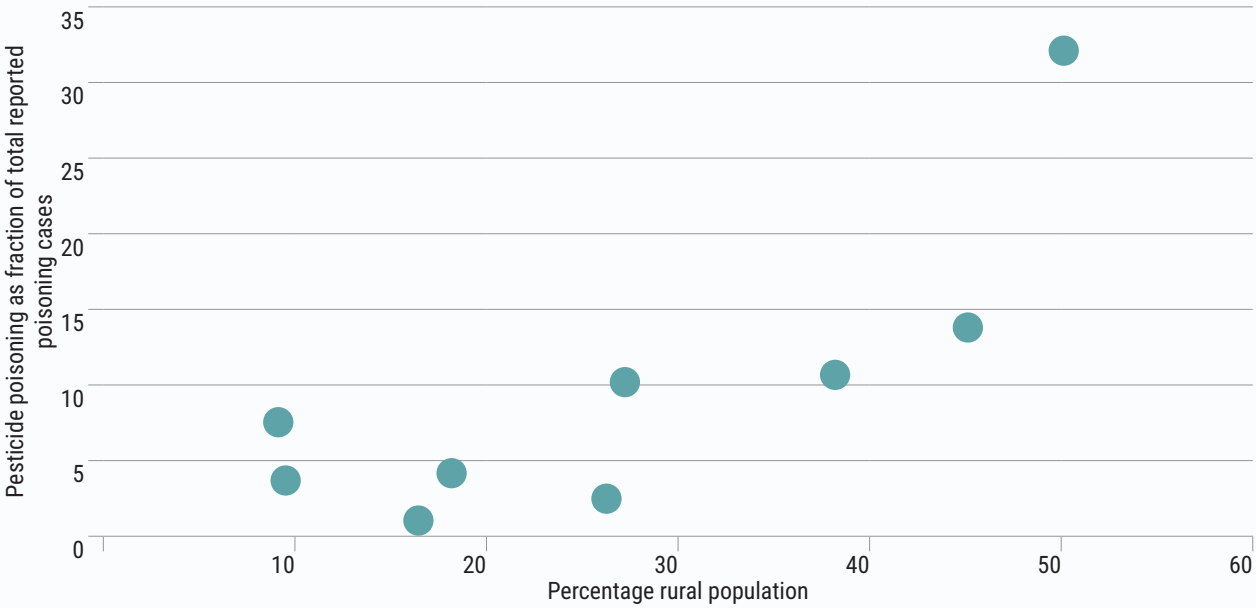
Cases of fatality due to pesticide poisoning reported to the poison centres examined in this exercise were positively correlated with the

Table 4.4-4 Average number of reported pesticide poisoning cases in countries with national coverage; averages calculated over the last three years reported (where available). UNEP (2020).

Country	Period	Average annual number of reported pesticide poisoning cases	Pesticide poisoning as a fraction of total poisoning (per cent)	Average annual pesticide poisoning incidence (per 100,000 inhabitants)
Algeria	2018	461	10.2	1.1[1]
Brazil	2012-2014	12,132	-- [3]	6.2[2]
Chile	2016-2018	560	--	3.0[1]
Guatemala	2012-2013, 2018	36	32	0.22[1]
Jamaica	2013-2015	56	13.8	2.0[1]
Jordan	2016-2018	115	7.6	1.2[1]
Morocco	2016-2019	1,499	10.7	4.9[1]
Netherlands	2015-2017	1,326	3.8	8.0[1]
Peru	2015-2017	2,244	--	12[2]
Republic of Korea	2012-2014	6,052	--	9.0[2]
Switzerland	2016-2018	812	2.5	9.6[1]
United Kingdom	2018-2019	1,030	~ 0.3-1	1.5-3.3[1]
United States	2015-2017	77,968	4.1	24[1]

[1]calculated based on total annual population (FAOSTAT 2020); [2]national report; [3] -- = not available;

Figure 4.4-4 Pesticide poisoning as a fraction of total reported poisoning cases is positively correlated with the percentage rural population in the respective countries. UNEP (2020).



fractions of pesticide poisoning cases that were suicides, suggesting that most deaths resulted from self-poisoning. In countries where pesticide poisoning was mainly unintentional, moderate and severe poisoning cases made up 1-44 per cent (median 16 per cent) of all reported poisonings, implying that the large majority of unintentional pesticide poisonings were mild (UNEP 2020).

No consistent trends in reported pesticide poisoning over time were found in the poison centre data evaluated in this exercise. Some countries (e.g., Brazil and Jordan) showed increases in reported cases during the last decade, while Chile, the Republic of Korea and the United States showed clear declines. Reports on pesticide poisoning remained relatively stable for the period 2009-2019 in Jamaica, Morocco, Switzerland and the United Kingdom.

Comparisons between countries are inevitably subject to a number of caveats. The extent of coverage of poison centres varies from country to country. It may also vary over time in countries with fewer resources. In addition, changes in the population's engagement with agriculture over time, or changes in agricultural practices such as types of crops grown, cannot be taken into account although these trends may differ between countries. As an example of differences between countries, many pesticide poisonings in the United Kingdom were caused by exposure to pre-diluted amateur pesticide formulations for household use (Perry *et al*. 2014); on the other hand, in low and middle income countries (e.g., Sri Lanka) the majority of pesticide poisonings were due to exposures to more toxic high-concentration agricultural formulations (Dawson *et al*. 2010).

The pesticide groups most reported to have caused poisoning are pyrethroid, organophosporous and carbamate insecticides, anticoagulant rodenticides (especially coumarins), glyphosate and disinfectants. The groups that led to the most severe cases of poisoning were organophosphorus insecticides, the fumigant aluminium phosphide, and bipyridylium herbicides (paraquat and diquat) (UNEP 2020).

Table 4.4-5 Fraction of pesticide poisoning according to circumstances of exposure and severity of pesticide poisoning cases, calculated as average percentage of moderate, severe and fatal cases (values are average percentages over the last three years reported). UNEP (2020).

Country	Period	Circumstances of exposure			Severity of poisoning
		Unintentional	Intentional	Other/ unknown	Moderate, severe and fatal poisoning cases
		per cent of total reported pesticide poisoning cases in the country			
Algeria	2018	70	28	2	*n.a.*[1]
Brazil	2012-2014	38	55	7	n.a.
Chile	2016-2018	87	13		17
Guatemala	2012-2013, 2018	60	30	10	n.a.
India	1999-2012	34	65	1	n.a.
Jordan	2016-2018	97	3		18
Morocco	2016-2018	60	39	1	37
Peru	2017	83	17		44
Philippines	2016-2018	49	50	1	n.a.
Poland	2004-2014	49	51		41
Switzerland	2016-2018	93	5	2	15
Uganda	2010-2014	37	63		n.a.
United Kingdom	2018-2019	92	8		1
United States	2015-2017	94	3	3	7

[1] *n.a.* = data not available

While poison centre statistics provide valuable information about the circumstances of pesticide poisoning, it is currently impossible to draw clear conclusions about differences in poisoning incidences among regions or country income groups. Poison centre data are also insufficient to extrapolate to larger geographical entities (UNEP 2020).

Acute pesticide poisoning studies

Acute unintentional pesticide poisoning has also been evaluated through dedicated studies and surveys looking at signs, symptoms and effects following exposure to pesticides. These are often cross-sectional studies that investigate a population (e.g., a group of farmers) at a specific point in time. Symptoms and health effects are examined by medical specialists, but they may also be self-reported (e.g., through a questionnaire).

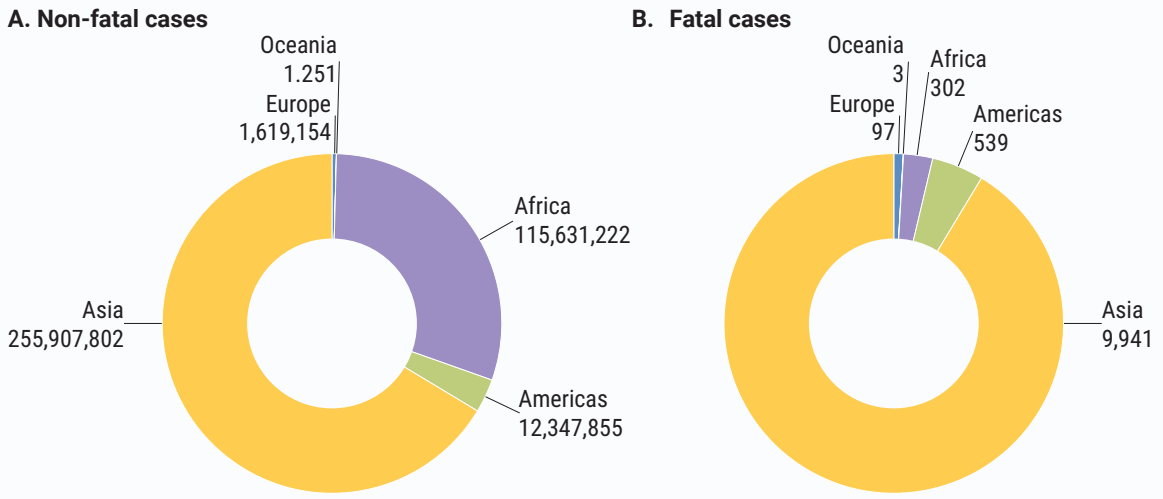
Since the 1980s such studies have shown a high incidence of acute pesticide poisoning in many parts of the world, including Central America (Henao and Arbelaz 2002; Wesseling, Corriols and Bravo 2005), Africa (London 2003; Lekei, Ngowi and London 2014; Lekei *et al*. 2020) and Asia (Ko *et al*. 2012). However, no detailed reviews of the outcomes of such studies have been available.

A major global systematic review of acute unintentional pesticides poisoning was conducted by Boedeker *et al*. (2020). A total 141 of countries were covered, including 58 using pesticide poisoning data from scientific articles published between 2006 and 2018 and an additional 83 using data from WHO's Cause of Death Query online (CoDQL), part of the WHO Mortality Database. Most of the scientific studies consulted focused on occupational poisoning of farmers and agricultural workers. Self-poisoning was excluded from the analysis (but see Chapter 4.4.4). All degrees of pesticide poisoning were combined in the assessment, ranging from mild symptoms to severe toxic effects.

This analysis found that the global incidence of unintentional acute pesticide poisoning was 43 per cent of the farming/occupational community (i.e., 43 cases of pesticide poisoning occur annually for every 100 persons working in agriculture). National incidences in the farming community were highly variable, ranging from <1 per cent in the United States to 84 per cent in Burkina Faso. Generally, poisoning incidences were higher in Africa and South and Southeast Asia than in the Americas, Europe and Oceania (Figure 4.4-5).

In 57 per cent of the reviewed studies poisoning was self-reported, while in 39 per cent

Figure 4.4-5 Regional distribution of the estimates of non-fatal and fatal cases of pesticide poisoning, all severities combined, estimated by Boedeker *et al*. (2020).



researchers or medical staff identified the cases. As self-reporting by non-specialized persons could have overestimated pesticide poisoning cases, the relatively high fraction of self-reported studies may have biased poisoning estimates. However, only an 11 per cent increase was found in the summary average across studies reporting poisoning incidence from self-reporting when compared with the summary average from studies using scientist-reported cases, suggesting the bias was modest.

On the basis of this data set, Boedeker *et al*. (2020) estimated that globally about 385 million cases of unintentional acute pesticide poisoning occur annually, all poisoning severities combined. They also estimated that about 10,900 fatal cases per year result from unintentional acute pesticide poisoning.

Another recent report evaluated studies from 11 countries in Eastern Europe, Central Asia and Africa, all of which included self-reported cases of acute pesticide poisoning among smallholder farmers and farm workers (Pesticide Action Network UK [PAN UK] 2020). Similarly to the review by Boedeker *et al*., high annual incidences of acute pesticide poisoning in farming communities were observed, ranging from 10 per cent in Moldova to 82 per cent in Belarus. Extrapolating from these results, the authors suggest that over 200 million farmers could be poisoned by pesticides every year.

The estimates of unintentional pesticide poisoning by Boedeker *et al*. (2020) and PAN UK (2020) are an order of magnitude higher than previous estimates. This may be because pesticide use has greatly increased since the 1990s, especially in Asia and Latin America. Another reason could be that the 1990 estimates were based primarily on national poisoning statistics (which tend to under-report incidences of poisoning in the field) while the recent estimates are mainly based on dedicated pesticide poisoning surveys. The significance of the latter explanation is underlined by the relatively low national pesticide poisoning incidences reported more recently by poison centres (Table 4.4-4) even when the fraction of the rural/farming population in these countries is taken into account.

Boedeker *et al*. (2020) do not distinguish among different poisoning severities in their estimates; the 385 million cases include mild effects as well as moderate and severe cases. It is not clear what fraction of these cases would require access to health services and/or medical treatment. The chapter on poison centre statistics, however, indicates that approximately 6 per cent of pesticide poisoning cases were reported to poison centres and 16 per cent of the reported cases were moderate or severe. Combining these figures would result in an estimate of a minimum of 3.7 million cases of moderate or severe non-fatal cases of unintentional pesticide poisoning annually, about four times more than previous estimates.

On the other hand, the estimate by Boedeker *et al*. (2020) that there are about 11,000 unintentional fatal poisonings is lower than previous ones, suggesting such cases are currently less common because treatment of severe poisoning is more effective or because the most hazardous pesticides are being withdrawn globally from agricultural use.

Overall, recent reviews of poisoning studies and surveys suggest that unintentional acute pesticide poisoning remains widespread. Moreover, it appears to have increased significantly compared to the often cited estimates from the 1990s, especially in farming communities in low and middle income countries.

However, data from different sources are variable and incomplete. There is a clear need to conduct more pesticide exposure and poisoning surveys following standardized protocols. Furthermore, the value of pesticide poisoning statistics from poison centres would increase considerably if information were gathered about reporting rates.

It should be noted that unintentional acute poisonings are only part of the acute impact of pesticides on health, and that incidences, case fatality and trends over time are different for self-poisoning (Chapter 4.4.4).

Box 4.4-1 Aspects of pesticide epidemiology. Bradford Hill (1965); Bonita *et al*. (2006); EFSA PPR Panel (2017).

With respect to pesticides, the science of epidemiology investigates whether pesticides are associated with the distribution (frequency, pattern) of a disease[1] or health problem in humans, with the ultimate aim to control such problems.

All pesticide epidemiological studies are observational, in the sense that the investigator does not intervene, i.e. does not determine the degree of exposure of study persons to the pesticide. Experimental epidemiological studies with pesticides are not conducted as it is not considered ethical to willingly expose persons to a pesticide and study its health effects. However, there is a growing body of indirect evidence assessing the efficacy of various policies and practices on pesticide exposure and thus on the burden of pesticide-related diseases.

Different types of observational epidemiological studies can be recognized:

Ecological (or correlational) study: Comparison of aggregate disease estimates in different groups of people in different localities (or the same group of people at different times) with respect to the potential aggregate exposure estimates to a pesticide in this group.

Example result: The percentage of persons with dermatitis at the time of the study is higher in a region with high pesticide use compared to a region with low use.

This type of study can only identity correlations, using the location under study as the unit of analysis, but cannot determine causality; it is often exploratory in nature and may start more in-depth investigations.

Cross-sectional (or prevalence) study: The prevalence of a disease and possible pesticide exposure factors are investigated in a selected group of people at a specific point in time and at the individual participant level. This is often done through surveys.

Example result: Persons in the selected group of people who applied a pesticide three times or more during the last growing season, suffered more from headaches than those who had not applied pesticides.

This type of study can identify correlations, but can generally not determine causality. It is also exploratory in nature and can help to identify risk factors suitable for further scrutiny.

Case-control study: Persons having a disease (cases) are compared with persons without such an effect (controls) for the occurrence or degree of past exposure to a pesticide. The investigators measure disease occurrence at one point in time and try to identify exposure to the pesticide in the past. The main difficulty with case-control studies is to quantify exposure to the pesticide, which occurred retrospectively. This type of study is in particular useful for uncommon diseases.

Example result: Persons having a type of cancer were significantly more exposed to a pesticide during the previous 10 years than persons not having that type of cancer.

With appropriate controls and a good estimation of past exposure to the pesticide, case-control studies can be an important piece of evidence for validating postulated associations and moving towards establishing causality.

Cohort (or follow-up) study: A group of persons free of a disease (a cohort) is classified into subgroups according to exposure to a pesticide. The whole cohort is then followed (often up to many years) to see if a disease develops, and whether this development differs between the groups having different levels of exposure to the pesticide.

Example result: Those persons who, in the course of the study, were more exposed to a pesticide developed significantly more neurological deficiencies at a later age than persons who were less exposed during the study period.

Since the time sequence from exposure to disease can be determined, cohort studies provide the best information about the causation of disease. It is worth noting that the study design per se does not preclude the general study quality. Also, even when the exposure is captured prospectively, it may still be difficult to determine and quantify it.

Causal inference is the scientific process of determining whether observed associations between pesticide exposure and a disease are likely to be causal. The process of judging causation can be difficult and contentious, and pesticide epidemiology is no exception. Epidemiologists systematically evaluate various considerations for causation, either quantitatively or empirically; The "Bradford Hill criteria" is an example of a set of such considerations used empirically. They include questions such as: "does the cause precede the effect?", "have similar results been shown in other studies?", "is increased exposure associated with increased effect?, "is a toxicological mechanism known that can result in the effect?", and others. Based on causal inference methodology, epidemiologists can make a judgement about the likelihood that the pesticide may have caused the disease.

1 The term "disease" is used here for all types of health effects that may be caused by pesticides

4.4.3 Chronic pesticide poisoning

Epidemiology of pesticide exposure

Chronic exposure to pesticides has been associated with a wide variety of adverse health effects. They include different types of cancer, reduced nervous system functions, disturbed neurodevelopment of children, diabetes, decreased male and female fertility, birth defects and Parkinson's disease, among others (American Academy of Pediatrics 2012; Bergman et al. 2013; Mostafalou and Abdollahi 2013; Blair *et al*. 2015; Hertz-Picciotto *et al*. 2018).

One of the most recent WHO *Global Assessment of the Burden of Disease from Environmental Risks* report indicates positive associations between pesticides and chronic disease, such as certain cancers and Parkinson's disease (Prüss-Üstün *et al*. 2016). However, evidence for causal relationships was often considered limited.

Studying the presence and magnitude of associations between exposure to pesticides and health outcomes in humans is a challenging field of epidemiology. Its complexity lies in specific characteristics of pesticide epidemiology, such as the large number of active substances in the market; the number of different pesticides which may be used by an individual farmer; and the different patterns of pesticide use, the variety of study designs and the inherent limitations of each design; the long period that often passes between exposure to a pesticide and the development of disease; the frequent lack of quantitative (and qualitative) data on exposure to individual pesticides; and the other chemicals that might be associated with a condition being studied (Blair *et al*. 2015; EFSA PPR Panel 2017).

Ohlander *et al*. (2020) conducted a global review of methods used to assess exposure to pesticides in occupational epidemiology studies. The large majority of exposure assessment methods were indirect, based in particular on self-reported exposures. Direct methods such as biomonitoring of blood or urine were used in only 21 per cent of studies. This situation did not change during the 25 years covered by the study, indicating that imprecise exposure estimations continue to hamper the determination of clear associations between pesticides and health outcomes.

It is also generally considered that while epidemiological studies may point to an association between pesticide exposure and a health outcome, it is very difficult to prove that a pesticide has caused an adverse health effect based on epidemiological studies alone.

Table 4.4-6 Examples of large cohort studies addressing the health effects of pesticide exposure. IARC (2020).

Country	Study	Number and type of participants	Year of recruitment
United State (Iowa and North Carolina)	Agricultural Health Study (AHS)	52,394 licensed private pesticide applicators (mostly farmers) 32,345 of their spouses 4,916 commercial pesticide applicators	1993-1997
United States (Wisconsin)	Marshfield Epidemiologic Study Area (MESA) – Farm Resident Cohort	5,432 farm residents	1998
United States (California)	Center for the Health Assessment of Mothers and Children of Salinas (CHAMACOS) Study	Pregnant women, and subsequently >800 children born from these women	1999
France	AGRIculture and CANcer (Agrican) cohort study	187,471 persons within the agricultural population	2005-2007
Australia	Pesticide-exposed workers cohort	14,092 persons, of which 4,775 agricultural workers	1960s- 1980s
Norway	Norway Farmer Cohort	8,482 farmers	1990-1992

More evidence is often needed to infer such causality (Box 4.4-1).

With the aim of obtaining better data on the long-term effects of exposure to pesticides, a number of dedicated longitudinal studies have been carried out. In cohort studies populations at risk (e.g., farmers, agricultural workers, rural families) are followed over a long period to determine the occurrence of disease (Box 4.4-1). However, achieving valid and precise exposure estimates remains difficult even in large cohort studies (EFSA PPR Panel 2017). A particular challenge is that while disease outcomes are ascertained after long follow-up periods, exposure estimates (or measurement of biomarkers) will inevitably be based on a limited time frame even in the case of very large, well-resourced studies.

Thus, trends or changes in the lifetime pesticide exposure patterns of the study participants are not captured and, unlike 'traditional' health-related behaviours (e.g., smoking) which may remain relatively stable for large periods of time, these patterns are not adequately studied for pesticides.

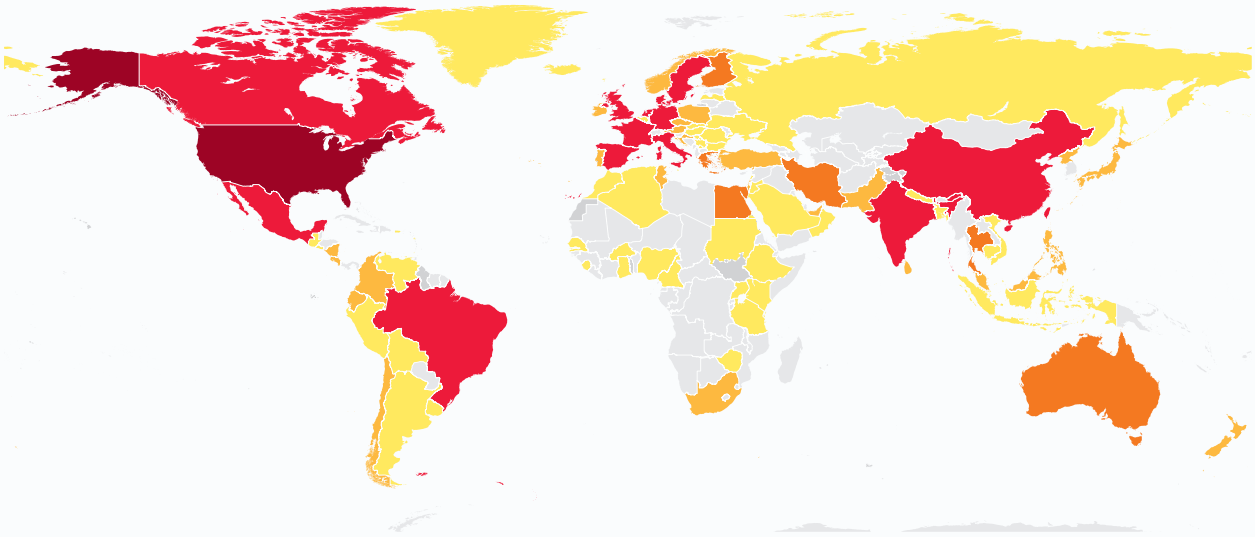
To coordinate activities and combine forces, in 2010 the United States National Cancer Institute (NCI) and the International Agency for Research on Cancer (IARC) established AGRICOH, a consortium of agricultural cohort studies (International Agency for Research on Cancer [IARC] 2020). As of June 2020, 28 cohorts from five continents were participating in AGRICOH. Some examples are shown in Table 4.4-6.

Reviews of chronic pesticide effects

Numerous epidemiological studies have been conducted with the aim of assessing associations between exposure to pesticides and health outcomes such as the development of specific diseases. However, the majority of studies on occupational exposure to pesticides (83 per cent) originate in high and upper-middle income countries. Very few (1.1 per cent) are from low income countries (Figure 4.4-6) (Ohlander *et al.* 2020).

Three comprehensive reviews of the multiple chronic adverse effects of pesticides have been

Figure 4.4-6 **The large majority of studies on occupational exposure to pesticides have been conducted in high and upper-middle income countries.**



Note: Shown is the number of articles on occupational exposure to pesticides and various health outcomes published between 1993 to 2017 per country serving as study location, retained in the review by Ohlander ***et al.*** 2020. Numbers in brackets represent the number of different study locations (countries).

Table 4.4-7 Characteristics of the three major reviews of chronic adverse health effects of pesticides.

Parameter	Inserm (2013)	Ntzani *et al.* (2013)	Ntzani *et al.* (2020)
Publication period	Up to June 2012	1 January 2006-30 September 2012	1 January 2006-30 June 2019
Number of publications included	Not summarized	602	1,166 (including the studies covered by Ntzani *et al.* 2013)
Number of health outcomes assessed	18	24 disease categories	29 disease categories
Method of quality control of publications described	No	Yes	Yes
Meta-analyses performed	No	Yes	Yes

published in the last 10 years (Table 4.4-7). In France, the National Institute for Health and Medical Research carried out an extensive review of the relationship between pesticide exposure and 16 different health outcomes and a formal meta-analysis was not carried out (Institut national de la santé et de la recherche [Inserm] 2013) (Table 4.4-7). At the request of the European Food Safety Authority (EFSA), Ntzani *et al*. (2013) conducted a systematic review as well as multiple meta-analyses of epidemiological studies published until 2012 examining the association between pesticide exposure and health outcomes in 24 major disease categories.

Since these two evaluations, a large number of epidemiological studies and reviews on this topic have been published due, in particular, to increased interest in the health effects of pesticides but also to the fact that a number of large cohort studies initiated years ago have begun to yield results. An update of the review done for EFSA (Ntzani *et al*. 2013) was therefore commissioned for this report, including all epidemiological studies published between January 2012 and June 2019 (Ntzani et al. 2020). It produced a database containing a total of 19,178 possible associations between pesticides and health outcomes (see Box 4.4-2 for more details). The conclusions of these three reviews regarding these possible associations between exposure and disease or other health outcomes are summarized in Table 4.4-8.

Based on the review and meta-analyses by Ntzani *et al*. (2020), positive associations between pesticides and certain health outcomes were reported (Table 4.4-8). Specifically:

- non-Hodgkin lymphoma and exposure to pesticides in general and insecticides in general;
- Parkinson's disease and exposure to pesticides in general, herbicides in general, insecticides in general, fungicides in general, organochlorine insecticides in general, and certain individual organophosphate insecticides;
- Parkinsonism and exposure to pesticides in general;
- diabetes and exposure to pesticides in general, organochlorine pesticides in general, and certain individual organochlorines;
- asthma and exposure to pesticides in general;
- spontaneous pregnancy loss and exposure to pesticides in general;
- childhood leukemia and maternal exposure during pregnancy to pesticides in general, herbicides or insecticides;
- childhood leukemia and paternal exposure or childhood exposure to pesticides in general.

Based on the epidemiological evidence, chronic adverse health effects are considered likely as a result of exposure to pesticides in general or to certain groups of pesticides. In only a few cases were associations between specific pesticides and a health outcomes characterized and quantified. This difficulty is inherent in regard to identifying specific exposures in epidemiological studies. However, such limitations do not necessarily preclude taking risk reduction measures before risk characterization becomes more precise.

Box 4.4-3 Systematic review and meta-analyses of epidemiological studies examining the association between pesticide exposure and health outcomes.

At the request of the European Food Safety Authority, Ntzani *et al.* (2013) conducted a systematic literature review of epidemiological studies examining the association between pesticide exposure and any health outcome published after 2006. In 2019, UNEP solicited the Department of Hygiene and Epidemiology of the University of Ioannina School of Medicine, in Greece, to update their review. The results of that work are available as a technical support document to the current report (Ntzani *et al.* 2020).

Method

The updated systematic review in included observational (cohort, cross-sectional and case-control) studies assessing the association between pesticide exposure and health-related outcomes in adult, adolescents, or children published between January 2012 and June 2019.

Literature searches were conducted using PubMed, EMBASE, TOXNET OpenSigle and ProQuest Digital Dissertations and Theses; publications were identified without language or geographical restrictions.

Only studies with sufficient quantitative information to estimate effect sizes were included in the assessment. Studies assessing the effects of acute or accidental pesticide exposure, or of pesticide poisoning, were excluded.

Relevant information was extracted from eligible studies and recorded in a spreadsheet database.

Meta-regression analyses (MA), using random effect models, were carried out for specific disease entities, by pesticide group and window of exposure where available. Effects sizes were expressed as Relative Risks (RR) and corresponding 95% confidence intervals for binary study outcomes, or Summary Mean Differences and corresponding standard deviations for continuous outcomes.

Results

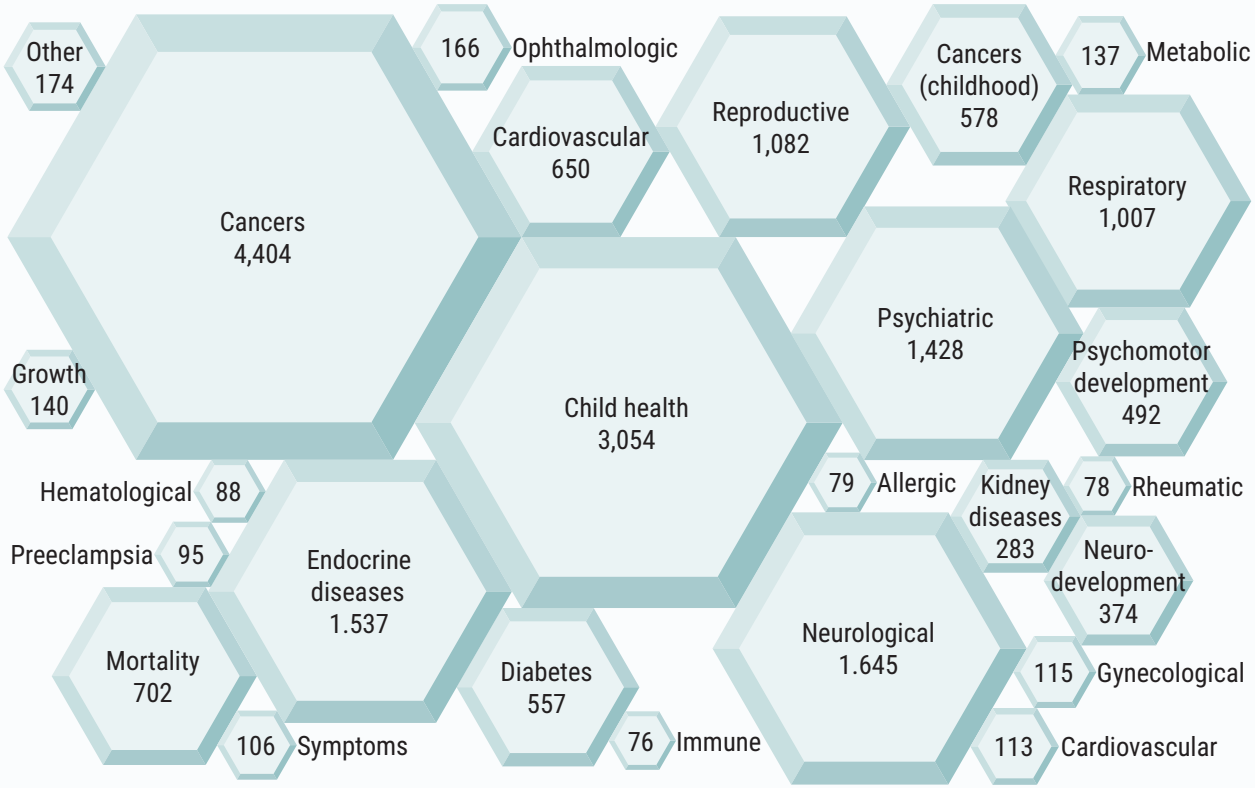
Based on the criteria used, and covering the entire period 2006 – 2019, 1,166 publications were considered eligible.

The largest number of publications was from the Americas (45%) followed by Europe (29%), Asia (17%) and Africa (4%). Sample size ranged from studies as small as including 37 participants to large cohorts with 1,832,969 participants. Cross sectional studies (34%), cohort studies (29%) and case control studies (36%) were present in comparable frequencies.

More than 19,000 postulated associations (i.e. links between an individual pesticide (group) and a specific health outcome) have been included in the database.

Most frequently studied health outcomes were cancers, child diseases, neurological diseases, neuropsychiatric disorders and endocrine system diseases (see bubble plot).

Results of assessments where a meta-analysis was possible are listed in Table 4.4-9. Health outcomes reviewed but for which meta-analyses was not feasible are provided in Annex 4.1-1.[1]



Relative importance of health outcomes studied in pesticide epidemiological studies published between 2006 and 2019. N = total number of studied associations

[1] Annex 4.4-1 is found at: https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing

Table 4.4-8 Associations between pesticide exposure and diseases or other health outcomes, as concluded in two recent comprehensive reviews. Only associations for which a meta-analysis was performed by Ntzani *et al.* (2020) are listed. Associations studied without a meta-analysis are listed in Annex 4.4-1.

Health outcome	Pesticide or pesticide class	Inserm (2013) *(systematic review)*	Ntzani *et al.* (2020); Karalexi *et al.* (submitted) *(systematic review and meta-analyses[1])*	
		Positive association	Positive association	Number of studies
Effects on adults				
Non-Hodgkin lymphoma	Pesticides[2]		Yes	11
	Insecticides		Yes	4
	Organochlorines		No	4
	Dieldrin		No	3
	DDT	Yes	No	6
	DDE		No	9
	Oxychlordane		Yes	4
	Trans-nonachlor		No	4
	HCB		No	7
	Organophosphates	Yes	No	3
	Chlorpyrifos		No	4
	Malathion	Yes	No	3
	Diazinon		No	3
	Terbufos		No	3
	Carbaryl		No	4
	Carbofuran		No	3
	Permethrin		No	3
	Herbicides/phenoxy herbicides		No	6
	Glyphosate		No	3
	MCPA		No	3
Follicular lymphoma/ follicular B-cell carcinoma	DDE		No	4
Diffuse large B-cell lymphoma/diffuse large cell lymphoma	DDE		No	5
	HCB		No	3
Prostate cancer	Pesticides	Yes	No	8
	Captan		No	3
	Chlordane		No	3
	Oxychlordane		No	4
	DDE		No	5
	DDT		No	4
	Dieldrin		No	3
	HCB		No	3
	Trans-nonachlor		No	3
	Organochlorine insecticides		No	3
Breast cancer	Pesticides		Yes	8
	DDT		No	5
	DDE		No	7
	HCB		No	3

Health outcome	Pesticide or pesticide class	Inserm (2013) *(systematic review)*	Ntzani *et al.* (2020); Karalexi *et al.* (submitted) *(systematic review and meta-analyses[1])*	
		Positive association	Positive association	Number of studies
Bladder cancer	Pesticides		No	6
Lung cancer	Pesticides		No	3
Testicular cancer	DDE		No	5
Parkinson's disease	Pesticides	Yes	Yes	29
	Herbicides (broad definition)	Yes	Yes	9
	Paraquat		No	9
	Insecticides (broad definition)	Yes	Yes	10
	Organochlorine insecticides	Yes	Yes	4
	Rotenone		Yes	3
	2,4-D		No	4
	Benomyl		No	3
	Chlorpyrifos		No	4
	DDT		No	5
	Diazinon		Yes	5
	Dimethoate		No	3
	Malathion		Yes	4
	Mancozeb		No	3
	Maneb		No	5
	Parathion		No	6
	Ziram		No	3
	Fungicides (broad definition)		Yes	6
	Carbamate		Yes	4
	Organophosphorus		Yes	7
Parkinsonism	Pesticides		Yes	5
Diabetes (general)	Pesticides		Yes	6
	Organochlorine insecticides		No	3
	DDE		Yes	11
	DDT		No	7
	Dieldrin		No	4
	HCB		Yes	4
	Mirex		No	3
	Fungicides		No	3
	Herbicides		No	3
	Insecticides		No	3
Diabetes type 2	Organochlorine pesticides		Yes	3
	DDT		Yes	4
	DDE		Yes	11
	HCB		Yes	6
	Oxychlordane		Yes	5
	Trans-nonachlor		Yes	6
Gestational diabetes	DDE		No	3

Health outcome	Pesticide or pesticide class	Inserm (2013) *(systematic review)*	Ntzani *et al*. (2020); Karalexi *et al*. (submitted) *(systematic review and meta-analyses[1])*	
		Positive association	Positive association	Number of studies
Asthma (in adults)	Pesticides		Yes	6
Depression	Pesticides		No	6
Spontaneous abortion	Pesticides		Yes	5
Effects on child development				
Childhood acute leukemia (overall)	Pesticides (maternal exposure, any time)		No	3
	Pesticides (maternal exposure, during pregnancy)		Yes	11
	Herbicides (maternal exposure, during pregnancy		Yes	3
	Insecticides (maternal exposure, during pregnancy)		Yes	3
	Pesticides (paternal exposure, any time)		Yes	4
	Pesticides (paternal exposure, during preconception)		No	3
	Pesticides (parental exposure, any time)		Yes	6
	Pesticides (childhood exposure)		Yes	5
Childhood acute lymphoblastic leukemia	Insecticides (maternal exposure, any time)		No	3
	Pesticides (maternal exposure, during pregnancy)		Yes	3
	Herbicides (maternal exposure, during pregnancy)		No	3
	Insecticides (maternal exposure, during pregnancy)		No	4
	Pesticides (childhood exposure)		No	3
Childhood leukemia	Pesticides (exposure during pregnancy or childhood)	Yes		

1 Meta-analyses were only conducted if at least three relevant studies were available.
2 Pesticides means pesticides in general, without distinction of specific classes or active ingredients.

4.4.4 Self-poisoning

Self-poisoning involves exposure to both the active ingredient and co-formulants, such as solvents and surfactants, which may have their own toxicity, especially when ingested in large quantities, and be more toxic than the active ingredient itself.

Unlike unintentional acute pesticide poisoning, intentional self-poisoning with pesticides (often with the intent to attempt suicide) has been evaluated in detail since the early 1990s. Recent systematic reviews were conducted for the period 1990-2007 (Gunnel *et al*. 2007) and for 2006-2015 (Mew *et al*. 2017; Karunarathne *et al*. 2019). The number of suicides due to pesticides in the early 2000s was estimated at 372,000 cases per year, or 31 per cent of global suicides (Gunnel *et al*. 2007). In the mid-2010s pesticide suicides were estimated at 168,000 per year, or about 20 per cent of all suicides globally (Mew *et al*. 2017) (Table 4.4-9). The authors estimated that the annual total number of pesticide self-poisonings ranged from 1 to 2 million. More recently, Karunarathne *et al*. (2019) calculated that there had been more than 14 million deaths due to self-poisoning with pesticides between 1960, when synthetic organic pesticides started to become available on a large scale, and 2018. More than 95 per cent of these deaths occurred

Table 4.4-14 Global estimates of self-poisoning and suicides (deaths) by pesticides.

Study period	Estimated number of pesticide deaths from suicides per year	Estimated number of pesticide self-poisonings per year	Source
1980s	200,000	2 million	WHO (1990)
1990-2007	372,000		Gunnell *et al*. (2007)
2006-2015	168,000	1-2 million	Mew *et al*. (2017)
2015-2018	134,000		Karunarathne *et al*. (2019)
	Estimated total cumulative number of suicides		
1960-2018	14.2-14.9 million		Karunarathne *et al*. (2019)

in low and middle income countries (WHO and FAO 2019a).

Suicide by intentional pesticide ingestion primarily occurs in rural areas of low and middle income countries in Africa, Central America, Southeast Asia and the Western Pacific (WHO 2014). The introduction of hazardous pesticides into rural communities during the Green Revolution resulted in a large increase in pesticide suicide deaths. Self-poisoning is often carried out spontaneously, with little planning and little time to change one's mind (Eddleston and Phillips 2004). The easy availability of these pesticides has greatly increased the lethality of self-poisoning, changing their outcomes from non-fatal to fatal (Karunarathne *et al*. 2019). The number of suicides has not been found to be correlated with the volume of pesticides sold. It is rather the pattern of pesticide use and products' toxicity that influence the likelihood they will be used for self-poisoning (Gunnell *et al*. 2007).

Pesticide suicides peaked in the 1990s and their incidence started to decline around 2000 (Karunarathne *et al*. 2019). The almost 65 per cent reduction in pesticide suicides between 1990 and 2018 occurred in the context of a 9 per cent decrease in the WHO estimate of the overall number of suicides between 2000 and 2012 (WHO 2014). The main drivers of this change are likely to be marked economic growth in parts of the world (e.g., China, where about half the world's pesticide suicides previously occurred), population shifts from rural to urban areas, further mechanization of agriculture, and the introduction of regulations to ban or reduce access to the most toxic pesticides (Mew *et al*. 2017; Karunarathne *et al*. 2019). Bans on Highly Hazardous Pesticides (HHPs) in Sri Lanka, the Republic of Korea, Bangladesh and other countries have been associated with marked falls in both pesticide suicides and all suicides (Gunnell *et al*. 2017).

It has been reported from various countries that banning the most toxic pesticides has not been associated with adverse effects on agricultural yield (Manuweera *et al*. 2008; Cha *et al*. 2016; Chowdhury *et al*. 2018; Bonvoisin *et al*. 2020). Two estimates of the health care costs of treating pesticide self-poisoning cases in Sri Lanka have been published, showing that treatment costs for patients with severe pesticide poisoning greatly exceed all other costs for poisoned patients (Ahrensberg *et al*. 2019). Recently, Lee *et al*. (2020), using an economic modelling study across 14 countries, concluded that national bans of Highly Hazardous Pesticides (HHPs) are a potentially cost-effective and affordable intervention for reducing suicide deaths in countries with a high burden of suicides attributable to pesticides.

For many years pesticide self-poisoning was ignored as a public health and agricultural problem because it was considered a form of misuse. A number of parties to international treaties have argued against including pesticide self-poisoning in relevant provisions because self-poisoning was considered a form of misuse. It has therefore been excluded from international treaties such as the Rotterdam Convention. However, it has been argued that pesticide self-poisoning should be considered an occupational condition, akin to

alcoholism in bar staff, due to their availability and occupational use in small-scale farming communities where resources to store and use then safely are lacking (Eddleston 2018).

Despite the downward trend in pesticide suicide deaths during the last ten to 20 years, self-poisoning with these products is still the cause of one in five of the world's suicides (WHO and FAO 2019a). WHO and FAO have published guidance on approaches that regulators can take to prevent pesticide suicides (WHO and FAO 2019a).

4.4.5 Factors affecting occupational and residential pesticide exposure

Occupational risk factors

Occupational pesticide exposure and the risk of subsequent adverse health effects may be influenced by a variety of factors related to the demography of the populations involved, pesticide application practices, the organization of the work environment, and workplace behaviours. These factors include experience with pesticide

Table 4.4-10 Factors found to increase or reduce occupational pesticide exposure and/or health effects: outcomes from selected reviews and medium- or large-scale studies.

Factor	Study number **						
	1	2	3	4	5	6	7
Demographic factors							
Younger age			↑*	↔			
Higher level of education			↔	↔			
Longer experience, confidence in practices			↓	↔			
More lifetime days of pesticide application					↑		↔
Large farm size					↑		
Pesticide application and work environment							
Control of all decisions on the farm			↓				
Pesticide safety training	●	↓					↓
Higher dose, frequency, toxicity of the pesticide	↑					↑	↑
Previous pesticide incident			↑				
Use of personal protective equipment	↓	↔	↔	↔	↓	↓	
Use of backpack sprayers				↑	↔		
Leaking or contaminated spray equipment			↑				
Workplace behaviours							
Use of pesticide in conflict with label		↑			↑		
Hand washing after work	↓						
Showering, changing clothes after work	●				↔		
Pesticide drift, application against the wind		↑			↑		
Early re-entry into sprayed fields		↑				↑	

* ↓: found to reduce exposure/health effects; ↑: found to increase exposure/health effects; ↔: not found to have an effect on exposure/health; ●: insufficient evidence to draw conclusion

** Studies:

1. Quandt *et al*. (2006): Review of 80 studies published since 1990; farmworker pesticide exposure, mainly in North America
2. Calvert *et al*. (2008): Analysis of pesticide poisoning cases in the United States from 1998 to 2005; factors linked to moderate or severe poisoning
3. Tomenson and Matthews (2009): Study of smallholder farmers in 24 countries worldwide; knowledge, attitude and practice (KAP) surveys linked to self-reported moderate or severe health effects
4. Pasiani *et al*. (2012): Study of smallholder farmers in mid-western Brazil; KAP study linked self-reported health effects and cholinesterase inhibition
5. Kim *et al*. (2013): Study of male farmers across the Republic of Korea, linking practices to self-reported moderate or severe health effects
6. Jørs (2016): Review of 20 studies published between 1995 and 2014, linking specific risk factors (focus on personal protection and the degree of pesticide exposure) with self-reported acute pesticide poisoning and/or cholinesterase inhibition
7. Fuhriman *et al*. (2019): Study of predictors of pesticide exposure of smallholder farmers in Costa Rica using a pesticide exposure algorithm

application, pesticide safety training, work hygiene, the type and maintenance of spray equipment, and use of personal protective equipment (PPE) (Table 4.4-10).

Many factors have been proposed to have direct, indirect or modifying effects on the extent to which farmers and farmworkers are exposed to pesticides; taking these factors into account often forms the basis for pesticide risk reduction measures (Quandt *et al*. 2006). However, when Quandt *et al*. (2006) conducted a first comprehensive review of workplace, household and personal predictors of pesticide exposure of farm workers, primarily in North America, they observed that research connecting the characteristics of workers' environments and behaviours with actual measurements of pesticide exposure was meagre. A more recent partial review of factors affecting acute pesticide poisoning in low and middle income countries (Jørs 2016) also yielded few quantitative data. No current comprehensive review quantifying the importance of risk factors for pesticide exposure and poisoning appears to be available.

Table 4.4-10 summarizes the outcomes of the above-mentioned reviews, as well as selected medium and large scale studies. It is striking that there are a large variety of outcomes, with certain risk factors reducing or increasing risk in some studies but not in others. This seems to indicate that whether a certain factor impacts pesticide exposure or health effects greatly depends on local conditions of use.

Demographic factors such as age, education, and experience with using pesticides have not been found to consistently increase or decrease pesticide exposure or effects. Factors linked with pesticide application and workplace organization have been evaluated more often. Using pesticides in higher doses or frequencies, or with higher toxicity, was found to increase health effects. Training in judicious pesticide use mostly appears to reduce pesticide exposure, although broader reviews of the impact of training on occupational health and safety have concluded that the longer-term impact of training – if conducted in isolation – is rather limited (Chapter 2.7.21).

Workplace behaviours in conflict with label instructions, good application practices or minimum occupational hygiene, on the other hand, were consistently found to increase pesticide exposure or effects.

Overall, studies quantifying the effects of workplace behaviours, demographic factors or pesticide application practices on occupational exposure or health effects remain infrequent.

Personal protective equipment

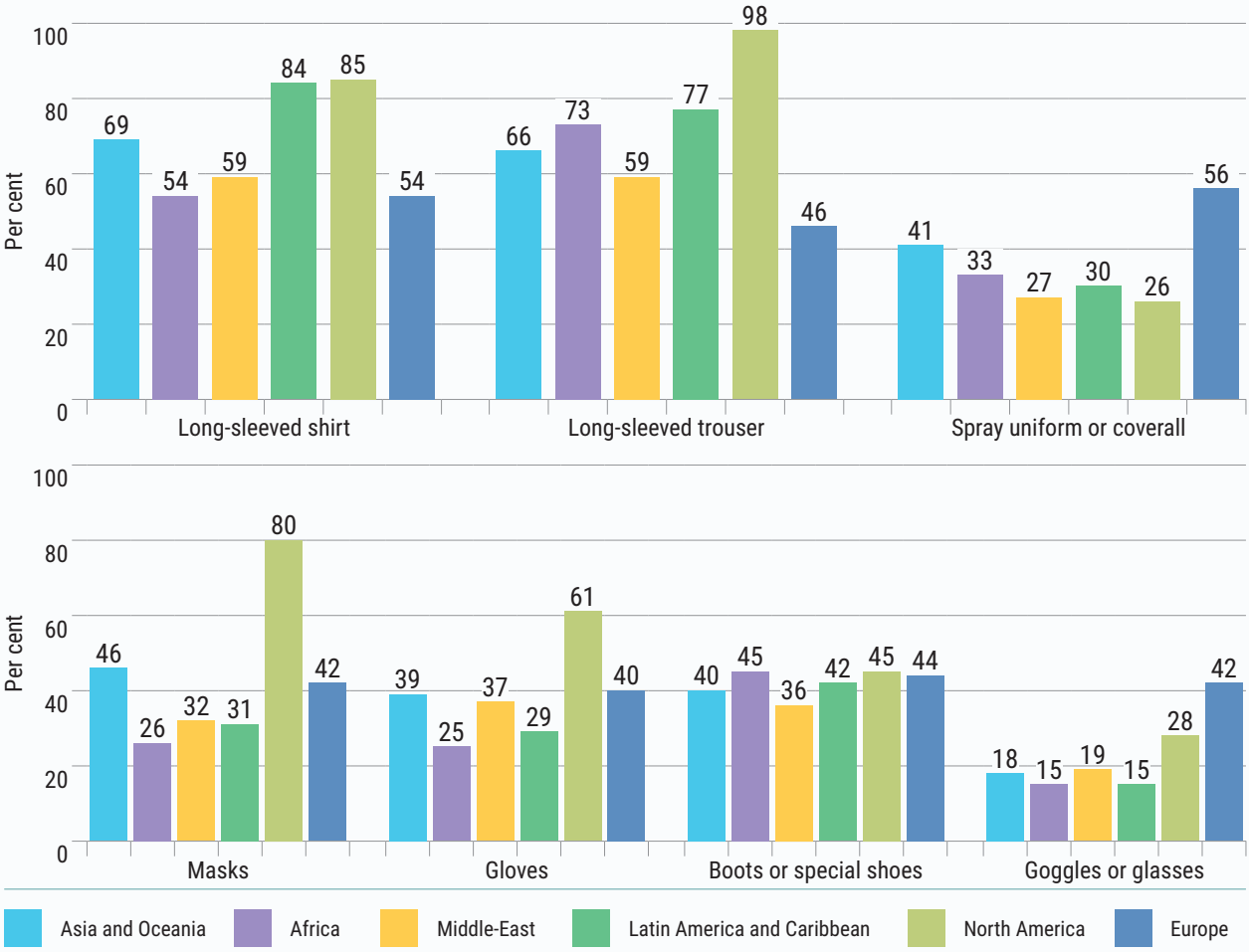
The use of personal protective equipment (PPE) is central to discussions of pesticide risk reduction for people who handle and apply pesticides or work in pesticide-treated fields or spaces. In many countries the authorization of moderate and high risk pesticides is conditional on the use of specific types of PPE. This equipment includes skin protection (aprons, coveralls, boots, gloves, hats/helmets), eye protection (face shields, goggles) and respiratory protective equipment (masks, respirators).

When studies on pesticide application under working conditions are reviewed, use of PPE either reduces exposure or does not seem to have a quantifiable effect; exposure reductions appear to be less than under controlled conditions. While not wearing PPE generally increases the incidence of poisoning, using it does not always reduce it (Kim *et al*. 2013; Garrigou *et al*. 2020). Calvert *et al*. (2008) noted that half of those poisoned by pesticides did wear.

Under controlled circumstances, proper use of adequate PPE significantly reduces occupational exposure to pesticides. Such exposure reduction factors have been incorporated into regulatory pesticide exposure models. For example, the European Food Safety Authority (EFSA 2014) considered that dermal pesticide exposure of pesticide applicators would be reduced by 90 per cent when chemical resistant gloves and coveralls or a single layer of work clothing are worn.

Based on a recent review of the scientific literature, Garrigou *et al*. (2020) found that both single-layer clothing and coveralls reduce occupational

Figure 4.4-7 Use of PPE items in different regions of the world.



Shown are the average percentages of study populations using a PPE item, from the global review by Sapbamrer and Thammachai (2020).

exposure to a pesticide. However, a considerable fraction still passes through the fabric, even of chemical-resistant coveralls. The amount varies depending on the type of fabric, the area of the body, and the kind of pesticide under study. Most available research on the effectiveness of such equipment where PPE wearing practices are uncontrolled have shown that protective coveralls are not as effective as expected (Garrigou et al. 2020) (Table 4.4-10).

A global review of factors affecting the use of PPE by Sapbamrer and Thammachai (2020) looked at 121 scientific articles published between 1999 and 2019. It was found that, on average, about 70 per cent of pesticide handlers (farmers and farm workers) wore long-sleeved shirts and long trousers, but less than 30 per cent (also) wore dedicated coveralls or spray uniforms. On average, less than half of pesticide handlers used specific PPE such as gloves, masks, boots and goggles. There was large variability among world regions, with PPE worn more regularly in high income than low income countries (Figure 4.4-7). However, even in OECD member countries, where conditions for wearing such equipment might seem most favourable, Garrigiou *et al*. (2020) found that rates of PPE use were much lower than recommended.

Various reasons have been given to explain why farmers, pesticide applicators and farm workers are not, or are only partially, using PPE. The right PPE may not be available or affordable in some regions, an argument often made concerning subsistence farmers and workers in lower income countries. Non-provision or non-requirement by employers has been found to limit PPE use by farm workers. Literacy, higher education, the perceived benefits of PPE and dedicated training may increase its use, although the results

of studies linking education and training to PPE use are mixed (Garrigou *et al*. 2020; Sapbamrer and Thammachai 2020). PPE use has also been linked to farm size, with more (appropriate) PPE being worn on larger farms, possibly due to greater financial means.

Non-use of PPE can also be explained by characteristics of the equipment itself. Many items are uncomfortable, restrict the user's movements and can be very hot to wear, especially in warm and humid conditions. Under certain circumstances this can result in heat stress, potentially leading to loss of coordination and dangerous acute health effects (Garrigou *et al*. 2020; Sapbamrer and Thammachai 2020).

Garrigou *et al*. (2020) noted a dilemma in regard to PPE-dependent risk prevention: the more an item of PPE protects the wearer from pesticides, the more likely it is to be uncomfortable or even impossible to wear. In addition, the more effective it is, the more likely it is to be costly. They conclude that "the possibility of having PPE that is comfortable, suitable to practical conditions, affordable, and protects from contamination by any and all handled products has yet to be demonstrated". It should be emphasized that this conclusion applies to high income OECD countries; it is likely to be even more true of PPE use in low income countries (Pesticide Action Network Asia Pacific 2010). This dilemma is especially relevant to the discussion of the use of Highly Hazardous Pesticides (HHPs) (Chapter 3.2.3), which require effective use of appropriate PPE.

Inadequate use of PPE in large parts of the world, together with the incomplete protection this equipment provides, means it should be considered a last line of defence after other measures have been taken (i.e., the "hierarchy of control" principle; see Chapter 6.2.5). The hierarchy of control is a fundamental principle of occupational health and safety around the world (Alli 2008; United States National Institute for Safety and Health 2015; European Agency for Safety and Health and Work 2018).

Residential exposure to pesticides

People who live close to agricultural fields may be exposed to pesticides through various pathways, the most important of which are spray drift, volatilization of pesticides beyond the treated area, take-home exposure (e.g., through pesticide residues on the clothing of workers or farmers), pesticide use in/around the residence, and dietary ingestion (Deziel *et al*. 2015).

Those who live further from fields are mainly exposed through residential pesticide use (e.g., disease vector control, nuisance pest control, domestic and garden uses) and by dietary ingestion (Chapter 4.4.6).

Considerable concern has been (and continues to be) expressed about the exposure of people living close to agricultural fields, especially in areas with high intensity of pesticide use or where pesticide application methods entail potentially high drift (e.g., aerial applications, orchard spraying) (Dansereau *et al*. 2006; Lee *et al*. 2011; Human Rights Watch 2018; Health Council of the Netherlands 2020).

Dereumeaux *et al*. (2020) reviewed the scientific literature that quantifies pesticide exposure of non-farmworker residents living close to agricultural fields, who are expected to be exposed mainly through spray drift and pesticide volatilization. Most studies had been conducted in North, Central and South America. The study results confirm that those living closer to pesticide-treated agricultural land tend to have higher levels of pesticide residues/metabolites in their households and/or biological samples, higher levels of oxidative stress markers, greater DNA damage, and decreased activity of cholinesterase than those who live farther away. Moreover, the amount of pesticides applied, the acreage treated, and the time of year compared to the spray season were positively correlated with levels of human exposure.

Dereumeaux *et al*. (2020) and the Health Council of the Netherlands (2020) have pointed out that while associations between proximity to agricultural fields and various adverse health outcomes have been observed, but that the studies are not always

conclusive. This is partly due to the difficulty of characterizing and quantifying residents' exposure to pesticides. Stronger epidemiological evidence exists, however, for certain health effects due to occupational exposure (Chapter 4.4.3).

Deziel *et al*. (2015), in a review of evidence in the published literature for the contribution of non-occupational pathways of pesticide exposure in women living in North American agricultural areas, calculated that, on average, pesticide concentrations in house dust 250 metres from treated fields were still 36 per cent of the levels found in houses about 20 metres from fields. Lee *et al*. (2011) found that in the United States 73 per cent of non-occupational exposure to pesticide drift occurred more than 400 metres from the application site. However, on the basis of current observational studies assessing residential exposure and human health effects, it does not seem possible to define a safe distance between a residence and field that could ensure protection of human health (Teysseire *et al*. 2020).

There is also increasing evidence that the "take-home"(or "para-occupational") exposure pathway may contribute considerably to residential exposure to pesticides. Pesticide residues can be transferred from the workplace to the household environment on agricultural workers' clothing, skin, vehicles and shoes. A recent review provided evidence that these workers' families are exposed to pesticides at higher levels than those of non-agricultural workers (López-Gálvez *et al*. 2019). Levels may depend on several factors, including seasonality, parental occupation, cohabitation with a farmworker, behaviour at work/home, age and gender.

Good work practices, including the availability of laundry facilities, storing work boots at work instead of at home, frequent washing of hands before leaving the workplace, and receipt of pesticide training, have all been associated with lower residential pesticide contamination, suggesting that community based interventions that disrupt the take-home pathway can be effective in reducing pesticide exposure (Fenske *et al*. 2013; López-Gálvez *et al*. 2019). However many of the ways to lower such transfer may not be available to subsistence farmers in low income countries.

While increasing evidence exists concerning residential exposure to pesticides, less is known about the adverse health effects that could result from such exposure. Teysseire *et al*. (2020) found that 71 per cent of the epidemiological studies included in their global assessment reported a significant association between at least one health outcome and residential exposure to pesticides. However, they did not evaluate the quality of these studies.

Recently the Health Council of the Netherlands (2020) reviewed the (mainly chronic) health risks posed to people living in the vicinity of agricultural land by the use of plant protection products. They conclude that the international epidemiological literature indicates the use of chemical agents for plant protection can be associated with impaired human health, although little is known about the exact level of risk involved or precisely which products are responsible.

The Health Council recognized that further research is not expected to provide clarity in the near future, nor can the approval procedure for these products ever fully eliminate risks. For these reasons it recommended application of the precautionary principle by redoubling efforts to reduce agricultural dependence on chemical plant protection products (Health Council of the Netherlands 2020).

On the basis of the above reviews it can be concluded that residential exposure to pesticides is likely to be widespread, especially in areas where there is intensive agriculture or residents live in close proximity to treated fields. Although risk mitigation measures such as drift reduction technologies, buffer zones and improved work hygiene have been recommended or required in many countries, this residential exposure remains extensive. The resulting health risks are less clear, but several epidemiological studies have found associations between residential exposure to pesticides and adverse health outcomes, indicating that health concerns about this exposure pathway are warranted.

4.4.6 Pesticide residues in food

Residue monitoring

Pesticide residues may remain in/on a final food product following pesticide applications to the crop or post-harvest treatments of the commodity, or because of contamination from environmental sources (e.g., legacy pesticides remaining at low levels in the soil). Such residues may pose a dietary risk to the consumer. Many governments therefore set maximum residue limits (MRLs) for pesticides that are permitted on food commodities. At the international level, the Codex Alimentarius Commission (the Codex) establishes globally harmonized MRLs to protect the health of consumers as well as to ensure fair practices in the food trade.

To assess whether MRLs are respected, pesticide residues should be monitored on a regular basis in raw agricultural commodities and in processed food items. Different types of pesticide residue monitoring are conducted:

- monitoring programmes that attempt to provide a representative assessment of the situation of pesticide residues in food products consumed in a country (these often follow random sampling methods);
- risk-based monitoring programmes focusing on pesticides or commodities originating from sources where relatively high residue levels have been found in the past or are expected (this is sometimes also referred to as "enforcement monitoring");
- research projects, which tend to be ad hoc residue sampling programmes conducted to answer specific scientific questions (such sampling can be random, stratified or otherwise directed).

Residue monitoring may be carried out in both domestic and imported food products.

Residue monitoring programmes exist in certain countries, but most of them are high or upper-middle income economies (Table 4.4-11). Regular monitoring of pesticide residues in low and lower-middle income countries is rare, and most information on pesticide residues in food is collected through ad hoc research studies. A recent global survey indicated that only 58 per cent of responding countries had systems

Figure 4.4-8 On average, 58 per cent of countries have systems in place for monitoring pesticide residues in food and feed, but large differences exist among world regions (total number of responding countries = 50). WHO and FAO (2019a).

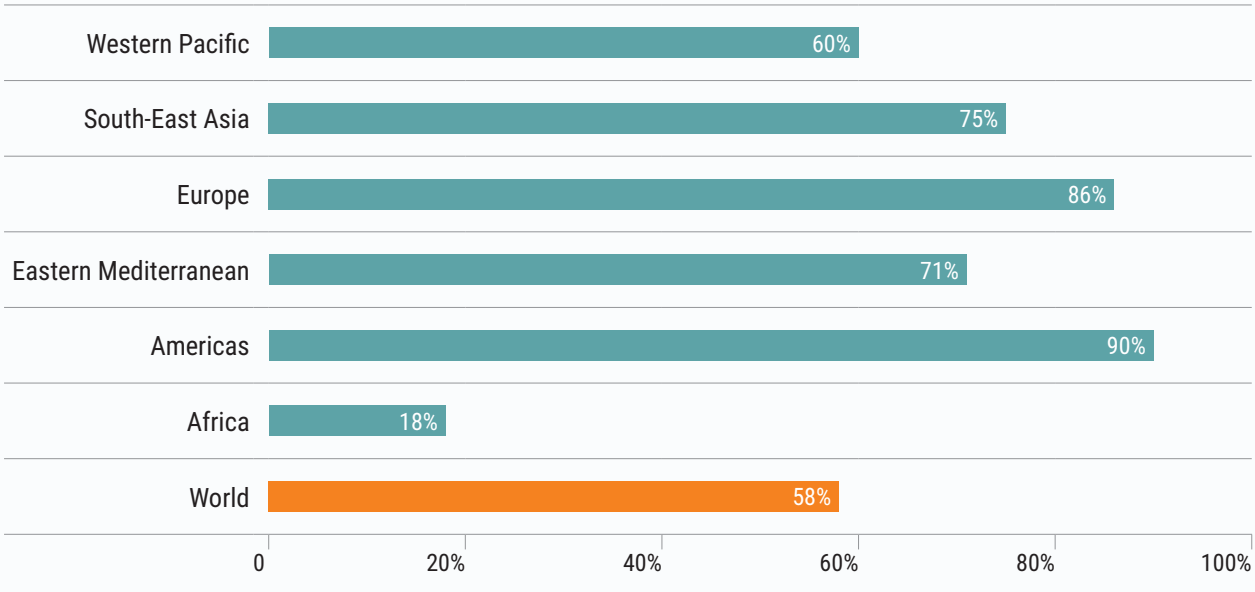


Table 4.4-11 Examples of national pesticide residue monitoring programmes.

Country	Programme and *executing agency*	Type of monitoring	Source
Australia	National Residue Survey (NRS), Department of Agriculture, Australia (DoA)	Risk based	DoA (2019)
Brazil	Programa de Análise de Resíduos de Agrotóxicos em Alimentos (PARA), Agencia Nacional de Vigilância Sanitária (ANVISA)		ANVISA (2019)
	Plano de Nacional de Controle de Resíduos e Contaminantes (PNCRC), Ministério da Agricultura, Pecuária e Abastecimento (MAPA)		PNCRC (2018; n.d.)
Canada	National Chemical Residue Monitoring Program (NCRMP), Canadian Food Inspection Agency (CFIA)	Random and risk based	CFIA (n.d.)
EU Member States, Norway and Iceland	EU-coordinated control programme (EUCP), European Food Safety Authority (EFSA)	Random	EFSA (2020b)
	National pesticide residue control programmes	Mainly risk based	
India	Monitoring of Pesticide Residues at National Level (MPRNL) scheme, Department of Agriculture, Cooperation and Farmers Welfare and Indian Council of Agriculture Research (ICAR)	Random	ICAR (n.d.)
New Zealand	Food Residues Survey Programme (FRSP) (plant products), National Chemical Residues Programme (NCRP) (animal products), New Zealand Food Safety (NZFS)	Random	NZFS (2020; n.d.)
United States	Pesticide Residue Monitor*ing* Program, United States Food and Drug Administration (US FDA)	Risk based sampling, focused sampling and total diet studies	US FDA (2019)
	Pesticide Data Program (PDP), United States Department of Agriculture (USDA) (The PDP has also operated an on-line database with residue monitoring data since 1994.)	Representative, random	USDA (2019)

in place to monitor pesticide residues in food and feed, with a prominent lack of such surveillance in Africa (Figure 4.4-8) (WHO and FAO 2019b).

The outcomes of pesticide residue monitoring programmes or studies tend to be interpreted in two ways: the fraction of sample commodities that exceeds applicable MRLs, and/or the dietary risk of consuming food and drinking water containing measured levels of pesticide residues.

Exceedance of an MRL is an indication that a pesticide has not been applied in accordance with Good Agricultural Practices. However, because MRLs are not toxicological reference values exceedance of the MRL does not necessarily mean the consumer is exposed to an unacceptable dietary risk (Box 4.4-3). To know whether consumers run a health risk, a dietary risk assessment needs to be conducted. Such an assessment is generally based on the consumption of all food sources that may contain residues of the pesticide in question.

Pesticide residue concentrations in food

There have been no global reviews of pesticide residues in food commodities and their potential risks to consumers. Regular monitoring programmes (as listed in Table 4.4-11) tend to publish detailed annual reports of their results, sometimes including dietary risk assessments. On the other hand, data from ad hoc residue monitoring studies, as conducted in many lower income countries, have never been systematically compiled and analysed (FAO/WHO Joint Meeting on Pesticide Residues, personal communication).

Box 4.4-3 Maximum Residue Limits (MRLs).

A **Maximum Residue Limit** (**MRL**) is the maximum concentration of a residue that is legally permitted or recognized as acceptable in or on a food or agricultural commodity or animal feedstuff (FAO/WHO 2014). MRLs can be defined nationally or globally, the latter by the Codex Alimentarius Commission (Chapter 3.2.3). Codex MRLs are primarily intended to facilitate international trade, while protecting the health of consumers.

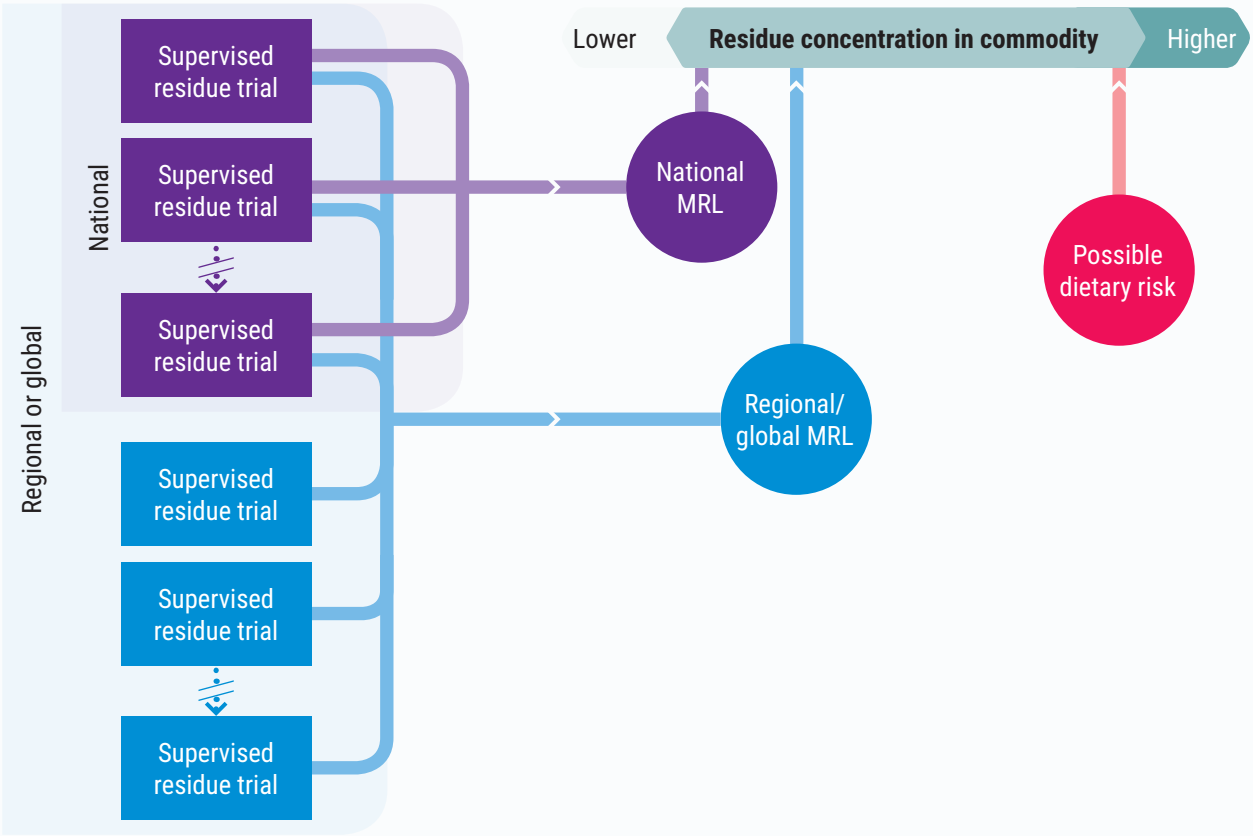
An MRL is derived on the basis of a toxicological assessment of the pesticide and its residue, and a review of residue data from supervised trials. They are always set for a specific pesticide active ingredient combined with a specific commodity (e.g. MRL of deltamethrin on tomatoes).

An MRL is estimated from residue levels measured in a series of supervised trials in which a specific pesticide is applied according to Good Agricultural Practice (GAP), i.e. the directions for the authorized use on the pesticide label. The use conditions leading to the highest residues (the critical GAP) are generally used to estimate the MRL for a given pesticide–commodity combination.

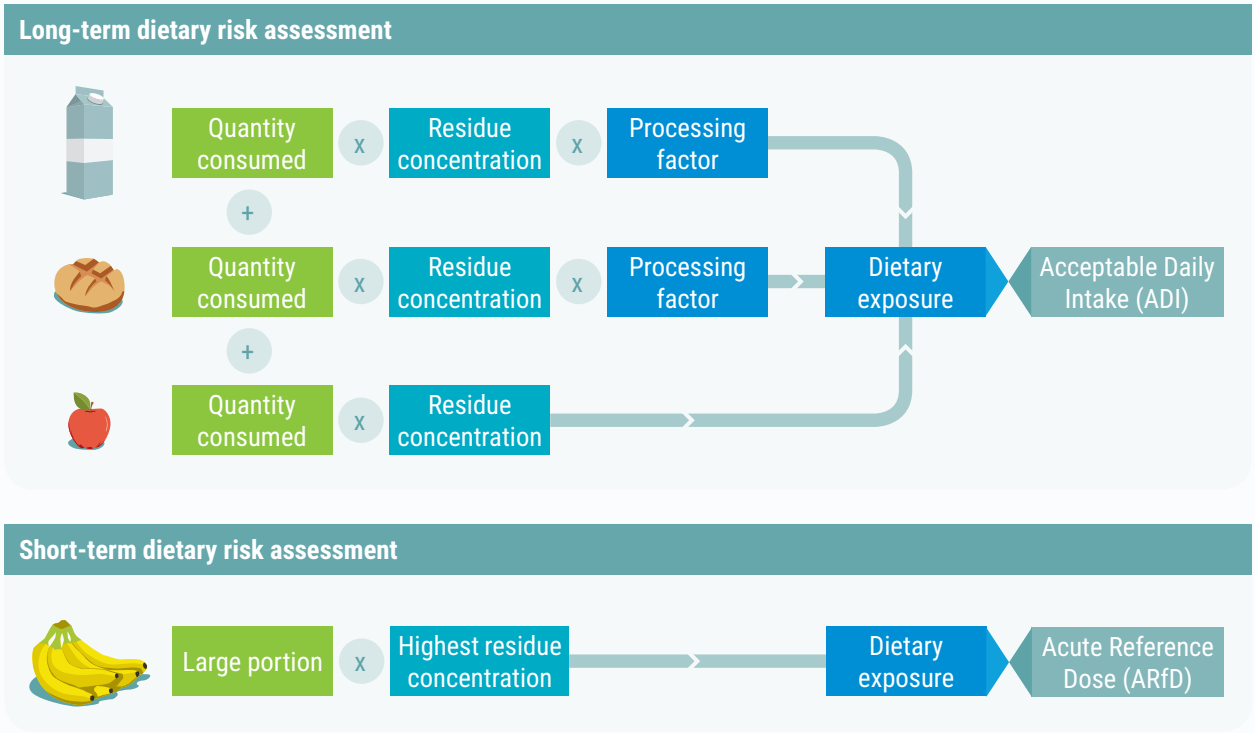
If a new MRL is established, a dietary risk assessment is also conducted (see below) to assess whether the residue levels used to estimate the MRL result in acceptable risks to the consumer. This is done for individual commodities and for all commodities combined for which the pesticide authorized. If the dietary risk is acceptable, the MRL can be adopted.

An MRL is not a toxicological reference value. It is a food standard reflecting the critical GAP in the country or region: it indicates the maximum residue that can be encountered on a commodity given the agronomic conditions in the country, and which does not pose an unacceptable dietary risk. MRLs can thus be different among countries because GAPs are different. For instance, a global (Codex) MRL may be higher than a national MRL because Codex includes pesticide residue studies from all over the world in their estimate, which may include situations where pesticide applications rates are higher due to climatic conditions; the critical GAP and thus the MRL may then be higher too. However, the higher MRL can only be adopted if the dietary risk is acceptable.

Exceedance of MRLs therefore signals that the GAP, i.e. recommended best pesticide application practices, have not been followed. It does not automatically mean that consumers run an increased dietary risk (although this may occur if exceedance of the MRL is large).



National or regional/global MRLs may be – but are not always – different, because they may be based on different residue trials with different GAPs. However, MRLs will always be below residue concentrations that can be expected to result in dietary risks.



A long-term (chronic) dietary risk assessment combines consumption of all food items on which residues of a pesticide may be present, and compares the total dietary exposure to the ADI of that pesticide. In a short-term (acute) dietary risk assessment, exposure following consumption of a large portion (single meal) of a food item is compared to the ARfD.

In a dietary risk assessment, exposure of consumers to residues expected or found on a range of commodities is calculated. Exposure estimates are based on average national or regional diets; i.e. quantities of food items consumed per day (for chronic dietary risk) or in a single large portion (for acute dietary risk). Only residue levels in the edible parts of the commodity are considered, and the effect of processing (e.g. cooking, baking, pasteurizing, brewing) on residues is also taken into account when relevant.

In the long-term (chronic) risk assessment, the estimated chronic dietary exposure is compared with the acceptable daily intake (ADI) value, which is the daily intake of a pesticide in food and/or drinking-water which, during an entire lifetime, appears to be without appreciable risk to the health of the consumer.

For the short-term (acute) risk assessment, the quantity of residues estimated to be ingested in a single day is compared to the Acute Reference Dose (ARfD), which is the amount of a pesticide in food and/or drinking-water, which can be ingested in a period of 24 hours or less without appreciable health risk to the consumer.

Both for the acute and the chronic assessment, risks are considered acceptable if the estimated dietary exposure is less than the ADI or ARfD.

Various models are used for dietary risk assessment, such as the WHO/FAO International Estimated Daily Intake (IEDI) and International Estimate of Short Term Intake (IESTI) models by the Codex Alimentarius, the Pesticide Residue Intake Model (PRIMo) in the European Union, and the Dietary Exposure Evaluation Model – Food Commodity Intake Database (DEEM-FCID)/Calendex by the USEPA.

Generally, dietary exposure is calculated separately for each pesticide, either on an individual commodity or by combining all commodities on which the pesticide is to be authorized or measured. However, in reality consumers may be exposed to residues of several different pesticides at the same time. The estimation methods for such cumulative (or combined) exposure, though still under development, are now increasingly used.

FAO and WHO 2014, Ambrus & Yang 2015, FAO 2016, Brancato ***et al***. 2018, USEPA undated, WHO undated, Yeung ***et al***. 2018.

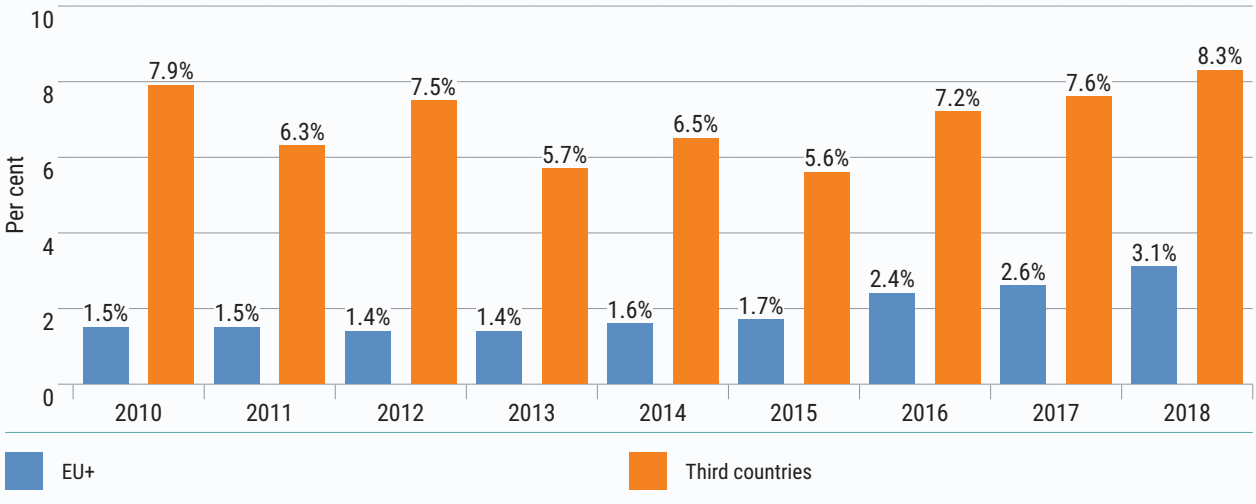
Residue monitoring programmes in Europe, North America, Australia, New Zealand and India generally show that exceedances of MRLs are relatively limited, typically ranging from <1 to 10 per cent of samples taken in a given year. In Table 4.4-12 only Brazil has a considerably higher exceedance rate for plant based food commodities. It should be noted that comparisons of residue monitoring programmes between countries is not straightforward, as countries may

Table 4.4-12 Recent levels of exceedance of pesticide maximum residue limits (MRLs) in countries with regular pesticide residue monitoring programmes in food commodities.

Country	Year	Food commodities	Number of samples (origin)	Free of pesticide residues [1]	MRL exceedance[1, 2]	Source
Australia	2018-2019	Animal based	9,952	--	0.2 per cent	DoA (2019)
		Plant based	5,241		0.8 per cent	
Brazil	2017-2018	Selected plant based	4,616 (domestic)	49 per cent	23 per cent	ANVISA (2019)
	2018	Animal based	12,495	--	0.4 per cent	PNCRC (2018)
Canada	2018-2019	Selected plant based	3,348	60 per cent	0.7 per cent	CFIA (2019)
EU Member States, Norway and Iceland	2018	Selected animal and plant based	57,286 (domestic)	58 per cent	3.1 per cent	EFSA (2020b)
			24,495 (imported)		8.3 per cent	
India	2017-2018	Animal and plant based	23,660	81.9 per cent	2.2 per cent	Ministry of Agriculture and Farmers Welfare (2019)
New Zealand	2017-2019	Fruits and vegetables	591	--	6.2 per cent	NZFS (2020)
United States	2017	Animal and plant based	1,799 (domestic)	52.5 per cent	4.8 per cent	US FDA (2019)
			4,270 (imported)	50.0 per cent	10.4 per cent	
	2018	Mainly plant based	6,981 (domestic)	47.8 per cent	6.0 per cent	USDA (2019)
			3,385 (imported)		9.0 per cent	

1 MRL exceedance is expressed as percentage of total number of commodity samples.
2 Generally including samples with pesticide residues for which no MRL or tolerance has been established for that crop.

Figure 4.4-9 Pesticide MRL exceedances of food commodities imported into the EU+ (EU Member States, Norway and Iceland) between 2010 and 2018 ranged between 5.5 and 8.5 per cent of samples (orange bars). Lower MRL exceedances are found for food originating from within the EU+, but these have been steadily increasing from 1.5 in 2010 to 3.1 per cent in 2018 (blue bars). Based on EFSA (2013-2019).



differ in regard to the balance between random or "targeted" selection of samples for monitoring in their programmes.

In both the EU and the United States, MRL exceedance rates for imported food commodities are about twice the rates for domestically produced food. This suggests that agricultural practices in the exporting countries are not always able to meet national MRLs, e.g., due to higher pesticide application rates and frequencies or because pre-harvest intervals are not adequately respected.

MRL exceedance rates in the countries listed in Table 4.4-12 generally do not fluctuate much over time. MRL violations in regard to domestically produced food in the highly regulated EU have been steadily increasing during the last five years while still remaining fairly low (Figure 4.4-9). It is not clear why this has been the case.

Since very few regular pesticide residue monitoring programmes exist in low and middle income countries, information on violations of MRLs in these countries is limited.

In a global review on bananas, Gomes *et al*. (2020) found only a few published studies reporting the presence of pesticides on this fruit. In the available studies, however, depending on the standard-setting body (Brazil, the Codex, the EU), between 32 and 79 per cent of samples did not meet MRLs (either the MRL was exceeded or the pesticide found in the bananas did not have an established MRL).

No recent global reviews of pesticide residues in other commodities or cropping systems were available. However, ad hoc residue monitoring in low and middle income countries seems to indicate that the degree of exceedance of MRLs, across many commodities, may be significantly higher than the 1-10 per cent found in the high income countries mentioned above. Examples include studies in Ghana (Osei-Fosu *et al*. 2014), Thailand (Sapbamrer and Hongsibsong 2014), Pakistan (Faheem *et al*. 2015), Burkina Faso (Lehmann *et al*. 2017) and Bolivia (Skovgaard *et al*. 2017). Many of these studies compare residue levels with Codex MRLs, which were established based on global critical GAPs. The relatively high exceedance levels observed suggest that farmers in low and middle income countries are often not able to meet the GAPs defined internationally by the Codex. Higher application rates and frequencies, inadequate adherence to pre-harvest intervals, and use of pesticides that do not have an MRL (including diversion of pesticides registered for use on non-food crops to use on foods) may all contribute to increased MRL exceedances. It is also possible that farmers who do not grow for export markets are not aware of (or are less likely to comply with) the GAPs and MRLs that may have been established for a pesticide.

Dietary risk assessment

Dietary risk assessments are conducted to evaluate exposure to pesticide residues through food intake and the resulting risks to human health (Box 4.4-4). They generally have two different purposes (EFSA 2018e):

- Pre-registration (prospective) assessments are conducted to assess the risks to consumers resulting from pesticide residues expected on food, e.g., if a new active ingredient (or a new use of an already authorized active ingredient on a different crop) needs to be authorized.

- Post-registration assessments are conducted to assess the risk of actual exposure of consumers resulting from pesticide residues measured in pesticide monitoring programmes.

The risks of pesticide residues in food are mostly evaluated on a compound by compound basis. If potential exposure of consumers is below the relevant health based reference value (ADI or ARfD, see Box 4.4-4), the use of that pesticide is considered acceptable. However, consumers are frequently exposed to more than one pesticide residue in food at the same time. Assessing the risks of multiple pesticides in food is referred to as "cumulative risk assessment" (Boobis *et al*. 2008) or more commonly as "combined risk assessment" (European Food Safety Authority Scientific Committee 2019). Cumulative dietary risk assessments are considered most relevant for groups of pesticides with similar modes of action, or cumulative assessment groups (CAGs) (Boobis

et al. 2008,). Unlike the dietary risk assessment of individual pesticides, there is currently no internationally agreed protocol for the evaluation of combined dietary risks. However, several methods have been developed in the last decade and are increasingly applied at the national or regional levels (Meek *et al*. 2011; Jensen *et al*. 2015; Chang *et al*. 2018; EFSA 2020c; EFSA 2020d).

No global review of pesticide dietary risk assessments is currently available. Recent national or regional dietary risk assessments covering important groups of pesticides and food commodities have been compiled in Table 4.4-13. Apart from assessments in Brazil and Tunisia, all the others are from high income countries. In virtually all cases (countries, consumer age groups, and food commodities) actual acute and

Table 4.4-13 Recent regional or national dietary risk assessments of pesticide residues in food: assessments are of the risks of exposure to single pesticides and of combined exposure to multiple pesticides.

Country	Residue assessment period	Food commodities
Single pesticide[1]		
Eight countries: Australia, Brazil, Canada, Czech Republic, France, Italy, Netherlands, United States	2008-2017	Both plant and animal based
For a limited number of pesticides, in certain countries, exposure of children exceeded 20 per cent of the ARfD (carbofuran, cypermethrin, cyfluthrin, fenpropathrin, prothioconazole).	Crépet *et al*. (2021)	
Tunisia (adults 19-65)	2009-2010	42 core food groups covering 97 per cent of the Tunisian diet
The authors conclude that there is low dietary exposure to pesticide residues of the Tunisian adult population.	Bouktif Zarrouk *et al*. (2020)	
New Zealand	2017-2019	Fruits and vegetables
	NZFS (2020)	
EU	2018	Selected plant and animal based commodities
182 a.i.'s (for chronic risk)	Deterministic (PRIMo)	Acute risk: For 143 pesticides there was no exposure concern. The remaining 33 pesticides exceeded the acute health based guidance value in 327 samples (1.4 per cent). EFSA considers it unlikely that this limited number of exceedances of the ARfD would pose concerns for consumer health.
Chronic risk: EFSA concluded that according to current scientific knowledge, chronic dietary exposure at the assessed levels for the food commodities analysed is unlikely to pose concerns for consumer health.	EFSA (2020b)	
Germany	2009-2014	Mainly plant based

chronic dietary risks of pesticides in food were considered very small. The only exception is dietary exposure of children in Israel, where the large majority of this population was exposed to residue levels above the acceptable daily intakes of up to 10 different pesticides.

Dietary risk assessments have been published in low and middle income countries, but only for a small number of specific pesticides or commodities. There are few, if any, national assessments which include a broader range of pesticides and/or commodities. It is therefore difficult to evaluate whether the higher degree of MRL exceedance observed in these countries (see above) leads more often to unacceptable dietary risks.

	Pesticides[2]	Method	Outcome (as reported by the study authors)	Reference
	38 a.i.'s assessed	Probabilistic model of acute dietary exposure of adults and children	None of the exposure values for the 38 pesticides exceeded the ARfD.	
	170 a.i.'s assessed, of which 21 were detected	Total diet study	The ADI was not exceeded for any pesticide except the bromide ion (methyl bromide, which has been banned).	
	All MRL non-compliant a.i.'s from residue monitoring	Not defined	None of the survey samples exceeding the relevant MRLs resulted in any food safety concerns to consumers of all ages.	
	176 a.i.'s (for acute risk)			
	700 a.i.'s	Probabilistic model (MCRA)	693 pesticides were unlikely to pose chronic or acute dietary risks.	

Country	Residue assessment period	Food commodities
Short-term dietary exposure to chlorpyrifos, dimethoate/ omethoate may present a public health concern.		
Dietary risks of copper, dimethylvinphos, halfenprox and tricyclazole remained inconclusive.	Sieke *et al*. (2018)	
Israel (children only)	2006-2010	Fruits, vegetables, tubers
90 per cent of the children had uptakes in excess of the ADI for between two and eleven compounds; 5.6 per cent of children had one exceedance and 4.8 per cent had none.	Freeman *et al*. (2016)	
Cumulative[2]		
EU	2014-2016	Both plant and animal based
EU	2014-2016	Both plant and animal based
Brazil	2005-2015	Plant based
Brazil	2005-2015	Plant based
The same conclusion was made for cumulative chronic exposure to TR and DT in the case of individuals 10 years or older (up to 1 and 6.7 per cent of the respective ADIs).	Oliveira Jardim *et al*. (2018b)	
Denmark	2013-2014	Plant based
United States	2011-2014	Fruits and vegetables
Denmark	2004-2011	Fruits, vegetables, cereals

1 Types of dietary risk assessment: Single pesticide = Risk estimation for all relevant food commodities combined, containing one specific pesticide; Cumulative = Risk estimation for all relevant food commodities combined, and combined exposure to multiple pesticides

	Pesticides[2]	Method	Outcome (as reported by the study authors)	Reference
	26 a.i.'s most often found in the food items	Deterministic (similar to WHO Global Environment Monitoring System [GEMS])	Surveyed children had higher potential exposures than the general population for >33 per cent of the compounds.	
	Pesticides that have certain acute effects on the nervous system: >119 a.i.'s belonging to the same CAG	Two probabilistic models	For all populations studied (toddlers, children, adults) it is concluded with varying degrees of certainty that cumulative exposure to pesticides that have the studied acute effects on the nervous system does not exceed the threshold for regulatory consideration.	EFSA (2020c)
	Pesticides that have certain chronic effects on the thyroid: 133 a.i.'s belonging to the same CAG	Two probabilistic models	For all populations studied (toddlers, children, adults) it is concluded with varying degrees of certainty that cumulative exposure to pesticides that have the studied chronic effects on the thyroid does not exceed the threshold for regulatory consideration.	EFSA (2020d)
	Organophosphorus (OP), carbamate(CA) and pyrethroid (PY) insecticides	Relative potency factors, probabilistic model (MCRA)	The cumulative acute exposure did not exceed the ARfD for OP, CA and PY insecticides at the 99.9 percentile of the intake distribution. It does not therefore represent a health concern for the population under consideration (10 years or older).	Oliveira Jardim *et al*. (2018a)
	Triazole (TR) and dithiocarbamate (DT) fungicides	Relative potency factors, probabilistic model (MCRA)	The cumulative acute exposure of TR accounted for up to 0.5 per cent of the ARfD at the 99.9 percentile of the intake distribution and therefore did not represent a health concern for the relevant population (women of child-bearing-age).	
	198 a.i.'s	Hazard Index (HI)	HI = 0.44 for children and 0.16 for adults, indicating that adverse health effects from chronic pesticide exposure through food are very unlikely.	Larsson *et al*. (2018)
	Seven neonicotinoid insecticides	Relative potency factor (RPF) approach	The estimated average daily intakes were several orders of magnitude lower than the current chronic reference dose (cRfD). Chronic dietary risks of neonicotinoids through fruit and vegetable consumption are therefore unlikely.	Chang *et al*. (2018)
	~ 330 a.i.'s and metabolites	HI	HI = 0.44 for children and 0.18 for adults, indicating that food consumption did not pose chronic dietary risks for adults or children, including for high consumption of fruits.	Jensen *et al*. (2015)

2 a.i. = active ingredient; CAG = cumulative assessment group (i.e. pesticides with a common toxicological mode of action); ADI = acceptable daily intake; ARfD = acute reference dose; cRFD = chronic reference dose; HI = Hazard Index; MCRA = Monte Carlo simulation for risk assessment

4.4.7 Pesticides and antimicrobial resistance

Introduction

Antimicrobial resistance (AMR) refers to microorganisms – bacteria, fungi, viruses and protozoans – that have acquired resistance to antimicrobial substances (FAO 2015; WHO 2015; WHO 2020e). When microbes become resistant to medicines, the options for treating the diseases they cause are reduced. This resistance to antimicrobial medicines is occurring in all parts of the world for a broad range of microorganisms. The direct consequences of infections with resistant microorganisms can be severe, including longer illnesses, increased mortality, loss of protection for patients undergoing operations and other medical procedures, and increased costs (WHO 2015; WHO 2020e).

AMR in human and animal pathogens is primarily a result of the use, misuse and overuse of antibiotic and antimicrobial drugs to treat infections in humans as well as in animals (e.g., livestock, aquaculture). Although the scale of the problem is driven by human activity, AMR is ancient, predating human use of antimicrobials (D'Costa *et al*. 2011). Many antibacterial drugs are natural products produced by microorganisms and resistance has evolved in environmental microbial populations over evolutionary time. Critically, AMR in human and animal pathogens is not only conferred by mutation but also by acquisition of mobile resistance genes through a process called horizontal gene transfer, whereby resistance genes can be mobilized from harmless soil bacteria to unrelated clinical pathogens. The environment is also contaminated with antimicrobials and antimicrobial-resistant microbes through human and animal faecal wastes, which can accelerate the development and spread of resistance.

Emerging evidence suggests that metals, non-antimicrobial pharmaceuticals and even plant protection products such as herbicides may have previously unknown antimicrobial properties that can contribute to development of AMR (Kurenbach *et al*. 2015; Maier *et al*. 2018). Such contamination can occur from, for example, human and animal wastes (i.e., faeces, manure; see Chapter 9 on the risks of organic fertilizers), pharmaceutical manufacturing waste, and use of antimicrobial pesticides and other pesticides on crops (Wellcome 2018).

Slowing down antimicrobial resistance development to ensure the continuity of successful treatment and prevention of infectious diseases has become a worldwide priority, with global action plans developed in the human health as well as the agriculture and food sectors (WHO 2015; FAO 2015; EU 2020b; WHO, FAO and World Organization for Animal Health 2020). The United Nations Environment Assembly (UNEA) has also recognized that antimicrobial resistance is an increasing threat to global health, food security and sustainable development, and has underlined the need to further understand the role of environmental pollution in the development of antimicrobial resistance (UNEP 2018).

Use of antimicrobial pesticides

Antimicrobials in agriculture are used in terrestrial and aquatic animal and plant production for both treatment and non-therapeutic purposes such as animal growth promotion. The large majority of agricultural uses are as veterinary medicines in livestock production and aquaculture. These uses are not discussed here, as most of them do not fall under the commonly used definition of a pesticide (Chapter 2.2).[8]

Antimicrobials used as pesticides in crops can be broadly categorized as antibiotics (used to control bacterial plant diseases) and fungicides (used to control fungal plant diseases). In some cases these antimicrobials are the same, or closely related to, antimicrobials used in human medicine (Table 4.4-14).

The yearly amount of *antibiotics* used on crops has been considered relatively low in comparison to the quantities used in livestock, with estimates ranging from 0.2 to 0.4 percent of total agricultural

8 In aquaculture many antimicrobials and antibiotics are administered orally (e.g., in food pellets) and would be regarded as veterinary medicines. However, in both treatments of prawn and shrimp and the disinfection of ponds the antimicrobial is applied to the environment of the animals, in which case it could be considered a pesticide.

Table 4.4-14 Antimicrobials used as both pesticides and human medicines. Berger *et al.* (2017); Wellcome (2018); WHO (2019); FAO and WHO (2019).

Antimicrobial class	Antimicrobial pesticide	Use as human medicine	WHO CIA List classification (only for antibiotics)
Antibiotics			
Aminoglycosides		Yes	
	streptomycin	Yes	Critically Important
	gentamicin	Yes	High priority, critically Important
	kasugamycin	No (and no known cross-resistance with amino-glycosides used in human medicine)	
Tetracyclines		Yes	
	oxytetracycline	Yes	Highly Important
Quinolones and fluoroquinolones		Yes	
	oxolinic acid	Yes	Highest priority, critically Important
Antifungals			
Azoles		Yes	Not applicable
	e.g. difenoconazole, epoxiconazole, propiconazole, tebuconazole	No; e.g. itraconazole, voriconazole, posaconazole – but observed cross-resistance	

WHO classification of Critically Important Antimicrobials (WHO 2019):
Critically important: Antimicrobial classes which are C1 (this class is the sole or one of limited available therapies to treat serious bacterial infections in people) and C2 (this class is used to treat infections in people caused by either bacteria that may be transmitted to humans from non-human sources, or bacteria that may acquire resistance genes from non-human sources)
Highly important: Antimicrobial classes which meet the criteria for either classes C1 or C2, above.
Highest priority: Antimicrobial agents for which risk management strategies are needed most urgently.

antibiotic consumption (FAO 2015). Historically, the largest use of antibiotics on crops has been to control fire blight of apple and pear, but they are also used to control bacterial diseases in vegetables, fruits and flowers (Stockwell and Duffy 2012; Wellcome 2018). A recent review (Taylor and Reeder 2020) suggests that antibiotics are being recommended far more frequently and on a greater variety of crops than previously thought. The authors found that antibiotics are used in low and middle income countries in all regions of the world except Africa. Rice appears to be the main crop on which antibiotics are used. Streptomycin was the most frequently recommended antibiotic, followed by kasugamycin and tetracycline.

According to Taylor and Reeder (2020), although the quantities of antibiotics used for crop protection remain relatively low compared with medical and veterinary uses, application concentrations of antibiotics used to treat plant diseases are several orders of magnitude higher than residue concentrations of veterinary and human antibiotics in manures and sludges applied to soils, so that the impact on selection for AMR in the environment may be significant. The study further notes that mixtures of antibiotics with other pesticides are common, and that they have been found to promote cross-resistance or co-selection for antibiotic resistance.

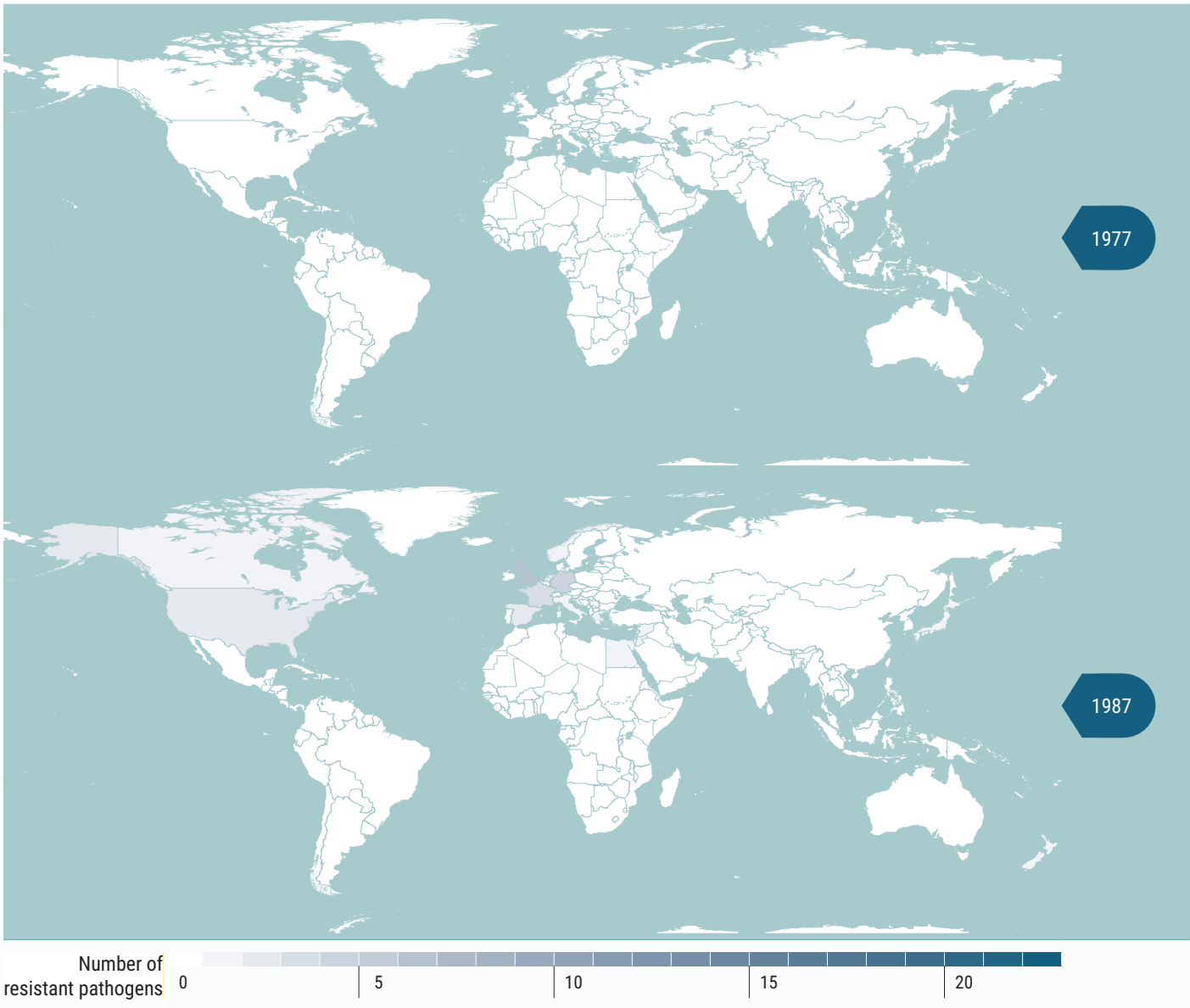
The use of *fungicides* is much higher than that of antibiotics (Chapter 2.3), although many fungicides are not known to be important for AMR development in human health. Of particular relevance are the azole fungicides, which are

among the most used fungicides in the world and are used in both agriculture and human medicine (Berger *et al*. 2017).

In contrast to antibiotic pesticides, copper based compounds are very commonly used on a wide variety of crops to manage bacterial and fungal plant diseases. While copper is not used in human medicine, it has been shown to co-select for resistance in bacteria and so may contribute to resistance development of antimicrobials in human medicine (Wellcome 2018).

Chemical disinfectants are included under the definitions of pesticide or biocide in certain countries and may be regulated accordingly. They are of critical importance for food safety to control microbial cross-contamination and ensure general hygiene at many stages of the food value chain. Chemical disinfectants are also used for decontamination and in healthcare facilities, as well as for disinfection of drinking water. Very large volumes of chemical disinfectants are used globally in both the food and the health sectors. The use of certain biocides is being questioned due to the possibility that

Figure 4.4-10 **The number of fungal species with reported antifungal resistance has been increasing over time, both in agriculture and in human health. Increasing colour intensity reflects a higher number of reports. The plant maps depict records of resistance of crop pathogens to azole fungicides (blue scale).**



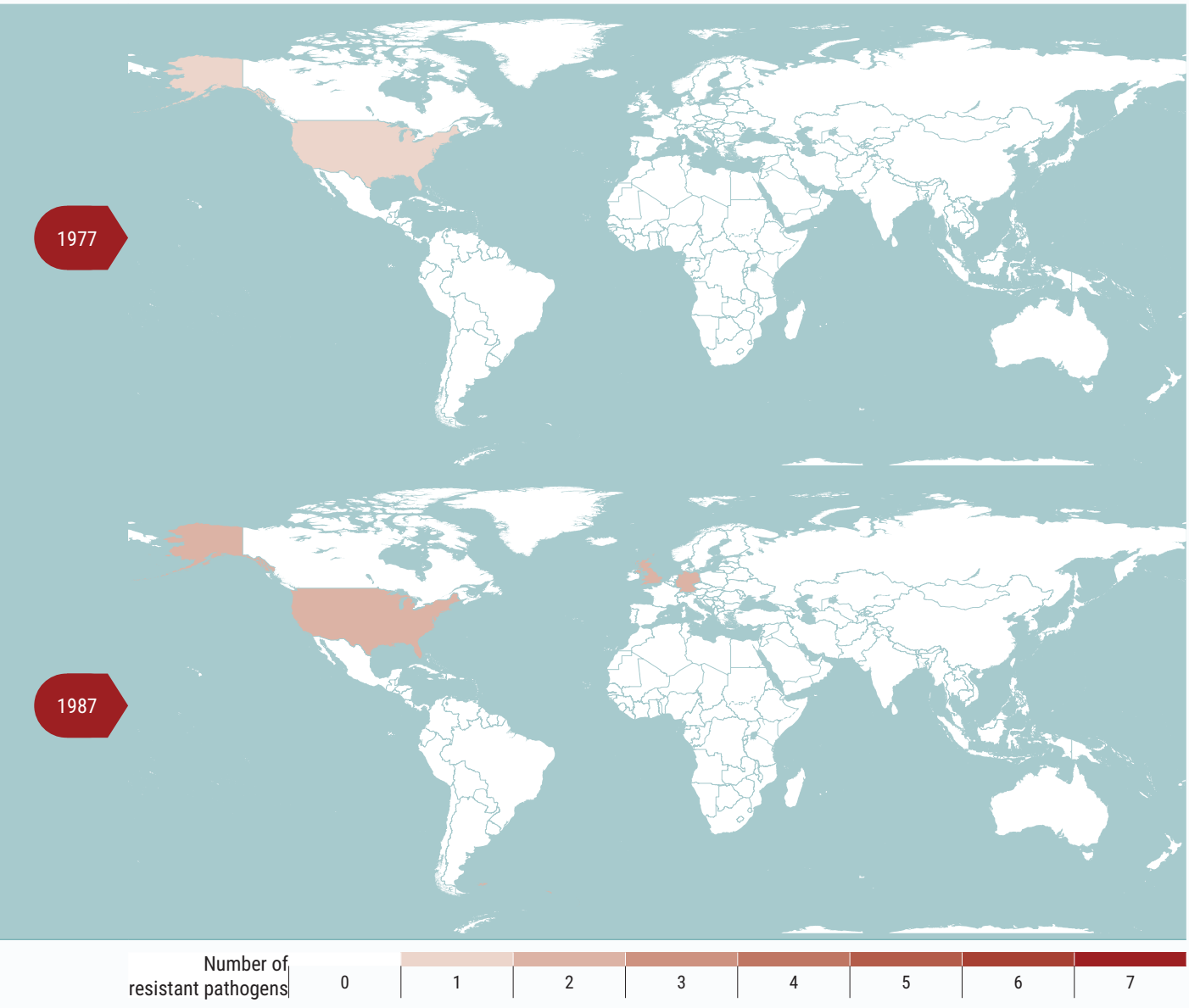
exposure could select for resistance to different antimicrobials including antibiotic drugs (FAO and WHO 2019).

Impact of pesticide-induced antimicrobial resistance

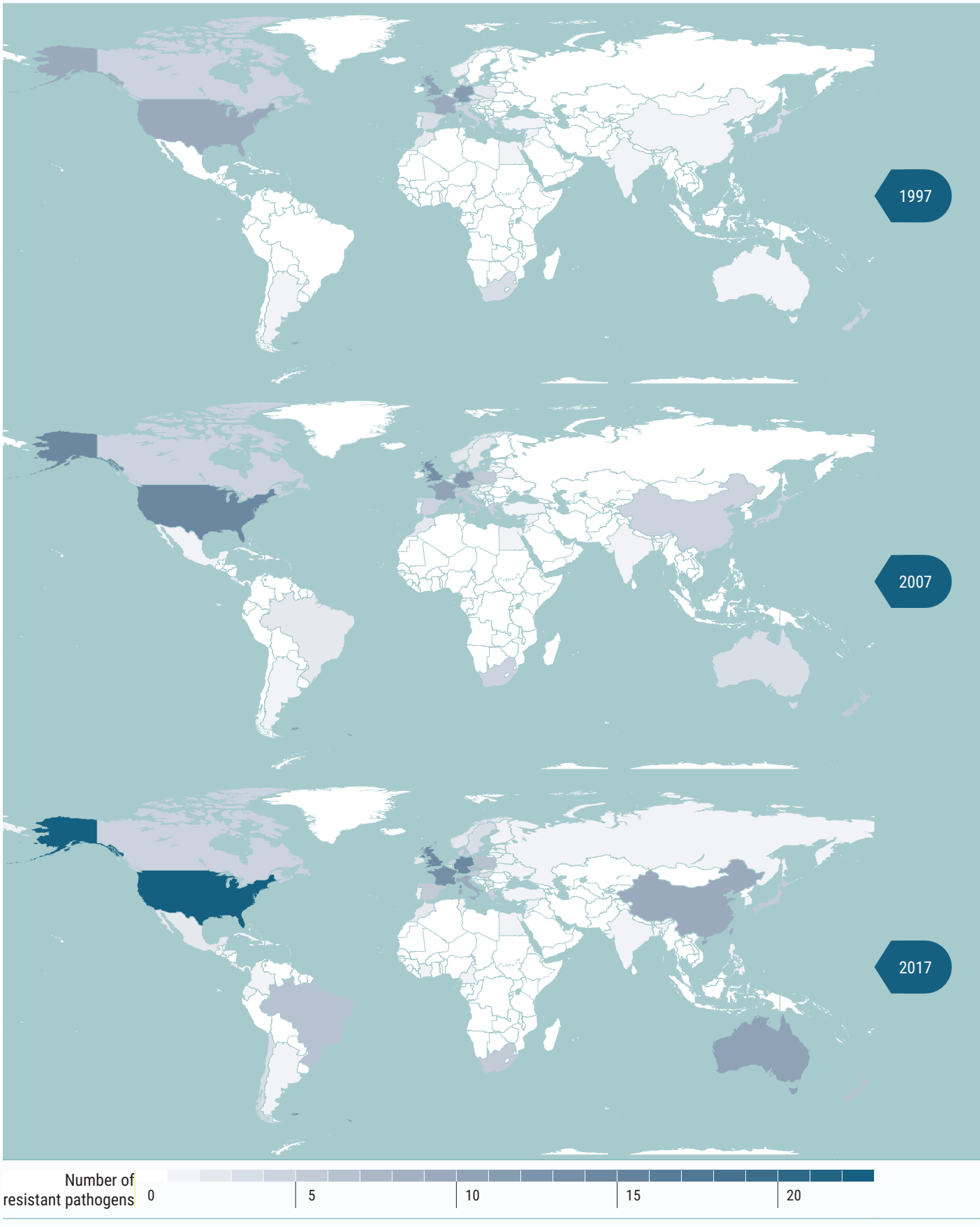
A recent expert meeting (FAO and WHO 2019) concluded that antimicrobial pesticides may contaminate soils following crop applications, which may lead to augmentation of antimicrobial-resistant bacteria and genes in the environment. However, the extent to which the treatment of crops with antimicrobial agents promotes AMR in bacteria found on edible portions of fresh plant produce is uncertain (FAO and WHO 2019).

While further research is needed to determine the effects of antimicrobial based pesticides on human health, there are specific concerns where antimicrobial pesticides are the same as, or closely related to, antimicrobials used in human medicine (Wellcome 2018; WHO 2019). This is the case for the antibiotics belonging to the classes of aminoglycosides, quinolones

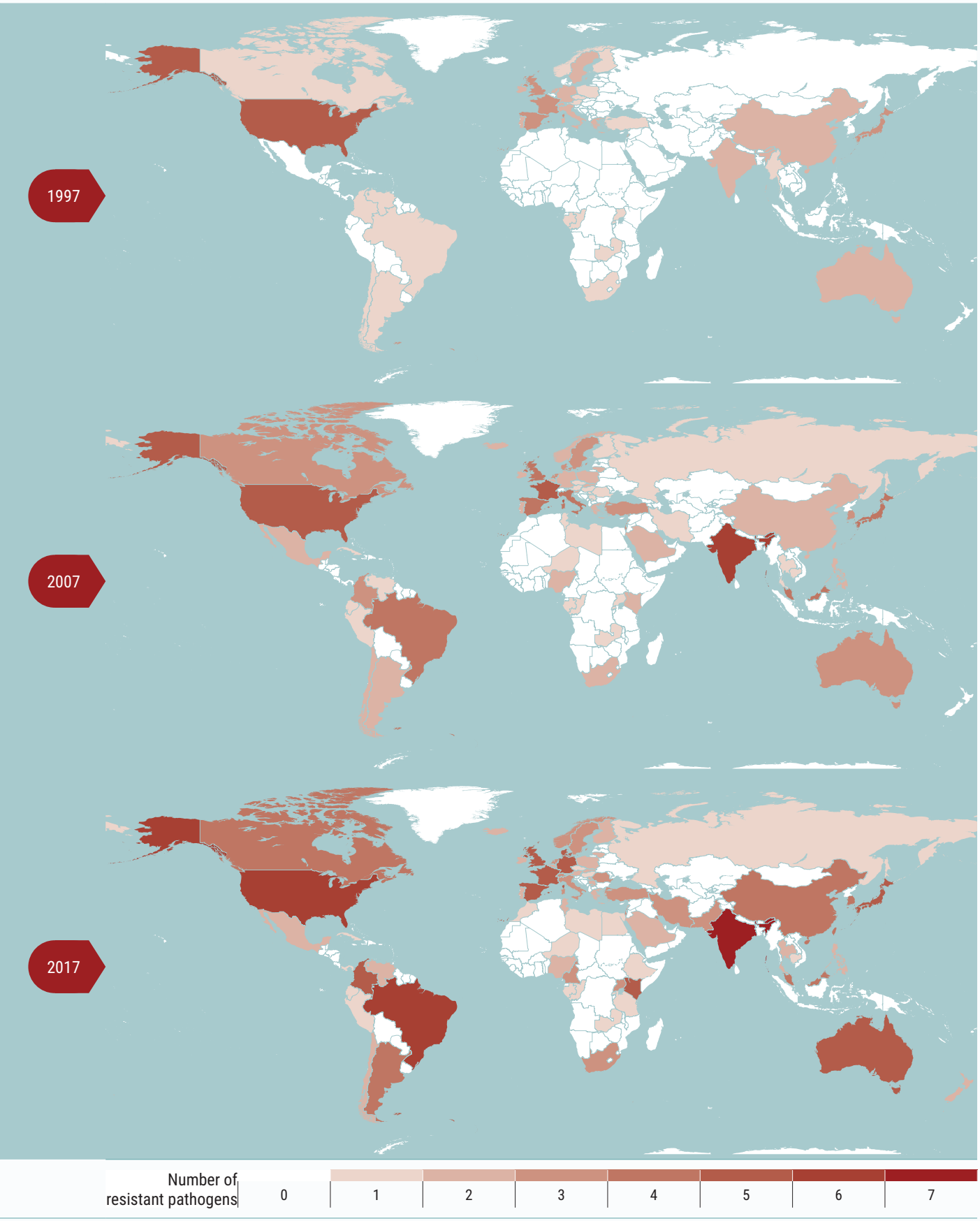
The human maps depict records of resistance of the human pathogens *Aspergillus fumigatus, Candida albicans, Candida auris, Candida glabrata, Cryptococcus gatti* and *Cryptococcus neoformans* to azole medicines (red scale). From Fisher *et al.* (2018).



The plant maps depict records of resistance of crop pathogens to azole fungicides (blue scale).



The human maps depict records of resistance of the human pathogens to azole medicines (red scale).



and tetracyclines, representatives of which are used in both agriculture and human medicine (Table 4.4-14). Some of them, such are oxolinic acid and gentamicin, are considered of critical importance to human medicine by WHO (2019) and their use in agriculture therefore urgently requires risk management measures.

Of further concern is the possibility of selection of antibiotic-resistant bacteria and genes through the processes of co-resistance, cross-resistance and co-regulation with certain metal ions. Copper resistance is widespread in plant pathogenic bacteria isolated from many continents, and evidence indicates that contamination of soil with copper ions also promotes AMR in soil bacteria (FAO and WHO 2019).

Less well publicized than the antibiotic resistance of bacteria is the rapid emergence of multi-drug resistant pathogenic fungi. Human fungal diseases are currently surging, and the global mortality numbers for fungal diseases have been reported to exceed those for malaria or breast cancer (Fisher *et al*. 2018). Recently there has been growing interest in the fungus *Aspergillus fumigatus*, airborne spores of which can enter the human respiratory system by inhalation and cause severe and possibly fatal invasive mould infections, especially in people who are immunocompromised. The main treatment for these infections currently is with antifungal medicines from the azole class. However, these medications are ineffective against resistant *Aspergillus* strains, leading to higher human mortality (Wellcome 2018).

The number of crop pathogens with reported resistance to azole fungicides has been increasing steadily during the last few decades. A similar expansion of resistance can be seen in human fungal diseases against azole based antifungal medicines (Figure 4.4-10). Infections with *Aspergillus fumigatus* that were resistant to all triazole antifungals were detected first in Europe and are now widespread across the world.

There is increasing evidence that use of azole fungicides in some specific agricultural sectors may be at least partly responsible for resistance selection in *Aspergillus fumigatus* and for subsequent medical treatment failure. Of particular importance is the fact that many patients with resistant infections did not have previous exposure to medical triazole antifungals, suggesting they became infected with a strain already carrying the mutation. Such strains would have become resistant in the environment following exposure to agricultural or other non-medical azole fungicides (Berger *et al*. 2017; Fisher *et al*. 2018; Wellcome 2018).

Bacteria with increased tolerance to chemical disinfectants have been recovered from food production environments. There is theoretical and experimental evidence that certain microbiocidal agents may co-select for AMR, including antibiotic resistance. Examples include the use of chlorhexidine resulting in colistin resistance, or triclosan inducing isoniazid resistance. However, such evidence is based on laboratory studies and there is an absence of empirical data indicating that use of biocides drives this co-selection under the conditions present in the food production or processing environments (FAO and WHO 2019).

At present, insufficient evidence is available to link biocide use in food production to the development of AMR. However, the identified association between biocide tolerance and resistance and one or more classes of antimicrobials underscores the need for increased awareness and judicious use of these products (FAO and WHO 2019).

Risk mitigation

The use of antimicrobial pesticides and disinfectants in agriculture and the food industry has either been identified as a driver of antimicrobial resistance, or serious concerns exist that this could be the case. Some countries have therefore taken measures to mitigate such risks and to ensure that antimicrobials can continue to be used effectively in human medicine.

Due to the risk of AMR to human antibiotics, use of antibiotics as bactericides in crop protection has been banned or restricted in certain parts of the world. For example, compounds such a streptomycin, oxytetracycline, gentamicin are no longer registered for use in agriculture in the EU. Some antibiotics are authorized for

use in crop protection in Mexico, New Zealand and the United States, but are fairly strictly regulated. Indeed, limiting the use of antibiotics in agriculture, particularly that of critically important antimicrobials and for prophylactic uses supporting unsustainable farming practices, has been proposed as a direct route for controlling agricultural antibiotic release into the environment, and likely also antibiotic resistance (Pruden *et al*. 2013; Wellington *et al*. 2013). However, in most countries the use of antimicrobials in agriculture is not regulated or monitored and their use may be more frequent than previously thought (Taylor and Reeder 2020).

Copper based fungicides and bactericides are widely used globally. They are often considered relatively low-risk pesticides which can even be used in organic agriculture. More recently copper compounds have come under increased scrutiny, mainly due to environmental persistence and toxicity to soil organisms and processes. Therefore, they have been listed in the EU as a candidate for substitution.

There are strategies for avoiding or limiting the use of medically important antimicrobials as pesticides, including modelling to predict high-risk periods for crop disease, practices that reduce the spread of crop pathogens, integrated pest management (IPM), and alternative treatments that reduce disease. However, these strategies are not always used globally and growers need support to use them, including access to these treatments and training (Wellcome 2018; FAO and WHO 2019).

4.4.8 Knowledge gaps on human health effects of pesticides

Much knowledge has been amassed in the last few decades about the human health effects of pesticides. The European Academies of Science recently reviewed the methods and procedures used in the EU to assess potential harmful effects on human health of the use of plant protection products (SAPEA 2018). They indicated that although the system is precautionary, there is scope for further improvement in the scientific data that underpin pesticide risk assessments and the methods by which such data are analysed.

SAPEA suggested that improvements to the range and quality of data informing risk assessment could come from advances in: toxicology, where newly emerging methods should enable the collection of data more directly relevant to human toxicity; epidemiology, where the development of new biomarkers for pesticide exposure and new study designs could improve surveillance for unanticipated adverse effects of pesticides; and exposure sciences, where there is scope for refining information on the distribution and determinants of personal exposures to pesticides (SAPEA 2018).

Other bodies and scientists have also recently identified gaps in knowledge and priorities for research on the human health effects of pesticides, toxicological testing and risk assessment approaches (Milner and Boyd 2017; European Food Safety Authority Scientific Committee 2019; Liu *et al*. 2019; Robinson *et al*. 2020; United States Environmental Protection Agency [US EPA] 2020c; US EPA 2020d). The Government of Canada has prepared a fact sheet for the consideration of sex and gender in pesticide risk assessments (Government of Canada 2020a; Government of Canada 2020b)

On the basis of these reviews, and of this report, the following gaps in scientific knowledge can be identified which are important for future reduction in human health risks and impacts of pesticide use:

toxicological evaluations of more complex human health outcomes such as immunotoxicity, childhood leukemias, developmental neurotoxicity, chronic neurological diseases like Parkinson's disease, neuropsychological effects and mental illnesses, as well as endocrine disorders such as some hormonal cancers, endometriosis, metabolic syndrome, type-2 diabetes, and reproductive senescence;

- toxicological assessments of co-formulants and of formulated products;
- assessments of detailed mechanisms by which chemicals interact with the body, and the adverse outcome pathways (AOPs) through which they might cause harm;

- new toxicological methods that reduce use of animals in testing, including the application of in vitro toxicogenomics;
- the risk of combined effects from exposure to multiple active substances, either simultaneously or in sequence;
- further development and standardization of methods for cumulative dietary risk assessments;
- ascertain whether prevailing human health risk assessments are sufficiently protective for potentially sensitive populations such as immuno-depressed persons, female farmers and agricultural workers, pregnant and nursing women, and children.
- levels and determinants of exposure, especially of pesticide applicators using handheld or backpack sprayers, and of workers who enter crops that have been treated with pesticides;
- better characterization and quantification of exposure to pesticides in epidemiological studies;
- "real life" behaviours of pesticide operators in regard to the use of equipment and application techniques, and the extent to which such approaches can be used to manage or reduce exposures;
- standardized risk assessment procedures for nanopesticides and for biopesticides;
- the impact of the use of pesticides on the development of antimicrobial resistance;
- consideration of the rapidly growing body of epidemiological evidence, especially when reviewing approvals for products that are already on the market;
- post-marketing surveillance of approved pesticides to (e.g., through establishing or strengthening poison centres) verify that they do not cause unanticipated human health problems.

References

Abdou, M. (2015). An assessment of the impact of chemical insecticides used to control desert locust (*Schistocerca gregaria*) (Orthoptera, Acrididae) on Lacertid lizard *Acanthodactylus* spp (Sauria, *Lacertidae*) in Niger. *American Chemical Science Journal* 7(4), 193-200. http://doi.org/10.9734/ACSj/2015/16556.

Adhikari, K. and Hartemink, A.E. (2016). Linking soils to ecosystem services – A global review. *Geoderma* 262, 101-111. https://doi.org/10.1016/j.geoderma.2015.08.009.

Agencia Nacional de Vigilância Sanitária (ANVISA) (Brazil) (2019). Relatório das amostras analisadas no período de 2017 a 2018. Programa de Análise de Resíduos de Agrotóxicos em Alimentos (PARA). http://portal.anvisa.gov.br/programa-de-analise-de-registro-de-agrotoxicos-para. Accessed 26 December 2020.

Agostini, M.G., Roesler, I., Bonetto, C., Ronco, A.E. and Bilenca, D. (2020). Pesticides in the real world: The consequences of GMO-based intensive agriculture on native amphibians. *Biological Conservation* 241, 108355. https://doi.org/10.1016/j.biocon.2019.108355.

Ahrensberg. H., Madsen, L.B., Pearson, M., Weerasinghe, M., Eddleston, M., Jayamanne, S. *et al*. (2019). Estimating the government health-care costs of treating pesticide poisoned and pesticide self-poisoned patients in Sri Lanka. *Global Health Action* 12, 1692616. https://doi.org/10.1080/16549716.2019.1692616.

Albuquerque, A.F., Ribeiro, J.S., Kummrow, F., Nogueira, A.J.A., Montagner, C.C. and Umbuzeiro, G.A. (2016). Pesticides in Brazilian freshwaters: A critical review. *Environmental Science: Processes and Impacts* 18(7), 779-787. https://doi.org/10.1039/c6em00268d.

Ali, U., Syed, J.H., Malik, R.N., Katsoyiannis, A., Li, J., Zhang, G. and Jones, K.C. (2014). Organochlorine pesticides (OCPs) in South Asian region: A review. *Science of The Total Environment* 476-477, 705–717. https://doi.org/10.1016/j.scitotenv.2013.12.107.

Alix, A., Bakker, F., Baret, K., Brühl, C.A., Coulson, M., Hoy, S. *et al.* (2012). *ESCORT 3: Linking Non-target Arthropod Testing and Risk Assessment with Protection Goals*. Pensacola, FL: Society for Environmental Toxicology and Chemistry (SETAC). https://www.researchgate.net/publication/262853453_ESCORT_3_Linking_Non-Target_Arthropod_Testing_and_Risk_Assessment_with_Protection_Goals/link/0deec53c7f2ed12a18000000/download.

Alli, B.O. (2008). *Fundamental Principles of Occupational Health and Safety. Second Edition*. Geneva: International Labour Office. https://www.ilo.org/global/publications/ilo-bookstore/order-online/books/WCMS_093550/lang--en/index.htm.

Amaral, M.J., Carretero, M.A., Bicho, R.C., Soares, A.M.V.M. and Mann, R.M. (2012a). The use of a lacertid lizard as a model for reptile ecotoxicology studies: Part 1 – Field demographics and morphology. *Chemosphere* 87, 757-764. https://doi.org/10.1016/j.chemosphere.2011.12.075.

Amaral, M.J., Bicho, R.C., Carretero, M.A., Sanchez-Hernandez, J.C., Faustino, A.M.R., Soares, A.M.V.M. and Mann, R.M. (2012b), The use of a lacertid lizard as a model for reptile ecotoxicology studies: Part 2 – Biomarkers of exposure and toxicity among pesticide exposed lizards. *Chemosphere* 87, 765-774. https://doi.org/10.1016/j.chemosphere.2012.01.048.

Ambrus, A. and Yang, Y.Z. (2016). Global Harmonization of Maximum Residue Limits for Pesticides. *Journal of Agricultural and Food Chemistry* 64(1), 30-35. https://doi.org/10.1021/jf505347z.

American Academy of Pediatrics (2012). Policy statement: pesticide exposure in children. *Pediatrics* 130(6), e1757. https://pediatrics.aappublications.org/content/pediatrics/130/6/e1757.full.pdf.

Ansara-Ross, T.M., Wepener, V., van den Brink, P.J. and Ross, M.J. (2012). Pesticides in South African fresh waters. *African Journal of Aquatic Science* 37(1), 1-16. https://doi.org/10.2989/16085914.2012.666336.

Araújo, C.V.M., Shinn, C., Moreira-Santos, M., Lopes, I., Espíndola, E.L.G. and Ribeiro, R. (2014). Copper-driven voidance and mortality in temperate and tropical tadpoles. *Aquatic Toxicology* 146, 70-75. https://doi.org/10.1016/j.aquatox.2013.10.030.

Arts, G.H., Buijse-Bogdan, L.L., Belgers, J.D.M., van Rhenen-Kersten, C.H., van Wijngaarden, R.P.A., Roessink, I. *et al.* (2006). Ecological impact in ditch mesocosms of simulated spray drift from a crop protection program for potatoes. *Integrated Environmental Assessment and Management* 2, 105-125. https://www.researchgate.net/deref/http%3A%2F%2Fdx.doi.org%2F10.1897%2F1551-3793(2006)2%5B105%3AEIIDMO%5D2.0.CO%3B2.

Arts, G.H.P., Beltman, W.H.J., Holterman, H.J., van Vliet, P.J.M., Wipfler, E.L. and van de Zande, J.C. (2017). *Including Multistress in the Risk Assessment of Plant Protection Products: Current State of Knowledge, Based on a Literature Review and an Evaluation of Tank Mixture Applications in a Spraying Schedule for Strawberries*. Wageningen, The Netherlands: Wageningen Environmental Research. https://doi.org/10.18174/408758.

Australian Pesticides and Veterinary Medicines Authority (2019). *Risk Assessment Manual – Environment*. https://apvma.gov.au/sites/default/files/publication/46416-risk_assessment_manual_environment_0.pdf.

Australian Pesticides and Veterinary Medicines Authority (2020). Listing of chemical reviews. https://apvma.gov.au/chemicals-and-products/chemical-review/listing. Accessed 1 December 2020.

Aziz, M.A., Tollington, S., Barlow, A., Goodrich, J., Shamsuddoha, M., Islam, M.A. and Groombridge, J.J. (2017). Investigating patterns of tiger and prey poaching in the bangladesh sundarbans: implications for improved management. *Global Ecology and Conservation* 9, 70-81. https://doi.org/10.1016/j.gecco.2016.12.001.

Babut, M., Arts, G.H., Barra Caracciolo, A., Carluer, N., Domange, N., Friberg, N. *et al.* (2013). Pesticide risk assessment and management in a globally changing world – report from a European interdisciplinary workshop. *Environmental Science and Pollution Research* 20(11), 8298-8312. https://doi.org/10.1007/s11356-013-2004-3.

Bai, Z., Caspari, T., Gonzalez, M.R., Batjes, N.H., Mäder, P., Bünemann, E.K. *et al*. (2018). Effects of agricultural management practices on soil quality: A review of long-term experiments for Europe and China. *Agriculture, Ecosystems and Environment* 265, 1-7. https://doi.org/10.1016/j.agee.2018.05.028.

Bain, D., Buttemer, W.A., Astheimer, L., Fildes, K. and Hooper, M.J. (2004). Effects of sublethal fenitrothion ingestion on cholinesterase inhibition, standard metabolism, thermal preference and prey-capture ability in the Australian central bearded dragon (*Pogona viticeps*, Agamidae). *Environmental Toxicology and Chemistry* 23(1), 109-116. https://setac.onlinelibrary.wiley.com/doi/abs/10.1897/02-555.

Baker, N.J., Bancroft, B.A. and Garcia, T.S. (2013). A meta-analysis of the effects of pesticides and fertilizers on survival and growth of amphibians. *Science of The Total Environment* 449, 150-156. http://dx.doi.org/10.1016/j.scitotenv.2013.01.056.

Baldi, I., Gruber, A., Rondeau, V., Lebailly, P., Brochard, P. and Fabrigoule, C. (2011). Neurobehavioral effects of long-term exposure to pesticides: results from the 4-year follow-up of the PHYTONER Study. *Occupational and Environmental Medicine* 68, 108e115. https://doi.org/10.1136/oem.2009.047811.

Battaglin, W.A., Smalling, K.L., Anderson, C., Calhoun, D., Chestnut, T. and Muths, E. (2016). Potential interactions among disease, pesticides, water quality and adjacent land cover in amphibian habitats in the United States. *Science of The Total Environment* 566-567, 320-332. http://dx.doi.org/10.1016/j.scitotenv.2016.05.062.

Begon, M., Harper, J.L. and Townsend, C.R. (1996). *Ecology – Individuals, Populations and Communities.* Third edition. Malden, MA and Oxford, UK: Blackwell Publishing.

Beketov, M.A., Kefford, B.J., Schäfer, R.B. and Liess, M. (2013). Pesticides reduce regional biodiversity of stream invertebrates. *Proceedings of the National Academy of Sciences of the United States of America* 110(27), 11039-11043. https://doi.org/10.1073/pnas.1305618110.

Berger, S., El Chazli, Y., Babu, A.F. and Coste, A.T. (2017). Azole resistance in *Aspergillus fumigatus*: A consequence of antifungal use in agriculture? *Frontiers in Microbiology* 8, 1024. https://doi.org/10.3389/fmicb.2017.01024.

Bergman, Å, Heindel, J.J., Kasten, T., Kidd, K.A., Jobling, S., Neira, M. *et al.* (2013). The impact of endocrine disruption: A consensus statement on the state of the science. *Environmental Health Perspectives* 121(4), A104-A106. https://doi.org/10.1289/ehp.1205448.

Bicho, R.C., Amaral, M.J., Faustino, A.M.R., Power, D.M., Rêma, A., Carretero, M.A., Soares, A.M.V.M. and Mann, R.M. (2013). Thyroid disruption in the lizard *Podarcis bocagei* exposed to a mixture of herbicides: A field study. *Ecotoxicology* 22, 156-165. https://doi.org/10.1007/s10646-012-1012-2.

Bienentreu, J.-F. and Lesbarrères, D. (2020). Amphibian disease ecology: Are we just scratching the surface? *Herpetologica* 76(2), 153-166. https://doi.org/10.1655/0018-0831-76.2.153.

Bishop, P.J., Angulo, A., Lewis, J.P., Moore, R.D., Rabb, G.B. and Garcia Moreno, J. (2012). The Amphibian Extinction Crisis – what will it take to put the action into the Amphibian Conservation Action Plan? *S.A.P.I.EN.S* 5(2). http://journals.openedition.org/sapiens/1406.

Blair, A., Ritz, B., Wesseling, C. and Freeman, L.B. (2015). Pesticides and human health. Editorial. *Occupational and Environmental Medicine* 72, 81-82. https://doi.org/10.1136/oemed-2014-102454.

Boedeker, W., Watts, M., Clausing, P. and Marquez, E. (2020). The global distribution of acute unintentional pesticide poisoning: Results of a systematic review. *BMC Public Health* 20, 1875 (2020). https://doi.org/10.1186/s12889-020-09939-0.

Boivin, A. and Poulsen, V. (2017). Environmental risk assessment of pesticides: State of the art and prospective improvement from science. *Environmental Science and Pollution Research* 24(8), 6889-6894. https://doi.org/10.1007/s11356-016-8289-2.

Bonita, R., Beaglehole, R., Kjellström, T. and World Health Organization (2006). *Basic Epidemiology*. Second edition. Geneva: World Health Organization. https://apps.who.int/iris/handle/10665/43541.

Bonvoisin T., Utyasheva, L., Knipe, D., Gunnell, D. and Eddleston, M. (2020). Suicide by pesticide poisoning in India: A review of pesticide regulations and their impact on suicide trends. *BMC Public Health* 20, 251. https://doi.org/10.1186/s12889-020-8339-z.

Boobis, A.R., Ossendorp, B.C., Banasiak, U., Hamey, P.Y., Sebestyen, I. and Moretto, A. (2018). Cumulative risk assessment of pesticide residues in food. *Toxicology Letters* 180, 137-150. https://doi.org/10.1016/j.toxlet.2008.06.004.

Bouktif Zarrouk, M., Gharbi, E., Maatouk, I., Leblanc J.-C. and Landoulsi, A. (2020). Dietary exposure of Tunisian adult population aged from 19 to 65 years old to pesticides residues. *Food Additives and Contaminants: Part A* 37(7), 1165-1179. https://doi.org/10.1080/19440049.2020.1744065.

Bouvier, J.-C., Ricci, B., Agerberg, J. and Lavigne, C. (2011). Apple orchard pest control strategies affect bird communities in southeastern France. *Environmental Toxicology and Chemistry* 30(1), 212-219. https://pubmed.ncbi.nlm.nih.gov/20928901/.

Boyd, I.L. (2018). An inside view on pesticide policy. *Nature Ecology and Evolution* 2(6), 920-921. https://doi.org/10.1038/s41559-018-0557-8.

Boye, K., Lindström, B., Boström, G. and Kreuger, J. (2019). Long-term data from the Swedish National Environmental Monitoring Program of Pesticides in Surface Waters. *Journal of Environmental Quality* 48(4), 1109. https://doi.org/10.2134/jeq2019.02.0056.

Boyle, N.K., Pitts-Singer, T.L., Abbott, J., Alix, A., Cox-Foster, D.L., Hinarejos, S. *et al.* (2019). Workshop on pesticide exposure assessment paradigm for non-*Apis* bees: Foundation and summaries. *Environmental Entomology* 48(1), 4-11. https://doi.org/10.1093/ee/nvy103.

Bradford Hill, A. (1965). The environment and disease: Association or causation? *Proceedings of the Royal Society of Medicine* 58, 295-300. https://www.ncbi.nlm.nih.gov/pmc/articles/PMC1898525/.

Brain, R.A. and Anderson, J.C. (2019). The agro-enabled urban revolution, pesticides, politics, and popular culture: a case study of land use, birds, and insecticides in the USA. *Environmental Science and Pollution Research* 26, 21717-21735. https://doi.org/10.1007/s11356-019-05305-9.

Brander, S.M., Gabler, M.K., Fowler, N.L., Connon, R.E. and Schlenk, D. (2016). Pyrethroid pesticides as endocrine disruptors: Molecular mechanisms in vertebrates with a focus on fishes. *Environmental Science and Technology* 50, 8977-8992. https://doi.org/10.1021/acs.est.6b02253.

Brevik, K., Schoville, S.D., Mota-Sanchez, D. and Chena, Y.H. (2018). Pesticide durability and the evolution of resistance: A novel application of survival analysis. *Pest Management Science* 74, 1953-1963. https://doi.org/10.1002/ps.4899.

Brittain, C.A., Vighi, M., Bommarco, R., Settele, J. and Potts. S.G. (2010). Impacts of a pesticide on pollinator species richness at different spatial scales. *Basic and Applied Ecology* 11(2), 106-115. https://doi.org/10.1016/j.baae.2009.11.007.

Brock, T.C.M., Alix, A., Brown, C.D., Capri, E., Gottesbüren, B.F.F., Heimbach. F. *et al*. (eds.) (2010). *Linking Aquatic Exposure and Effects – Risk Assessment of Pesticides.* London and New York: CRC Press. https://www.routledge.com/Linking-Aquatic-Exposure-and-Effects-Risk-Assessment-of-Pesticides/Brock/p/book/9781138112643.

Brodie, J. and Landos, M. (2019). Pesticides in Queensland and Great Barrier Reef waterways – potential impacts on aquatic ecosystems and the failure of national management. *Estuarine, Coastal and Shelf Science* 230, 106447. https://doi.org/10.1016/j.ecss.2019.106447.

Bruce-Vanderpuije, P., Megson, D., Reiner, E.J., Bradley, L., Adu-Kumi, S. and Gardella, J.A. (2019). The state of POPs in Ghana – A review on persistent organic pollutants: Environmental and human exposure. *Environmental Pollution* 245, 331-342. https://doi.org/10.1016/j.envpol.2018.10.107.

Brühl, C.A., Schmidt, T., Pieper, S. and Alscher, A. (2013). Terrestrial pesticide exposure of amphibians: An underestimated cause of global decline? *Scientific Reports* 3, 1135. https://doi.org/10.1038/srep01135.

Brühl, C.A. and Zaller, J. G. (2019). Biodiversity decline as a consequence of an Inappropriate Environmental Risk Assessment of pesticides. *Frontiers in Environmental Science* 7, 2013-2016. https://doi.org/10.3389/fenvs.2019.00177.

Buckle, A. and Prescott, C. (2018). Anticoagulants and risk mitigation. In *Anticoagulant Rodenticides and Wildlife. Emerging Topics in Ecotoxicology (Principles, Approaches and Perspectives) vol 5*. van den Brink, N.W., Elliott, J.E., Shore, R.F. and Rattner, B.A. (eds.). Cham: Springer. 319-355. https://doi.org/10.1007/978-3-319-64377-9_12.

Calvert, A.M., Bishop, C.A., Elliot, R.D., Krebs, E.A., Kydd, T.M., Machtans, C.S. and Robertson, G.J. (2013). A synthesis of human-related avian mortality in Canada. *Avian Conservation and Ecology* 8(2), 11. http://dx.doi.org/10.5751/ACE-00581-080211.

Calvert, G.M., Beckman, J., Bonnar Prado, J., Bojes, H., Schwartz, A., Mulay, P. *et al.* (2016). Acute occupational pesticide-related illness and injury – United States, 2007-2011. United States Centers for Disease Control and Prevention. *Morbidity and Mortality Weekly Report* 63 (55), 11-15. https://www.cdc.gov/mmwr/volumes/63/wr/mm6355a3.htm.

Calvert, G.M., Karnik, J., Mehler, L., Beckman, J., Morrissey, B., Sievert, J. *et al.* (2008). Acute pesticide poisoning among agricultural workers in the United States, 1998-2005. *American Journal of Industrial Medicine* 51, 883-898. https://doi.org/10.1002/ajim.20623.

Camenzuli, L., Scheringer, M. and Hungerbühler, K. (2016). Local organochlorine pesticide concentrations in soil put into a global perspective. *Environmental Pollution* 217, 11-18. https://doi.org/10.1016/j.envpol.2015.08.028.

Campbell, L.J., Pawlik, A.H. and Harrison, X.A. (2020). Amphibian ranaviruses in Europe: Important directions for future research. *FACETS* 5, 598-614. https://doi.org/10.1139/facets-2020-0007.

Canadian Food Inspection Agency (2019). *Pesticides and Metals in Selected Foods – April 1, 2018 to March 31, 2019. Food chemistry – Targeted surveys – Final Report*. https://www.inspection.gc.ca/DAM/DAM-food-aliments/STAGING/text-texte/pesticides_Metals_in_Selected_Foods_2018-2019_1596227305870_eng.pdf.

Canadian Food Inspection Agency (n.d.). *National Chemical Residue Monitoring Program and Chemistry Food Safety Oversight Program Annual Report 2015-2016*. https://www.inspection.gc.ca/food/chemical-residues-microbiology/food-safety-testing-bulletins/2019-04-17/annual-report-2015-2016/eng/1554904884535/1554905013036.

Cardoso, P., Barton, P. S., Birkhofer, K., Chichorro, F., Deacon, C., Fartmann *et al*. (2020). Scientists' warning to humanity on insect extinctions. *Biological Conservation* 242, 108426. https://doi.org/10.1016/j.biocon.2020.108426.

Castaño-Sánchez, A., Hose, G.C. and Reboleira, A.S.P.S. (2020). Ecotoxicological effects of anthropogenic stressors in subterranean organisms: A review. *Chemosphere* 244, 125422. https://doi.org/10.1016/j.chemosphere.2019.125422.

Cattan, P., Charlier, J.B., Clostre, F., Letourmy, P., Arnaud, L., Gresser, J. and Jannoyer, M. (2019). A conceptual model of organochlorine fate from a combined analysis of spatial and mid- to long-term trends of surface and ground water contamination in tropical areas (FWI). *Hydrology and Earth System Sciences* 23(2), 691-709. https://doi.org/10.5194/hess-23-691-2019.

Cavallaro, M.C., Main, A.R., Liber, K., Phillips, I.D., Headley, J.V., Peru, K.M. and Morrissey, C.A. (2019). Neonicotinoids and other agricultural stressors collectively modify aquatic insect communities. *Chemosphere* 226, 945-955. https://doi.org/10.1016/j.chemosphere.2019.03.176.

Cedergreen, N. (2014). Quantifying synergy: A systematic review of mixture toxicity studies within environmental toxicology. *PLoS One* 9(5): e9658. https://doi.org/10.1371/journal.pone.0096580.

Cha, E.S., Chang, S.S., Gunnell, D., Eddleston, M., Khang, Y.H. and Lee, W.J. (2016). Impact of paraquat regulation on suicide in South Korea. *International Journal of Epidemiology* 45(2), 470-479. https://doi.org/10.1093/ije/dyv304.

Chang, C.-H., MacIntosh, D., Lemos, B., Zhang, Q. and Lu, C. (2018). Characterization of daily dietary intake and the health risk of neonicotinoid insecticides for the US population. *Journal of Agricultural and Food Chemistry* 66(38), 10097-10105. https://doi.org/10.1021/acs.jafc.8b02982.

Chen, Y., Zang, L., Liu, M., Zhang, C., Shen, G., Du, W. *et al*. (2019). Ecological risk assessment of the increasing use of the neonicotinoid insecticides along the east coast of China. *Environment International* 127, 550-557. https://doi.org/10.1016/j.envint.2019.04.010.

Chiron, F., Chargé, R., Julliard, R., Jiguet, F. and Muratet, A. (2014). Pesticide doses, landscape structure and their relative effects on farmland birds. *Agriculture, Ecosystems and Environment* 185, 153-160. http://dx.doi.org/10.1016/j.agee.2013.12.013.

Chowdhury, F.R., Dewan, G., Verma, V.R., Knipe, D.W., Isha, I.T., Faiz, M.A. *et al*. (2018). Bans of WHO Class I pesticides in Bangladesh – suicide prevention without hampering agricultural output. *International Journal of Epidemiology* 47(1), 175-184. https://doi.org/10.1093/ije/dyx157.

Close, M.E. and Skinner, A. (2012). Sixth national survey of pesticides in groundwater in New Zealand. *New Zealand Journal of Marine and Freshwater Research* 46(4), 443-457. https://doi.org/10.1080/00288330.2012.707131.

Cloyd, R.A. (2012). Indirect effects of pesticides on natural enemies. In *Pesticides – Advances in Chemical and Botanical Pesticides*. Soundararajan, R.P. (ed.). IntechOpen. Chapter 6. https://doi.org/10.5772/48649.

Coates, P.S., Brussee, B.E., Howe, K.B., Fleskes, J.P., Dwight, I.A., Connelly, D.P. *et al*. (2017). Long-term and widespread changes in agricultural practices influence ring-necked pheasant abundance in California. *Ecology and Evolution* 7, 2546-2559. https://pubmed.ncbi.nlm.nih.gov/28428846/.

Cochrane Library (2020). About Cochrane Reviews. https://www.cochranelibrary.com/about/about-cochrane-reviews. Accessed 20 December 2020.

Convention on Biological Diversity (1992). Text of the Convention. https://www.cbd.int/convention/text/. Accessed 20 December 2020.

Convention on Biological Diversity (2019). Identification, monitoring, indicators and assessments. https://www.cbd.int/indicators/. Accessed 20 December 2020.

Copplestone, J.F. (1977). A global view of pesticide safety. In *Pesticide Management and Insecticide Resistance*. Watson, D.L. and Brown, A.W.A. (eds.). New York: Academic Press. 147-155. https://doi.org/10.1016/B978-0-12-738650-8.50014-X.

Corriols, M., Marín, J., Berroteran, J., Marina Lozano, L., Lundberg, I. and Thörn, Å. (2008). The Nicaraguan pesticide poisoning register: Constant underreporting. *International Journal of Health Services* 38(4), 773-787. https://doi.org/10.2190/HS.38.4.k.

Coscollà, C. and Yusa, V. (2016). Pesticides and agricultural air quality. In *The Quality of Air. Comprehensive Analytical Chemistry vol 73*. de la Guardia, M. and Armenta, S. (eds.). Elsevier. Chapter 17. https://www.elsevier.com/books/the-quality-of-air/de-la-guardia/978-0-444-63605-8.

Crépet, A., Luong, T.M., Baines, J., Boon, P.E., Ennis, J., Kennedy, M. *et al*. (2021). An international probabilistic risk assessment of acute dietary exposure to pesticide residues in relation to codex maximum residue limits for pesticides in food. *Food Control* 121, 107563. https://doi.org/10.1016/j.foodcont.2020.107563.

Croft, B.A. (1990). Natural enemies and pesticides: An overview. In *Arthropod Biological Control Agents and Pesticides*. Croft, B.A. (ed.). New York: Wiley and Sons. Chapter 1.

Croft, B.A. and Theiling, K.M. (1990). Pesticide effects on arthropod natural enemies: A database. In *Arthropod Biological Control Agents and Pesticides*. Croft, B.A. (ed.). New York: Wiley and Sons.

CropLife International (2020). Resistance management. https://croplife.org/crop-protection/stewardship/resistance-mangement. Accessed 20 December 2020.

Cutler, G.C., Scott-Dupree, C.D. and Drexler, D.M. (2014). Honey bees, neonicotinoids and bee incident reports: The Canadian situation. *Pest Management Science* 70(5), 779-783. https://pubmed.ncbi.nlm.nih.gov/23873722/.

D'Costa, V.M, King, C.E., Kalan, L., Morar, M., Sung, W.W.L., Schwarz, C. *et al.* (2011). Antibiotic resistance is ancient. *Nature* 477, 457-461. https://doi.org/10.1038/nature10388.

Daam, M.A., Chelinho, S., Niemeyer, J.C., Owojori, O J., De Silva, P.M.C.S., Sousa, J.P. *et al*. (2019). Environmental risk assessment of pesticides in tropical terrestrial ecosystems: Test procedures, current status and future perspectives. *Ecotoxicology and Environmental Safety* 181, 534-547. https://doi.org/10.1016/j.ecoenv.2019.06.038.

Daam, M.A., Leitão, S., Cerejeira, M.J. and Paulo Sousa, J. (2011). Comparing the sensitivity of soil invertebrates to pesticides with that of *Eisenia fetida*. *Chemosphere* 85(6), 1040-1047. https://doi.org/10.1016/j.chemosphere.2011.07.032.

Dahshan, H., Megahed, A.M., Abd-Elall, A.M.M., Abd-El-Kader, M.A.G., Nabawy, E. and Elbana, M.H. (2016). Monitoring of pesticides water pollution – The Egyptian River Nile. *Journal of Environmental Health Science and Engineering* 14(1), 1-9. https://doi.org/10.1186/s40201-016-0259-6.

Dainese, M., Martin, E.A., Aizen, M.A., Albrect, M., Bartomeus, I., Bommarco, R. *et al.* (2019). A global synthesis reveals biodiversity-mediated benefits for crop production. *Science* Advances 5(10), eaax0121. https://doi.org/10.1126/sciadv.aax0121.

Dansereau, C., Perez, M., Kegley, S.E., Tupper, K.A. and Wang, A. (2006). *Poisons on the Wind – Community Air Monitoring for Chlorpyrifos in the Yakima Valley.* Seattle and San Francisco: Farm Worker Pesticide Project and Pesticide Action Network North America. http://www.panna.org/sites/default/files/POW12-20-06YakimaSUM.pdf.

Dawson, A.H., Eddleston, M., Senarathna, L., Mohamed, F., Gawarammana, I., Bowe, S.J. *et al*. (2010). Acute human lethal toxicity of agricultural pesticides: A prospective cohort study. *PLoS Medicine* 7(10), e1000357. https://doi.org/10.1371/journal.pmed.1000357.

de Graaff, M.A., Hornslein, N., Throop, H.L., Kardol, P. and van Diepen, L.T.A. (2019). Effects of agricultural intensification on soil biodiversity and implications for ecosystem functioning: A meta-analysis. In *Advances in Agronomy vol. 155*. Academic Press. Chapter 1, 1-44. https://doi.org/10.1016/bs.agron.2019.01.001.

de Souza, R.M., Seibert, D., Quesada, H.B., de Jesus Bassetti, F., Fagundes-Klen, M.R and Bergamasco, R. (2020). Occurrence, impacts and general aspects of pesticides in surface water: A review. *Process Safety and Environmental Protection* 135, 22-37. https://doi.org/10.1016/j.psep.2019.12.035.

de Weert, J., Roex, E., Klein, J. and Janssen, G. (2014). *Establishment of the National Monitoring Network Plant Protection Products in Agriculture and Horticulture*. Delft: Deltares. http://publications.deltares.nl/1207762_008.pdf. (in Dutch).

Department of Agriculture, Australia (2019). *National Residue Survey 2018-19 – Annual Summary*. http://www.agriculture.gov.au/ag-farm-food/food/nrs.

Derbalah, A., Chidya, R., Jadoon, W. and Sakugawa, H. (2019). Temporal trends in organophosphorus pesticides use and concentrations in river water in Japan, and risk assessment. *Journal of Environmental Sciences* 79, 135-152. https://doi.org/10.1016/j.jes.2018.11.019.

Dereumeaux, C., Fillol, C., Quenel, P. and Denys, S. (2020). Pesticide exposures for residents living close to agricultural lands: A review. *Environment International* 134, 105210. https://doi.org/10.1016/j.envint.2019.105210.

Deziel, N.C., Beane Freeman, L.E., Graubard, B.I., Jones, R.R., Hoppin, J.A., Thomas, K. *et al.* (2017). Relative contributions of agricultural drift, para-occupational, and residential use exposure pathways to house dust pesticide concentrations: meta-regression of published data. *Environmental Health Perspectives* 125(3), 206-305. http://dx.doi.org/10.1289/EHP426.

Deziel, N.C., Friesen, M.C., Hoppin, J.A., Hines, C.J., Thomas, K. and Beane Freeman, L.E. (2015). A review of nonoccupational pathways for pesticide exposure in women living in agricultural areas. *Environmental Health Perspectives* 123(6), 515-524. http://dx.doi.org/10.1289/ehp.1408273.

Dhiman, R.C. and Yadav, R.S. (2016). Insecticide resistance in phlebotomine sandflies in Southeast Asia with emphasis on the Indian subcontinent. *Infectious Diseases of Poverty* 5, 106 https://doi.org/10.1186/s40249-016-0200-3.

Di Lorenzo, T., Cifoni, M., Fiasca, B., Di Cioccio, A., and Galassi, D.M.P. (2018). Ecological risk assessment of pesticide mixtures in the alluvial aquifers of central Italy: Toward more realistic scenarios for risk mitigation. *Science of The Total Environment* 644, 161-172. https://doi.org/10.1016/j.scitotenv.2018.06.345.

Dirbaba, N.B., Li, S., Wu, H., Yan, X. and Wang, J. (2018). Organochlorine pesticides, polybrominated diphenyl ethers and polychlorinated biphenyls in surficial sediments of the Awash River Basin, Ethiopia. *PLoS ONE* 13(10). https://doi.org/10.1371/journal.pone.0205026.

Dutcher, J.D. (2007). A review of resurgence and replacement causing pest outbreaks in IPM. In *General Concepts in Integrated Pest and Disease Management*. Ciancio, A. and Mukerji, K.G. (eds.). Dordrecht, The Netherlands: Springer. 27-43. https://www.springer.com/gp/book/9781402060601.

Eardley, C., Freitas, B.M., Kevan, P.G., Rader, R., Gikungu, M., Klein, A.M. *et al*. (2016). Background to pollinators, pollination and food production. In *The Assessment Report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on Pollinators, Pollination and Food Production*. Potts, S.G., Imperatriz-Fonseca, V.L. and Ngo, H.T. (eds.). Bonn: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Chapter 1, 1-25. https://ipbes.net/assessment-reports/pollinators.

Eddleston, M. (2018). Response to Jors *et al*., *Environmental Health Insights. Environmental Health Insights* 12, 1178630218788554. https://doi.org/10.1177/1178630218788554.

Eddleston, M. and Phillips, M.R. (2004). Self poisoning with pesticides. *BMJ* 328, 42-44. https://doi.org/10.1136/bmj.328.7430.42.

Egea-Serrano, A., Relyea, R.A., Tejedo, M. and Torralva, M. (2012). Understanding of the impact of chemicals on amphibians: A meta-analytic review. *Ecology and Evolution* 2(7), 1382-1397. https://onlinelibrary.wiley.com/doi/full/10.1002/ece3.249.

Elibariki, R. and Maguta, M.M. (2017). Status of pesticides pollution in Tanzania – A review. *Chemosphere* 178, 154-164. https://doi.org/10.1016/j.chemosphere.2017.03.036.

Emmerson, M., Morales, M.B., Oñate, J.J., Batáry, P., Berendse, F., Liira, J *et al*. (2016). Chapter Two – How agricultural intensification affects biodiversity and ecosystem services. *Advances in Ecological Research* 55, 43-97. http://dx.doi.org/10.1016/bs.aecr.2016.08.005.

Encarnação, T., Pais, A.A.C.C., Campos, M.G. and Burrows, H.D. (2019). Endocrine disrupting chemicals: Impact on human health, wildlife and the environment. *Science Progress* 102(1), 3-42. https://doi.org/10.1177/0036850419826802.

Eng, M.L., Stutchbury, B.J.M. and Morrissey, C.A. (2019). A neonicotinoid insecticide reduces fueling and delays migration in songbirds. *Science* 365 (6458), 1177-1180. https://doi.org/10.1126/science.aaw9419.

Eqani, S.A., Malik, R.N., Alamdar, A. and Faheem, H. (2012). Status of organochlorine contaminants in the different environmental compartments of Pakistan: A review on occurrence and levels. *Bulletin of Environmental Contamination and Toxicology* 88(3), 303–310. https://doi.org/10.1007/s00128-011-0496-4.

Ertl, H.M.H., Mora, M.A., Brightsmith, D.J. and Navarro-Alberto, J.A. (2018). Potential impact of neonicotinoid use on Northern bobwhite (*Colinus virginianus*) in Texas: A historical analysis. *PLoS ONE* 13(1), e0191100. https://doi.org/10.1371/journal.pone.0191100.

Estep, A.S., Sanscrainte, N.D., Waits, C.M., Bernard, S.J., Lloyd, A.M., Lucas K.J. *et al*. (2018). Quantification of permethrin resistance and *kdr* alleles in Florida strains of *Aedes aegypti* (L.) and *Aedes albopictus* (Skuse). *PLoS Neglected Tropical Diseases* 12 (10), e0006544. https://doi.org/10.1371/journal.pntd.0006544.

European Agency for Safety and Health at Work (2018). Directive 89/391/EEC - OSH "Framework Directive" of 12 June 1989 on the introduction of measures to encourage improvements in the safety and health of workers at work. https://osha.europa.eu/en/legislation/directives/the-osh-framework-directive/1. Accessed 28 December 2020.

European Chemical Agency and European Food Safety Authority (2018). Guidance for the identification of endocrine disruptors in the context of Regulations (EU) No 528/2012 and (EC) No 1107/2009. *EFSA Journal* 16(6), 5311. https://www.efsa.europa.eu/en/efsajournal/pub/5311.

European Commission (2018). *Technical Guidance for Deriving Environmental Quality* Standards. Guidance Document No. 27, Updated version 2018. https://circabc.europa.eu/sd/a/ba6810cd-e611-4f72-9902-f0d8867a2a6b/Guidance%20No%2027%20-%20Deriving%20Environmental%20Quality%20Standards%20-%20version%202018.pdf.

European Environment Agency (2012). *The Impacts of Endocrine Disrupters on Wildlife, People and their Environments: The Weybridge+15 (1996- 2011) Report*. https://www.eea.europa.eu/publications/the-impacts-of-endocrine-disrupters.

European Environment Agency (2019). Waterbase – water quality IBM. https://www.eea.europa.eu/data-and-maps/data/waterbase-water-quality-icm. Accessed 20 December 2020.

European Food Safety Authority (2009). Guidance document on risk assessment for birds and mammals. *EFSA Journal* 7(12), 1438. https://doi.org/10.2903/j.efsa.2009.1438.

European Food Safety Authority (2013a). Guidance on the risk assessment of plant protection products on bees (*Apis mellifera*, *Bombus* spp. and solitary bees). *EFSA Journal* 11(7), 3295. https://doi.org/10.2903/j.efsa.2013.3295.

European Food Safety Authority (2013b). Guidance on tiered risk assessment for plant protection products for aquatic organisms in edge-of-field surface waters. *EFSA Journal* 11(7), 3290. https://doi.org/10.2903/j.efsa.2013.3290.

European Food Safety Authority (2014). Guidance on the assessment of exposure of operators, workers, residents and bystanders in risk assessment for plant protection products. *EFSA Journal* 12(10), 3874. https://doi.org/10.2903/j.efsa.2014.3874.

European Food Safety Authority (2015). Scientific Opinion on the effect assessment for pesticides on sediment organisms in edge-of-field surface water. *EFSA Journal* 13. https://doi.org/10.2903/j.efsa.2015.4176.

European Food Safety Authority (2018a). Scientific Opinion on the state of the science on pesticide risk assessment for amphibians and reptiles. EFSA Panel on Plant Protection Products and their Residues: Ockleford, C., Adriaanse, P., Berny, P., Brock, T., Duquesne, S., Grilli, S. *et al. EFSA Journal* 16(2), 5125. https://doi.org/10.2903/j.efsa.2018.5125.

European Food Safety Authority (2018b). Conclusion on the peer review of the pesticide risk assessment for bees for the active substance clothianidin considering the uses as seed treatments and granules. *EFSA Journal* 16(2), 5177. https://doi.org/10.2903/j.efsa.2018.5177.

European Food Safety Authority (2018c). Conclusion on the peer review of the pesticide risk assessment for bees for the active substance imidacloprid considering the uses as seed treatments and granules. *EFSA Journal* 16(2), 5178. https://doi.org/10.2903/j.efsa.2018.5178.

European Food Safety Authority (2018d). Conclusions on the peer review of the pesticide risk assessment for bees for the active substance thiamethoxam considering the uses as seed treatments and granules. *EFSA Journal* 16(2), 5179. https://doi.org/10.2903/j.efsa.2018.5179.

European Food Safety Authority (2018e). Use of EFSA pesticide residue intake model. *EFSA Journal* 16(1), 5147. https://doi.org/10.2903/j.efsa.2018.5147.

European Food Safety Authority (2020a). Definitions of EFSA scientific outputs and supporting publications. https://www.efsa.europa.eu/en/efsajournal/scdocdefinitions. Accessed 1 December 2020.

European Food Safety Authority (2020b). The 2018 European Union report on pesticide residues in food. *EFSA Journal* 18(4), e60570. https://efsa.onlinelibrary.wiley.com/doi/10.2903/j.efsa.2020.6057.

European Food Safety Authority (2020c). Scientific report on cumulative dietary risk characterisation of pesticides that have acute effects on the nervous system. European Food Safety Authority: Craig, P.S., Dujardin, B., Hart, A., Hernandez-Jerez, A.F., Hougaard Bennekou, S., Kneuer, C. *et al*. *EFSA Journal* 18(4), 6087. https://doi.org/10.2903/j.efsa.2020.6087.

European Food Safety Authority (2020d). Scientific report on the cumulative dietary risk characterisation of pesticides that have chronic effects on the thyroid. *EFSA Journal* 18(4), 6088. https://doi.org/10.2903/j.efsa.2020.6088.

European Food Safety Authority Panel on Plant Protection Products and their Residues (2017). Scientific Opinion of the PPR Panel on the follow-up of the findings of the External Scientific Report "Literature review of epidemiological studies linking exposure to pesticides and health effects". *EFSA Journal* 15(10): 5007 https://doi.org/10.2903/j.efsa.2017.5007.

European Food Safety Authority Scientific Committee (2019). Guidance on harmonised methodologies for human health, animal health and ecological risk assessment of combined exposure to multiple chemicals. *EFSA Journal* 17(3), 5634. https://doi.org/10.2903/j.efsa.2019.5634.

European Parliamentary Research Service (2019). *Endocrine Disruptors: An Overview of Latest Developments at European Level in the Context of Plant Protection Products*. https://www.europarl.europa.eu/RegData/etudes/STUD/2019/631743/EPRS_STU(2019)631743_EN.pdf.

European Union (1998). Council Directive 98/83/EC of 3 November 1998 on the quality of water intended for human consumption. *Official Journal of the European Communities* L(330), 32-34. https://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:1998:330:0032:0054:EN:PDF.

European Union (2000). Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. https://eur-lex.europa.eu/legal-content/EN/TXT/HTML/?uri=CELEX:32000L0060&from=EN. Accessed 30 December 2020.

European Union (2013). Directive 2008/105/EC of the European Parliament and of the Council of 16 December 2008 on environmental quality standards in the field of water policy. Amended by Directive 2013/39/EU of 12 August 2013. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A02008L0105-20130913. Accessed 20 December 2020.

European Union (2020a). *Proposal for a Directive of the European Parliament and of the Council on the Quality of Water Intended for Human Consumption (recast) – Political Agreement*. https://op.europa.eu/en/publication-detail/-/publication/13def1fc-5711-11ea-8b81-01aa75ed71a1/language-en/format-PDF.

European Union (2020b). EU action on antimicrobial resistance. https://ec.europa.eu/health/antimicrobial-resistance/eu-action-on-antimicrobial-resistance_en. Accessed 28 December 2020.

Ewald, J.A., Wheatley, C.J., Aebischer, N.J., Moreby, S.J., Duffield, S.J., Crick, H.Q.P. and Morecroft, M.B. (2015). Influences of extreme weather, climate and pesticide use on invertebrates in cereal fields over 42 years. *Global Change Biology* 21(11), 3931-3950. https://onlinelibrary.wiley.com/doi/abs/10.1111/gcb.13026.

Faheem, N., Sajjad, A., Mehmood, Z., Iqbal, F., Mahmood, Q., Munsif, S. and Waseem, A. (2015). Pesticide exposure through fruits and meat in Pakistan. *Fresenius Environmental Bulletin* 24(12B), 4555-4566. https://www.prt-parlar.de/download/.

FAOSTAT (2020). Food and agriculture data. *Food and Agriculture Organization of the United Nations*. http://www.fao.org/faostat/en/#home. Accessed 26 December 2020.

Fenske, R.A., Lu, C., Negrete, M. and Galvin, K. (2013). Breaking the take home pesticide exposure pathway for agricultural families: Workplace predictors of residential contamination. *American Journal of Industrial Medicine* 56, 1063-1071. https://doi.org/10.1002/ajim.22225.

Fisher, M.S., Hawkins, N.J., Sanglard, D. and Gurr, S.J. (2018). Worldwide emergence of resistance to antifungal drugs challenges human health and food security. *Science* 360(6390), 739-742. https://doi.org/10.1126/science.aap7999.

FOCUS (2008). *Pesticides in Air: Considerations for Exposure Assessment. Report Prepared by the FOCUS Working Group on Pesticides in Air*. European Commission (EC) Document Reference SANCO/10553/2006 Rev 2 June 2008. https://esdac.jrc.ec.europa.eu/public_path/projects_data/focus/air/docs/FOCUS_AIR_GROUP_REPORT-FINAL.pdf.

Food and Agriculture Organization of the United Nations (2012). *International Code of Conduct on the Distribution and Use of Pesticides: Guidelines on Prevention and Management of Pesticide Resistance.* Rome. http://www.fao.org/publications/card/en/c/8dcf273c-c907-4e71-b5e5-8753a861de87/.

Food and Agriculture Organization of the United Nations (2015). *The FAO Action Plan on Antimicrobial Resistance 2016-2020.* Rome. http://www.fao.org/3/a-i5996e.pdf.

Food and Agriculture Organization of the United Nations (2016). *Submission and Evaluation of Pesticides Residues Data for the Estimation of Maximum Residue Levels in Food and Feed.* 3rd edition. http://www.fao.org/3/i5452e/I5452E.pdf.

Food and Agriculture Organization of the United Nations (2019). *The State of the World's Biodiversity for Food and Agriculture*. Bélanger, J. and Pilling, D. (eds.). http://www.fao.org/3/CA3129EN/CA3129EN.pdf.

Food and Agriculture Organization of the United Nations (2020). List of pesticides that have been evaluated by the Joint FAO/WHO Meeting on Pesticide Residues (JMPR) and the Joint FAO/WHO Meeting on Pesticide Specifications (JMPS). http://www.fao.org/agriculture/crops/thematic-sitemap/theme/pests/lpe/en/. Accessed 28 December 2020.

Food and Agriculture Organization of the United Nations and Intergovernmental Technical Panel on Soils (2015). *Status of the World's Soil Resources (SWSR) – Main Report.* Rome. http://www.fao.org/documents/card/en/c/c6814873-efc3-41db-b7d3-2081a10ede50/.

Food and Agriculture Organization of the United Nations and Intergovernmental Technical Panel on Soils (2017). *Global Assessment of the Impact of Plant Protection Products on Soil Functions and Soil Ecosystems*. Rome. http://www.fao.org/documents/card/en/c/I8168EN/.

Food and Agriculture Organization of the United Nations, Intergovernmental Technical Panel on Soils, Global Soil Biodiversity Initiative, Secretariat of the Convention of Biological Diversity and European Commission (2020). *State of Knowledge of Soil Biodiversity – Status, Challenges and Potentialities, Report 2020*. https://doi.org/10.4060/cb1928en.

Food and Agriculture Organization of the United Nations and World Health Organization (2014). *The International Code of Conduct on Pesticide Management.* Rome and Geneva. http://www.fao.org/3/I3604E/i3604e.pdf.

Food and Agriculture Organization of the United Nations and World Health Organization (2019). *Joint FAO/WHO Expert Meeting in collaboration with OIE on Foodborne Antimicrobial Resistance: Role of the Environment, Crops and Biocides – Meeting Report.* Microbiological Risk Assessment Series No. 34. Rome and Geneva. http://www.fao.org/3/ca6724en/ca6724en.pdf.

Forister, M.L., Cousens, B., Harrison, J.G., Anderson, K., Thorne, J.H., Waetjen, D. *et al*. (2016). Increasing neonicotinoid use and the declining butterfly fauna of lowland California. *Biology Letters* 12, 8. https://doi.org/10.1098/rsbl.2016.0475.

Fosu-Mensah, B.Y., Okoffo, E.D., Darko, G. and Gordon, C. (2016). Assessment of organochlorine pesticide residues in soils and drinking water sources from cocoa farms in Ghana. *SpringerPlus* 5, 869. https://doi.org/10.1186/s40064-016-2352-9.

Franz, J.M., Bogenschütz, H., Hassan, S.A., Huang, P., Naton, E., Suter, H. and Viggiani, G. (1980). Results of a joint pesticide test programme by the Working Group: Pesticides and beneficial arthropods. *Entomophaga* 25, 231-236. https://doi.org/10.1007/BF02371922.

Freeman, S., Kaufman-Shriqui, V., Berman, T., Varsano, R., Shahar, D.R. and Manor, O. (2016). Children's diets, pesticide uptake, and implications for risk assessment: An Israeli case study. *Food and Chemical Toxicology* 87, 88-96. https://doi.org/10.1016/j.fct.2015.11.009.

Frisvold, G.B., Bagavathiannan, M.V. and Norsworthy, J.K. (2017). Positive and normative modeling for Palmer amaranth control and herbicide resistance management. *Pest Management Science* 73(6), 1110-1120. https://doi.org/10.1002/ps.4537.

Fuhrimann, S., Staudacher, P., Lindh, C., van Wendel de Joode, B., Mora, A.M., Winkler, M.S. and Kromhout, H. (2019). Variability and predictors of weekly pesticide exposure in applicators from organic, sustainable and conventional smallholder farms in Costa Rica. *Occupational and Environmental Medicine* 77(1), 40-47. https://doi.org/10.1136/oemed-2019-105884.

Gallai, N., Garibaldi, L.A., Li, X., Breeze, T.D., Espírito Santo, M.M., Rodriguez Fernandez, J. *et al.* (2016). Economic valuation of pollinator gains and losses. In *The Assessment Report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on Pollinators, Pollination and Food Production*. Potts, S.G., Imperatriz-Fonseca, V.L. and Ngo, H.T. (eds.). Bonn: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Chapter 4, 205-273. https://ipbes.net/assessment-reports/pollinators.

Garrigou, A., Laurent, C., Berthet, A., Colosio, C., Jas, N., Daubas-Letourneux, V. *et al*. (2020). Critical review of the role of PPE in the prevention of risks related to agricultural pesticide use. *Safety Science* 123, 104527. https://doi.org/10.1016/j.ssci.2019.104527.

Geiger, F., Bengtsson, J., Berendse, F., Weisser, W.W., Emmerson, M., Morales, M.B. *et al.* (2010). Persistent negative effects of pesticides on biodiversity and biological control potential on European farmland. *Basic and Applied Ecology* 11(2), 97-105. https://doi.org/10.1016/j.baae.2009.12.001.

Gendron, A. (2013). Amphibian ecotoxicology. In *Encyclopedia of Aquatic Ecotoxicology*. Dordreccht: Springer. Férard, J.F. and Blaise, C. (eds.). https://doi.org/10.1007/978-94-007-5704-2_4.

Gibbons, D., Morrissey, C. and Mineau, P. (2015). A review of the direct and indirect effects of neonicotinoids and fipronil on vertebrate wildlife. Environm*ental Science and Pollution Research International* 22(1), 103-118. https://doi.org/10.1007/s11356-014-3180-5.

Gibbs, K.E., MacKey, R.L. and Currie, D.J. (2009). Human land use, agriculture, pesticides and losses of imperiled species. *Diversity and Distributions* 15(2), 242–253. https://doi.org/10.1111/j.1472-4642.2008.00543.x.

Gilburn, A.S., Bunnefeld, N., McVean Wilson, J., Botham, M.S., Brereton, T.M., Fox, R. and Goulson, D. (2015). Are neonicotinoid insecticides driving declines of widespread butterflies? *PeerJ* 11, 1-13. https://doi.org/10.7717/peerj.1402.

Godfray, H.C.J., Blacquière, T., Field, L.M., Hails, R.S., Petrokofsky, G., Potts, S.G. *et al.* (2015). A restatement of the natural science evidence base concerning neonicotinoid insecticides and insect pollinators. *Proceedings of the Royal Society B: Biological Sciences* 281, 20140558. http://dx.doi.org/10.1098/rspb.2015.1821.

Gomes, H.O., Jorge Menezes, J.M.C., da Costa, J.G.M., Coutinho, H.D.M., Teixeira, R.N.P. and do Nascimento, R.F. (2020). Evaluating the presence of pesticides in bananas: An integrative review. *Ecotoxicology and Environmental Safety* 189, 110016. https://doi.org/10.1016/j.ecoenv.2019.110016.

Goss, D. and Wauchope, R.D. (1990). The SCR/ARS/CES pesticides properties database: II Using it with soils data in a screening procedure. In *Pesticides in the Next Decade: The Challenge Ahead*. Weigmann, D.L. (ed.). Virginia Water Resources Research Center (United States). 471-493. https://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/technical/ecoscience/invasive/?cid=stelprdb1044493.

Gould, F., Brown, Z.S. and Kuzma, J. (2018). Wicked evolution: Can we address the sociobiological dilemma of pesticide resistance? *Science* 360(6390), 728-732. https://doi.org/10.1126/science.aar3780.

Government of Canada (2020a). Sex- and gender-based considerations in the scientific risk assessment of pesticides in Canada, 21 January. https://www.canada.ca/en/health-canada/services/consumer-product-safety/reports-publications/pesticides-pest-management/fact-sheets-other-resources/sex-and-gender-based-considerations-in-the-scientific-risk-assessment-of-pesticides-in-canada.html. Accessed 19 January 2022

Government of Canada (2020b). Consideration of sex and gender in pesticide risk assessment, 21 February. https://www.canada.ca/en/health-canada/services/consumer-product-safety/reports-publications/pesticides-pest-management/fact-sheets-other-resources/consideration-sex-gender-pesticide-risk-assessment-infographic.html. Accessed 19 January 2022

Grant, S., Thompson, K., Paxman, C., Elisei, G., Gallen C., Tracey, D. *et al.* (2018). *Marine Monitoring Program: Annual Report for Inshore Pesticide Monitoring 2016-2017*. Great Barrier Reef Marine Park Authority. https://hdl.handle.net/11017/3399.

Grilo, A., Moreira, A., Carrapiço, B., Belas, A. and Braz, B.S. (2021). Epidemiological study of pesticide poisoning in domestic animals and wildlife in portugal: 2014-2020. *Frontiers in Veterinary Science* 7, 1181. https://doi.org/10.3389/fvets.2020.616293/bibtex.

Gross, K. and Rosenheim, J.A. (2011). Quantifying secondary pest outbreaks in cotton and their monetary cost with causal-inference statistics. *Ecological Applications* 21(7), 2770-2780. https://www.jstor.org/stable/41416693.

Group of Chief Scientific Advisors (2018). *EU Authorisation Processes of Plant Protection Products – from a Scientific Point of View.* Scientific Advice Mechanism (SAM), Scientific Opinion 5. Directorate-General for Research and Innovation, European Commission, Brussels. https://op.europa.eu/en/publication-detail/-/publication/5306df12-79b9-11e8-ac6a-01aa75ed71a1/language-en/format-PDF/source-94583924.

Grung, M., Lin, Y., Zhang, H., Steen, A.O., Huang, J., Zhang, G. and Larssen, T. (2015). Pesticide levels and environmental risk in aquatic environments in China – A review. *Environment International* 81, 87-97. https://doi.org/10.1016/j.envint.2015.04.013.

Grzywacz, D., Rossbach, A., Rauf, A., Russell, D.A., Srinivasan, R. and Shelton, A.M. (2010). Current control methods for diamondback moth and other brassica insect pests and the prospects for improved management with lepidopteran-resistant Bt vegetable brassicas in Asia and Africa. *Crop Protection* 29, 68-79. https://doi.org/10.1016/j.cropro.2009.08.009.

Guedes, R.N.C., Beins, K., Navarro Costa, D., Coelho, G. E. and Bezerra, H.S. da S. (2020). Patterns of insecticide resistance in *Aedes aegypti*: Meta-analyses of surveys in Latin America and the Caribbean. *Pest Management Science* 76(6), 2144-2157. https://doi.org/10.1002/ps.5752.

Guida, Y. de S., Meire, R O., Torres, J.P.M. and Malm, O. (2018). Air contamination by legacy and current-use pesticides in Brazilian mountains: An overview of national regulations by monitoring pollutant presence in pristine areas. *Environmental Pollution* 242, Part A, 19-30. https://doi.org/10.1016/j.envpol.2018.06.061.

Guitart, R., Sachana, M., Caloni, F., Croubels, S., Vandenbroucke, V. and Berny, P. (2010). Animal poisoning in Europe. Part 3: Wildlife. The *Veterinary Journal* 183, 260-265. https://doi.org/10.1016/j.tvjl.2009.03.033.

Gunnell, D., Eddleston, M., Phillips, M.R. and Konradsen, F. (2007). The global distribution of fatal pesticide self-poisoning: Systematic review. *BMC Public Health* 7, 357. https://doi.org/10.1186/1471-2458-7-357.

Gunnell, D., Knipe, D., Chang, S.-S., Pearson, M., Konradsen, F., Lee, W.J. and Eddleston, M. (2017). Prevention of suicide with regulations aimed at restricting access to highly hazardous pesticides: A systematic review of the international evidence. *Lancet Global Health* 5, e1026-37. http://dx.doi.org/10.1016/S2214-109X(17)30299-1.

Hallmann, C.A., Foppen, R.P.B., van Turnhout, C.A.M., de Kroon, H. and Jongejans, E. (2014). Declines in insectivorous birds are associated with high neonicotinoid concentrations. *Nature* 511, 341. https://doi.org/10.1038/nature13531.

Hanson, M.L., Solomon, K.R., Van Der Kraak, G.J. and Brian, R.A. (2019). Effects of atrazine on fish, amphibians, and reptiles: update of the analysis based on quantitative weight of evidence. *Critical Reviews in Toxicology*, 49(8), 670-709. https://doi.org/10.1080/10408444.2019.1701985.

Hardman, J.M., Franklin, J.L., Jensen, K.I.N. and Moreau, D.L. (2006). Effects of pesticides on mite predators (Acari: Phytoseiidae) and colonization of apple trees by *Tetranychus urticae*. *Phytoparasitica* 34, 449-462. https://doi.org/10.1007/BF02981199.

Hasan, M.K., Shahriar, A. and Jim, K.U. (2019). Water pollution in Bangladesh and its impact on public health. *Heliyon* 5(8), e02145. https://doi.org/10.1016/j.heliyon.2019.e02145.

Hayes, T.B., Anderson, L.L., Beasley, V.R., De Solla, S.R., Iguchi, T., Ingraham, H. *et al*. (2011). Demasculinization and feminization of male gonads by atrazine: Consistent effects across vertebrate classes. *Journal of Steroid Biochemistry and Molecular Biology*, 127(1-2), 64-73. https://doi.org/10.1016/j.jsbmb.2011.03.015.

Hayes. T.B., Falso, P., Gallipeau, S. and Stice, M. (2010). The cause of global amphibian declines: A developmental endocrinologist's perspective. *Journal of Experimental Biology* 213, 921-933. https://doi.org/10.1242/jeb.040865.

Health Council of the Netherlands (2020). *Follow-up Advisory Report on Crop Protection and Local Residents.* The Hague The Hague: Health Council of the Netherlands. Full report in Dutch: https://www.gezondheidsraad.nl/documenten/adviezen/2020/06/29/vervolgadvies-gewasbescherming-en-omwonenden. Executive summary in English: https://www.healthcouncil.nl/documents/advisory-reports/2020/06/29/follow-up-advisory-report-on-crop-protection-and-local-residents.

Heap, I. (2019). The international herbicide-resistant weed database. www.weedscience.com. Accessed 20 December 2020.

Hearn, S. and Buffardi, A.L. (2016). *What is Impact?* Methods Lab. London: Overseas Development Institute. https://cdn.odi.org/media/documents/10352.pdf.

Henao, S. and Arbelaez, M.P. (2002). Epidemiological situation of acute pesticide poisoning in Central America, 1992-2000. *Epidemiological Bulletin* 23(3), 5-9. http://www1.paho.org/english/sha/EB_v23n3.pdf?ua=1.

Hertz-Picciotto, I., Sass, J.B., Engel, S., Bennett, D.H., Bradman, A., Eskenazi, B. *et al*. (2018). Organophosphate exposures during pregnancy and child neurodevelopment: Recommendations for essential policy reforms. *PLoS Medicine* 15(10), e1002671. https://doi.org/10.1371/journal.pmed.1002671.

Hill, M.P., MacFadyen, S. and Nash, M.A. (2017). Broad spectrum pesticide application alters natural enemy communities and may facilitate secondary pest outbreaks. *PeerJ* 5, e4179. https://doi.org/10.7717/peerj.4179.

Hladik, M.L. and Kolpin, D.W. (2016). First national-scale reconnaissance of neonicotinoid insecticides in streams across the USA. *Environmental Chemistry* 13(1), 12-20. https://doi.org/10.1071/EN15061.

Hladik, M.L., Main, A.R. and Goulson, D. (2018). Environmental risks and challenges associated with neonicotinoid insecticides. *Environmental Science and Technology* 52(6), 3329-3335. https://doi.org/10.1021/acs.est.7b06388.

Huang, F. (2020). Resistance of the fall armyworm, *Spodoptera frugiperda,* to transgenic *Bacillus thuringiensis* Cry1F corn in the Americas: lessons and implications for Bt corn IRM in China. *Insect Science* 28(3), 574-589. https://doi.org/10.1111/1744-7917.12826.

Huang, F., Andow, D.A. and Buschman, L.L. (2011). Success of the high-dose/refuge resistance management strategy after 15 years of Bt crop use in North America. *Entomologia Experimentalis et Applicata* 140, 1-16. https://doi.org/10.1111/j.1570-7458.2011.01138.x.

Huang, X., Cui, H. and Duan, W. (2020). Ecotoxicity of chlorpyrifos to aquatic organisms: A review. *Ecotoxicology and Environmental Safety* 200, 110731. https://doi.org/10.1016/j.ecoenv.2020.110731.

Huguier, P., Manier, N., Owojori, O.J., Bauda, P., Pandard, P. and Römbke, J. (2015). The use of soil mites in ecotoxicology: A review. *Ecotoxicology* 24(1), 1-18. https://doi.org/10.1007/s10646-014-1363-y.

Human Rights Watch (2018). *"You Don't Want to Breathe Poison Anymore": The Failing Response to Pesticide Drift in Brazil's Rural Communities*. https://www.hrw.org/sites/default/files/report_pdf/brazil0718_web.pdf.

Humann-Guilleminot, S, Binkowski, Ł.J., Jenni, L., Hilke, G., Glauser, G. and Helfenstein, F. (2019). A nation-wide survey of neonicotinoid insecticides in agricultural land with implications for agri-environment schemes. *Journal of Applied Ecology* 56(7), 1502-1514. https://doi.org/10.1111/1365-2664.13392.

Hvězdová, M., Kosubová, P., Košíková, M., Scherr, K.E., Šimek, Z., Brodský, L. *et al.* (2018). Currently and recently used pesticides in Central European arable soils. *Science of The Total Environment* 613-614, 361-370. https://doi.org/10.1016/j.scitotenv.2017.09.049.

Imfeld, G. and Vuilleumier, S. (2012). Measuring the effects of pesticides on bacterial communities in soil: A critical review. *European Journal of Soil Biology* 49, 22-30. https://doi.org/10.1016/j.ejsobi.2011.11.010.

Indian Council of Agriculture Research (ICAR) (n.d.). All India Network Project on Pesticide Residues (AINP-PR). Monitoring of pesticide residues. https://aicrp.icar.gov.in/pesticide/monitoring-of-pesticide-residues. Accessed 15 December 2020.

Institut national de la santé et de la recherche médicale (Inserm) (France) (2013). *Pesticides – effets sur la santé. Collection expertise collective*. http://www.ipubli.inserm.fr/handle/10608/4819.

Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais (Brazil) (2009). *Manual Para Requerimento de Avaliação Ambiental: Agrotóxicos e Afins. Diretoria de Qualidade Ambiental*. Brasília. http://www.leb.esalq.usp.br/disciplinas/Casimiro/LFN/Leitura_complementar/Manual_de_procedimento%20IBAMA.pdf.

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (2019). *The Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services: Summary for Policymakers*. S. Díaz, J. Settele, E. S. Brondízio E.S., H. T. Ngo, M. Guèze, J. Agard *et al*. (eds.). IPBES Secretariat, Bonn. https://www.ipbes.net/sites/default/files/2020-02/ipbes_global_assessment_report_summary_for_policymakers_en.pdf.

International Agency for Research on Cancer (2020a). IARC Monographs on the Identification of Carcinogenic Hazards to Humans. https://monographs.iarc.fr/monographs-available/. Accessed 1 December 2020.

International Agency for Research on Cancer (2020b). AGRICOH: A Consortium of Agricultural Cohort Studies. https://agricoh.iarc.fr/. Accessed 15 December 2020.

International Organization for Biological and Integrated Control – West Palearctic Regional Section (2019). IOBC-WPRS Pesticide Side Effect Database. https://www.iobc-wprs.org/ip_integrated_production/Pesticide_Side_Effect_Database.html#:~:text=The%20database%20on%20selectivity%20of,in%20the%20choice%20of%20pesticides. Accessed 20 December 2020.

International Service for the Acquisition of Agri-Biotech Applications (2019). GM approval database. http://www.isaaa.org/gmapprovaldatabase/default.asp. Accessed 20 December 2020.

International Union for Conservation of Nature (2020). IUCN Red List – Summary Statistics. https://www.iucnredlist.org/resources/summary-statistics. Accessed 20 December 2020.

Ito, H.C., Shiraishi, H., Nakagawa, M. and Takamura, N. (2020). Combined impact of pesticides and other environmental stressors on animal diversity in irrigation ponds. *PLoS One* 15(7), e0229052, https://doi.org/10.1371/journal.pone.0229052.

Ittner, L.D., Junghans, M. and Werner, I. (2018). Aquatic fungi: A disregarded trophic level in ecological risk assessment of organic fungicides. *Frontiers in Environmental Science* 6(105). https://doi.org/10.3389/fenvs.2018.00105.

Jahn, T., Hötker, H., Oppermann, R., Bleil, R. and Vele, L. (2014). *Protection of Biodiversity of Free Living Birds and Mammals in Respect of the Effects of Pesticides*. Dessau-Roßlau: Federal Environment Agency. https://www.umweltbundesamt.de/en/publikationen/protection-of-biodiversity-of-free-living-birds.

Jänsch, S., Frampton, G.K., Römbke, J., Van Den Brink, P.J. and Scott-Fordsmand, J.J. (2006). Effects of pesticides on soil invertebrates in model ecosystems and field studies: A review and comparison with laboratory toxicity data. *Environmental Toxicology and Chemistry* 25(9), 2480-2489. https://doi.org/10.1897/05-438R.1.

Jansen, J.-P. (2013). Pest select database: a new tool to use selective pesticides for IPM. *Communications in Agricultural and Applied Biological Sciences*78(2), 115-119. https://www.cra.wallonie.be/en/pest-select-database-a-new-tool-to-use-selective-pesticides-for-ipm-2013.

Jensen, B.H., Petersen, A., Nielsen, E., Christensen, T., Poulsen, M.E. and Andersen, J.H. (2015). Cumulative dietary exposure of the population of Denmark to pesticides. *Food and Chemical Toxicology* 83, 300-307. https://doi.org/10.1016/j.fct.2015.07.002.

Jeyaratnam, J. (1990). Acute pesticide poisoning: A major global health problem. *World Health Statistics Quarterly* 43(3), 139-144. https://apps.who.int/iris/handle/10665/51746.

Jørs, E. (2016). *Acute Pesticide Poisoning among Bolivian Small-holder Farmers – Frequency, Risk Factors and Prevention*. Odense: University of Southern Denmark. https://www.sdu.dk/-/media/files/om_sdu/institutter/ist/sundhedsfremme/erikphd.pdf?la=enandhash=8C6152C8C762A6CCB7E1C426C3A6E2FAA23A2255.

Jørs, E., Neupane, D. and London, L. (2018). Pesticide poisonings in low and middle-income countries. *Environmental Health Insights* 12, 1-3. https://doi.org/10.1177/1178630217750876.

Jurado, A., Vàzquez-Suñé, E., Carrera, J., López de Alda, M., Pujades, E. and Barceló, D. (2012). Emerging organic contaminants in groundwater in Spain: A review of sources, recent occurrence and fate in a European context. *Science of The Total Environment* 440, 82-94. https://doi.org/10.1016/j.scitotenv.2012.08.029.

Jurado, A., Walther, M. and Díaz-Cruz, M.S. (2019). Occurrence, fate and environmental risk assessment of the organic microcontaminants included in the Watch Lists set by EU Decisions 2015/495 and 2018/840 in the groundwater of Spain. *Science of the Total Environment* 663, 285-296. https://doi.org/10.1016/j.scitotenv.2019.01.270.

K'oreje, K.O., Okoth, M., Van Langenhove, H. and Demeestere, K. (2020). Occurrence and treatment of contaminants of emerging concern in the African aquatic environment: Literature review and a look ahead. *Journal of Environmental Management* 254. https://doi.org/10.1016/j.jenvman.2019.109752.

Kamel, F., Engel, L.S., Gladen, B.C., Hoppin, J.A., Alavanja, M.C.R. and Sandler, D.P. (2007). Neurologic symptoms in licensed pesticide applicators in the Agricultural Health Study. *Human and Experimental Toxicology* 26, 243-250. https://doi.org/10.1177/0960327107070582.

Karalexi, M.A., Tagkas C.F., Markozannes, G., Tseretopoulou, X., Hernández, A.F., Schüz, J. *et al.* (2021). Exposure to pesticides and childhood leukemia risk: A systematic review and meta-analysis. *Environmental Pollution* 285(117376), 1-12. https://doi.org/10.1016/j.envpol.2021.117376.

Karki, A.J., Cappelli, P., Dirks, C., Pekar, H., Hellenäs, K.E., Rosén, J. and Westerberg, E. (2020). New efficient methodology for screening of selected organic micropollutants in raw- and drinking water from 90 Swedish water treatment plants. *Science of The Total Environment* 724, 138069. https://doi.org/10.1016/j.scitotenv.2020.138069.

Karunarathne, A., Gunnell, D., Konradsen, F. and Eddleston, M. (2019). How many premature deaths from pesticide poisoning have occurred since the agricultural Green Revolution? *Clinical Toxicology* 58(4), 227-232. https://doi.org/10.1080/15563650.2019.1662433.

Kennedy, C.M., Lonsdorf, E., Neel, M.C., Williams, N.M., Ricketts, T.H., Winfree, R. *et al*. (2013). A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. *Ecology Letters* 16(5), 584-599. https://doi.org/10.1111/ele.12082.

Kim, J.-H., Kim, J., Cha, E.S., Ko, Y., Kim, D.H. and Lee, W.J. (2013). Work-related risk factors by severity for acute pesticide poisoning among male farmers in South Korea. *International Journal of Environmental Research and Public Health* 10, 1100-1112. https://doi.org/10.3390/ijerph10031100.

Kirchner, M., Jakobi, G., Körner, W., Levy, W., Moche, W., Niedermoser, B. *et al.* (2016). Ambient air levels of organochlorine pesticides at three high alpine monitoring stations: Trends and dependencies on geographical origin. *Aerosol and Air Quality Research* 16(3), 738-751. https://doi.org/10.4209/aaqr.2015.04.0213.

Klok, C., de Vries, P., Jongbloed, R. and Tamis, J. (2017). Literature review on the sensitivity and exposure of marine and estuarine organisms to pesticides in comparison to corresponding fresh water species. *EFSA Supporting Publications* 9(11), 357E. https://doi.org/10.2903/sp.efsa.2012.en-357.

Knauer, K. (2016). Pesticides in surface waters: A comparison with regulatory acceptable concentrations (RACs) determined in the authorization process and consideration for regulation. *Environmental Sciences Europe* 28(13). 1-7. https://doi.org/10.1186/s12302-016-0083-8.

Ko, Y., Kim, H.J., Cha, E.S., Kim, J. and Lee, W.J. (2012). Emergency department visits due to pesticide poisoning in South Korea, 2006-2009. *Clinical Toxicology* 50(2), 114-119. https://doi.org/10.3109/15563650.2012.655281.

Köck-Schulmeyer, M., Ginebreda, A., Postigo, C., Garrido, T., Fraile, J., López de Alda, M., and Barceló, D. (2014). Four-year advanced monitoring program of polar pesticides in groundwater of Catalonia (NE-Spain). *Science of the Total Environment* 470-471, 1087-1098. https://doi.org/10.1016/j.scitotenv.2013.10.079.

Köhler, H.-R. and Triebskorn, R. (2013). Wildlife ecotoxicology of pesticides: Can we track effects to the population level and beyond? *Science* 341(6147), 759-765. https://doi.org/10.1126/science.1237591.

Kohlschmid, E. and Ruf, D. (2016). Is the risk for soil arthropods covered by new data requirements under the EU PPP Regulation No. 1107/2009? *Environmental Science and Pollution* Research 23(23), 23884-23891. https://doi.org/10.1007/s11356-016-7536-x.

Komárek M., Čadková, E., Chrastný, V., Bordas, F. and Bollinger, J.C. (2010). Contamination of vineyard soils with fungicides: A review of environmental and toxicological aspects. *Environment International* 36(1), 138-151. https://doi.org/10.1016/j.envint.2009.10.005.

Kouzayha, A., Al Ashi, A., Al Akoum, R., Al Iskandarani, M., Budzinski, H. and Jaber, F. (2013). Occurrence of pesticide residues in Lebanon's water resources. *Bulletin of Environmental Contamination and Toxicology* 91(5), 503-509. https://doi.org/10.1007/s00128-013-1071-y.

Kovács-Hostyánszki A., Li, J., Pettis, J., Settele, J., Aneni, T., Espíndola, A. *et al.* (2016). Drivers of change of pollinators, pollination networks and pollination. In *The Assessment Report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on Pollinators, Pollination and Food Production*. Potts, S.G., Imperatriz-Fonseca, V.L. and Ngo, H.T. (eds.). Bonn: Secretariat of the Intergovernmental Science- Policy Platform on Biodiversity and Ecosystem Services. Chapter 2, 27-149. https://ipbes.net/assessment-reports/pollinators.

Krauss, J., Gallenberger, I. and Steffan-Dewenter, I. (2011). Decreased functional diversity and biological pest control in conventional compared to organic crop fields. *PLoS ONE*, 6(5), 1–9. https://doi.org/10.1371/journal.pone.0019502.

Krieger, R. (2010). *Hayes' Handbook of Pesticide Toxicology, Volumes 1 and 2.* 3rd edition. Krieger, R., Doull, J., van Hemmen, J., Hodgson, E., Maibach, H., Reiter, L., Ritter *et al*. (eds.). London: Academic Press. https://www.elsevier.com/books/hayes-handbook-of-pesticide-toxicology/krieger/978-0-08-092201-0.

Kurenbach, B., Marjoshi, D., Amábile-Cuevas, C.F., Ferguson, G.C., Godsoe, W., Gibson, P. and Heinemann, J.A. (2015). Sublethal exposure to commercial formulations of the herbicides dicamba, 2,4-dichlorophenoxyacetic acid, and glyphosate cause changes in antibiotic susceptibility in *Escherichia coli* and *Salmonella enterica* serovar *Typhimurium*. *mBio* 6(2), e00009-15. https://doi.org/10.1128/mBio.00009-15.

La Merrill, M.A., Vandenberg, L.N., Smith, M.T., Goodson, W., Browne, P., Patisaul, H.B. *et al.* (2020). Consensus on the key characteristics of endocrine-disrupting chemicals as a basis for hazard identification. *Nature Reviews Endocrinology* 16, 45-57 https://doi.org/10.1038/s41574-019-0273-8.

Landrigan, P.J., Fuller, R., Acosta, N.J.R., Adeyi, O., Arnold, R., Basu, N. *et al*. (2018). The *Lancet* Commission on pollution and health. *Lancet* 391, 462-512. http://dx.doi.org/10.1016/S0140-6736(17)32345-0.

Larsson, M.O., Sloth Nielsen, V., Bjerre, N., Laporte, F. and Cedergreen, N. (2018). Refined assessment and perspectives on the cumulative risk resulting from the dietary exposure to pesticide residues in the Danish population. *Food and Chemical Toxicology* 111, 207-267. https://doi.org/10.1016/j.fct.2017.11.020.

Leather, S.R. (2018). "Ecological Armageddon" – more evidence for the drastic decline in insect numbers. *Annals of Applied Biology* 172(1), 1-3. https://doi.org/10.1111/aab.12410.

Lee, S.-J., Mehler, L., Beckman, J., Diebolt-Brown, B., Prado, J., Lackovic, M. *et al*. (2011). Acute pesticide illnesses associated with off-target pesticide drift from agricultural applications: 11 states, 1998-2006. *Environmental Health Perspectives* 119(8), 1162-1169 https://doi.org/10.1289/ehp.1002843.

Lee, Y.Y., Chisholm, D., Eddleston, M., Gunnell, D., Fleischmann, A., Konradsen, F. *et al.* (2020). The cost-effectiveness of banning highly hazardous pesticides to prevent suicides due to pesticide self-ingestion across 14 countries: An economic modelling study. *The Lancet Global Health*. https://doi.org/10.1016/S2214-109X(20)30493-9.

Lehmann, E., Turrero, N., Kolia, M., Konaté, Y. and de Alencastro, L. F. (2017). Dietary risk assessment of pesticides from vegetables and drinking water in gardening areas in Burkina Faso. *Science of The Total Environment* 601-602, 1208-1216. https://doi.org/10.1016/j.scitotenv.2017.05.285.

Lekei, E., Ngowi, A.V., Kapeleka, J. and London, L. (2020). Acute pesticide poisoning amongst adolescent girls and women in northern Tanzania. *BMC Public Health* 20(303), 1-8. https://doi.org/10.1186/s12889-020-8374-9.

Lekei, E., Ngowi, A.V. and London, L. (2014). Hospital-based surveillance for acute pesticide poisoning caused by neurotoxic and other pesticides in Tanzania. *NeuroToxicology* 45, 318-326. http://dx.doi.org/10.1016/j.neuro.2014.02.007.

Lekei. E., Ngowi, A.V. and London, L. (2016). Underreporting of acute pesticide poisoning in Tanzania: Modelling results from two cross-sectional studies. *Environmental Health* 15(118). https://doi.org/10.1186/s12940-016-0203-3.

Lettoof, D.C., Lohr, M.T., Busetti, F., Bateman, P.W. and Davis, R.A. (2020). Toxic time bombs: Frequent detection of anticoagulant rodenticides in urban reptiles at multiple trophic levels. *Science of The Total Environment* 724, 138218. https://doi.org/10.1016/j.scitotenv.2020.138218.

Leusch, F.D.L., Neale, P.A., Arnal, C., Aneck-Hahn, N.H., Balaguer, P., Bruchet, A. *et al.* (2018). Analysis of endocrine activity in drinking water, surface water and treated wastewater from six countries. *Water Research* 139, 1018. https://doi.org/10.1016/j.watres.2018.03.056.

Levillain, J., Cattan, P., Colin, F., Voltz, M. and Cabidoche, Y.-M. (2012). Analysis of environmental and farming factors of soil contamination by a persistent organic pollutant, chlordecone, in a banana production area of French West Indies. *Agriculture, Ecosystems and Environment* 159, 123-132. https://doi.org/10.1016/j.agee.2012.07.005.

Levine, R.S. (1986). *Assessment of Mortality and Morbidity due to Unintentional Pesticide Poisonings.* Document WHO/VBC/86.929. Geneva: World Health Organization. https://apps.who.int/iris/bitstream/handle/10665/59938/WHO-VBC-86.929-eng.pdf;sequence=1.

Li, H., Cheng, F., Wei, Y., Lydy, M.J. and You, J. (2017). Global occurrence of pyrethroid insecticides in sediment and the associated toxicological effects on benthic invertebrates: An overview. *Journal of Hazardous Materials* 324 (Part B), 258-271. https://doi.org/10.1016/j.jhazmat.2016.10.056.

Li, J., Li, F. and Liu, Q. (2015). Sources, concentrations and risk factors of organochlorine pesticides in soil, water and sediment in the Yellow River estuary. *Marine Pollution Bulletin* 100(1), 516-522. https://doi.org/10.1016/j.marpolbul.2015.09.003.

Li, Y., Miao, R. and Khanna, M. (2020). Neonicotinoids and decline in bird biodiversity in the United States. *Nature* Sustainability 3, 1027-1035. https://www.nature.com/articles/s41893-020-0582-x.

Li, Y.F. and Macdonald, R.W. (2005). Sources and pathways of selected organochlorine pesticides to the Arctic and the effect of pathway divergence on HCH trends in biota: A review. *Science of The Total Environment* 342(1-3), 87-106. https://doi.org/10.1016/j.scitotenv.2004.12.027.

Liberati, A., Altman, D.G., Tetzlaff, J., Mulrow, C., Gøtzsche, P.C., Ioannidis, J.P.A. *et al*. (2009). The PRISMA Statement for reporting systematic reviews and meta-analyses of studies that evaluate health care interventions: Explanation and elaboration. *PLoS Medicine* 6(7), e1000100. https://doi.org/10.1371/journal.pmed.1000100.

Litchfield, M.H. (2005). Estimates of acute pesticide poisoning in agricultural workers in less developed countries. *Toxicological Reviews* 24(4), 271-271. https://doi.org/10.2165/00139709-200524040-00006.

Liu, Z., Huang, R., Roberts, R. and Tong W. (2019). Toxicogenomics: A 2020 Vision. *Trends in Pharmacological Sciences* 40(2), P93-P103. https://doi.org/10.1016/j.tips.2018.12.001.

Lohr, M.T. and Davis, R.A. (2018). Anticoagulant rodenticide use, non-target impacts and regulation: A case study from Australia. *Science of The Total Environment* 634, 1372-1384. https://doi.org/10.1016/j.scitotenv.2018.04.069.

London, L. (2003). Human rights, environmental justice, and the health of farm workers in South Africa. *International Journal of Occupational and Environmental Health* 9, 59-68. https://doi/10.1179/107735203800328876.

London, L. and Bailie, R. (2001). Challenges for improving surveillance for pesticide poisoning: Policy implications for developing countries. *International Journal of Epidemiology* 30(3), 564-570. https://doi.org/10.1093/ije/30.3.564.

Lopez, B., Ollivier, P., Togola, A., Baran, N., and Ghestem, J.P. (2015). Screening of French groundwater for regulated and emerging contaminants. *Science of The Total Environment* 518-519, 562-573. https://doi.org/10.1016/j.scitotenv.2015.01.110.

López-Gálvez, N., Wagoner, R., Quirós-Alcalá, L., Van Horne, Y.O., Furlong, M., Avila, E. and Beamer, P. (2019). Systematic literature review of the take-home route of pesticide exposure via biomonitoring and environmental monitoring. *International Journal of Environmental Research and Public Health* 16, 2177. https://doi.org/10.3390/ijerph16122177.

López-Perea, J.J. and Mateo, R. (2018). Secondary exposure to anticoagulant rodenticides and effects on predators. In *Anticoagulant Rodenticides and Wildlife. Emerging Topics in Ecotoxicology (Principles, Approaches and Perspectives) vol 5*. van den Brink, N.W., Elliott, J.E., Shore, R.F. and Rattner, B.A. (eds.). Cham: Springer. Chapter 7, 159-193. https://link.springer.com/chapter/10.1007/978-3-319-64377-9_7.

Łozowicka, B., Kaczyński, P., Wolejko, E., Piekutin, J., Sagitov, A., Toleubayev, K. *et al.* (2016). Evaluation of organochlorine pesticide residues in soil and plants from east Europe and central Asia. *Desalination and Water Treatmen*t 57(3), 1310-1321. https://doi.org/10.1080/19443994.2014.996008.

Lucas, J.A., Hawkins, N.J. and Fraaije, B.A. (2015). Chapter Two – The evolution of fungicide resistance. *Advances in Applied Microbiology* 90, 29-92. http://dx.doi.org/10.1016/bs.aambs.2014.09.001.

Lundin, O., Rundlöf, M., Smith, H.G., Fries, I. and Bommarco, R. (2015). Neonicotinoid insecticides and their impacts on bees: A systematic review of research approaches and identification of knowledge gaps. *PLoS ONE* 10(8), e0136928. https://doi.org/10.1371/journal.pone.0136928.

Machete, M. and Shadung, J. M. (2019). Detection of selected agricultural pesticides in river and tap water in Letsitele, Lomati and Vals–Renoster catchments, South Africa. *Water SA* 45(4), 716–720. https://doi.org/10.17159/wsa/2019.v45.i4.7554.

Magalhães, A.F.A. and Caldas, E.D. (2018). Underreporting of fatal poisonings in Brazil – A descriptive study using data from four information systems. *Forensic Science International* 287, 136-141. https://doi.org/10.1016/j.forsciint.2018.03.040.

Maier, L., Pruteanu, M., Kuhn, M., Zeller, G., Telzerow, A., Anderson, E.E. *et al.* (2018). Extensive impact of non-antibiotic drugs on human gut bacteria. *Nature* 555, 623-628. https://doi.org/10.1038/nature25979.

Main, A.R., Webb, E.B., Goyne, K.W. and Mengel, D. (2020). Reduced species richness of native bees in field margins associated with neonicotinoid concentrations in non-target soils. *Agriculture, Ecosystems and Environment* 287, 106693. https://doi.org/10.1016/j.agee.2019.106693.

Malaj, E., von der Ohe, P.C., Grote, M., Kühne, R., Mondy, C.P., Usseglio-Polatera, P. *et al*. (2014). Organic chemicals jeopardize the health of freshwater ecosystems on the continental scale. *Proceedings of the National Academy of Sciences of the United States of America* 111(26), 9549-9554. https://doi.org/10.1073/pnas.1321082111.

Mallinger, R.E., Werts, P. and Gratton, C. (2015). Pesticide use within a pollinator-dependent crop has negative effects on the abundance and species richness of sweat bees, *Lasioglossum* spp., and on bumble bee colony growth. *Journal of Insect Conservation* 19(5), 999-1010. https://doi.org/10.1007/s10841-015-9816-z.

Maltby, L., Blake, N., Brock, T.C.M. and Van Den Brink, P.J. (2005). Insecticide species sensitivity distributions: Importance of test species selection and relevance to aquatic ecosystems. *Environmental Toxicology and Chemistry* 24, 379-388. https://doi.org/10.1897/04-025R.1.

Maltby, L., Brock, T.C.M. and Van Den Brink, P.J. (2009). Fungicide risk assessment for aquatic ecosystems: Importance of interspecific variation, toxic mode of action, and exposure regime. *Environmental Science and Technology* 43(19), 7556-7563. https://doi.org/10.1021/es901461c.

Malyan, S.K., Singh, R., Rawat, M., Kumar, M., Pugazhendhi, A., Kumar, A. *et al.* (2019). An overview of carcinogenic pollutants in groundwater of India. *Biocatalysis and Agricultural Biotechnology* 21, 101288. https://doi.org/10.1016/j.bcab.2019.101288.

Manamsa, K., Crane, E., Stuart, M., Talbot, J., Lapworth, D. and Hart, A. (2016). A national-scale assessment of micro-organic contaminants in groundwater of England and Wales. *Science of The Total Environment* 568, 712-726. https://doi.org/10.1016/j.scitotenv.2016.03.017.

Manuweera, G., Eddleston, M., Egodage, S. and Buckley, N. (2008). Do targeted bans of insecticides to prevent deaths from self-poisoning result in reduced agricultural output? *Environmental Health Perspectives* 116, 492-495. https://doi.org/10.1289/ehp.11029.

Martin, O., Scholze, M., Ermler, S., McPhie, J., Bopp, S.K., Kienzler, A. *et al*. (2021). Ten years of research on synergisms and antagonisms in chemical mixtures: A systematic review and quantitative reappraisal of mixture studies. *Environmental International* 146, 106206. https://doi.org/10.1016/j.envint.2020.106206.

Martyniuk, C.J., Mehinto, A.C. and Denslow, N.D. (2020). Organochlorine pesticides: Agrochemicals with potent endocrine-disrupting properties in fish. *Molecular and Cellular Endocrinology* 507, 110764. https://doi.org/10.1016/j.mce.2020.110764.

Matthiessen, P, Wheeler, J.R. and Weltje, L. (2018). A review of the evidence for endocrine disrupting effects of current-use chemicals on wildlife populations. *Critical Reviews in Toxicology* 48(3), 195-216. https://doi.org/10.1080/10408444.2017.1397099.

Maute, K., French, K., Bull, C.M., Story, P. and Hose, G. (2015). Current insecticide treatments used in locust control have less of a short-term impact on Australian arid-zone reptile communities than does temporal variation. *Wildlife Research* 42(1), 50-59. http://dx.doi.org/10.1071/WR14194.

McManus, S.L., Richards, K.G., Grant, J., Mannix, A., and Coxon, C.E. (2014). Pesticide occurrence in groundwater and the physical characteristics in association with these detections in Ireland. *Environmental Monitoring and Assessment* 186(11), 7819-7836. https://doi.org/10.1007/s10661-014-3970-8.

Meek, M.E., Boobis, A.R., Crofton, K.M., Heinemeyer, G., van Raaij, M. and Vickers C (2011). Risk assessment of combined exposure to multiple chemicals: A WHO/IPCS framework. *Regulatory Toxicology and Pharmacology* 60, S1-S14. https://doi.org/10.1016/j.yrtph.2011.03.010.

Meffe, R. and de Bustamante, I. (2014). Emerging organic contaminants in surface water and groundwater: A first overview of the situation in Italy. *Science of The Total Environment* 481 (1), 280-295. https://doi.org/10.1016/j.scitotenv.2014.02.053.

Menzies, R., Soares Quinete, N., Gardinali, P. and Seba, D. (2013). Baseline occurrence of organochlorine pesticides and other xenobiotics in the marine environment: Caribbean and Pacific collections. *Marine Pollution Bulletin* 70 (1-2), 289-295. https://doi.org/10.1016/j.marpolbul.2013.03.003.

Mew, E.J., Padmanathan, P., Konradsen, F., Eddleston, M., Chang, S.-S., Phillips, M.R. and Gunnell, D. (2017). The global burden of fatal self-poisoning with pesticides 2006-15: Systematic review. *Journal of Affective Disorders* 219, 93-104. https://doi.org/10.1016/j.jad.2017.05.002.

Millennium Ecosystem Assessment (2005). *Ecosystems and Human Well-being: Biodiversity Synthesis*, Washington, D.C. World Resources Institute. https://www.millenniumassessment.org/documents/document.354.aspx.pdf.

Milner, A.M. and Boyd, I.L. (2017). Toward pesticidovigilance – Can lessons from Pharmaceutical monitoring help to improve pesticide regulation? *Science* 357(6357), 1232-1234. https://doi.org/10.1126/science.aan2683.

Mineau P (2013a). Bird impacts from pesticide use. In *Encyclopedia of Environmental Management vol 1*. Jørgensen, S.E. (ed.). Boca Raton, FL: CRC Press. 416-423. https://www.researchgate.net/publication/260338687_Bird_impacts_from_pesticide_use_Encyclopedia_of_Environmental_Management_1st_edition_SE_Jorgensen_ed_DOI_101081E-EEM-120046191_Pages_416-423_Taylor_and_Francis_New_York.

Mineau, P. (2013b). Avian mortality from pesticides used in agriculture in Canada. Unpublished manuscript, provided as an annex to Calvert *et al.* (2013). https://www.researchgate.net/publication/278212489_Avian_mortality_from_pesticides_used_in_agriculture_in_Canada.

Mineau, P. and Whiteside, M. (2006). Lethal risk to birds form insecticide use in the United States – a spatial and temporal analysis. *Environmental Toxicology and Chemistry* 25(5), 1214-1222. https://pubmed.ncbi.nlm.nih.gov/16704051/.

Mineau, P. and Whiteside, M. (2013). Pesticide acute toxicity Is a better correlate of U.S. grassland bird declines than agricultural intensification. *PLoS ONE* 8(2), e57457. https://doi.org/doi:10.1371/journal.pone.0057457.

Mingo, V., Lötters, S. and Wagner, N. (2016). Risk of pesticide exposure for reptile species in the European Union. *Environmental Pollution* 215, 164e169. http://dx.doi.org/10.1016/j.envpol.2016.05.011.

Mingo, V., Lötters, S. and Wagner, N. (2017). The impact of land use intensity and associated pesticide applications on fitness and enzymatic activity in reptiles—A field study. *Science of The Total Environment* 590-591, 114-124. http://dx.doi.org/10.1016/j.scitotenv.2017.02.178.

Ministry of Agriculture and Farmers Welfare (MAFW) (India) (2019). *Survey Report: Status of Pesticide Residues in India*. https://fssai.gov.in/upload/advisories/2019/10/5da705b31ca78Letter_Report_Pesticides_MRL_16_10_2019.pdf.

Mishra, K., Sharma, R.C. and Kumar, S. (2012). Contamination levels and spatial distribution of organochlorine pesticides in soils from India. *Ecotoxicology and Environmental Safety* 76(1), 215-225. https://doi.org/10.1016/j.ecoenv.2011.09.014.

Mochungong, P. and Zhu, J. (2015). DDTs, PCBs and PBDEs contamination in Africa, Latin America and South-southeast Asia—a review. *AIMS Environmental Science* 2(2), 374–399. https://doi.org/10.3934/environsci.2015.2.374.

Mohaupt, V., Völker, J., Altenburger, R., Birk, S., Kirst, I., Kühnel, D. *et al.* (2020). *Pesticides in European Rivers, Lakes and Groundwaters – Data Assessment.* ETC/ICM Technical Report 1/2020. https://www.eionet.europa.eu/etcs/etc-icm/products/etc-icm-report-1-2020-pesticides-in-european-rivers-lakes-and-groundwaters-data-assessment.

Mohr, S., Berghahn, R., Feibicke, M., Meinecke, S., Ottenströer, T., Schmiedling, I. *et al.* (2007). Effects of the herbicide metazachlor on macrophytes and ecosystem function in freshwater pond and stream mesocosms. *Aquatic Toxicology* 82(1), 73-84. https://doi.org/10.1016/j.aquatox.2007.02.001.

Mohr, S., Feibicke, M., Berghahn, R., Schmiediche, R. and Schmidt, R. (2008). Response of plankton communities in freshwater pond and stream mesocosms to the herbicide metazachlor. *Environmental Pollution* 152(3), 530-542. https://doi.org/10.1016/j.envpol.2007.07.010.

Montgomery, G.A., Dunn, R.R., Fox, R., Jongejans, E., Leather, S.R., Saunders, M.E. *et al.* (2020). Is the insect apocalypse upon us? How to find out. *Biological Conservation* 241, 108327. https://doi.org/10.1016/j.biocon.2019.108327.

Morrissey, C.A., Mineau, P., Devries, J.H., Sánchez-Bayo, F., Liess, M., Cavallaro, M.C. and Liber, K. (2015). Neonicotinoid contamination of global surface waters and associated risk to aquatic invertebrates: A review. *Environment International* 74, 291-303. https://doi.org/10.1016/j.envint.2014.10.024.

Mostafalou, S. and Abdollahi, M. (2013). Pesticides and human chronic diseases: Evidences, mechanisms, and perspectives. *Toxicology and Applied Pharmacology* 268(2), 157-177. https://doi.org/10.1016/j.taap.2013.01.025.

Mota-Sanchez, D. and Wise, J.C. (2019). Arthropod Pesticide Resistance Database. *Michigan State University*. http://www.pesticideresistance.org. Accessed 20 December 2020.

Moutinho, M.F., de Almeida, E.A., Espíndola, E.L.G., Daam, M.A. and Schiesari, L. (2020). Herbicides employed in sugarcane plantations have lethal and sublethal effects to larval *Boana pardalis* (Amphibia, Hylidae). *Ecotoxicology* 29, 1043-1051. https://doi.org/10.1007/s10646-020-02226-z.

Nakayama, S.M.M., Morita, A., Ikenaka, Y., Mizukawa, H. and Ishizuka, M. (2019). A review: Poisoning by anticoagulant rodenticides in non-target animals globally. *Journal of Veterinary Medical Science* 81(2), 298-313. https://doi.org/10.1292/jvms.17-0717.

National Academies of Sciences, Engineering and Medicine (2016). *Genetically Engineered Crops: Experiences and Prospects.* Washington D.C.: National Academies Press. https://doi.org/10.17226/23395.

National Poisons Information Service (United Kingdom) (2019). *National Poisons Information Service Report 2018/19.* https://www.npis.org/Download/NPISAnnualReport2018-19.pdf.

National Research Council (2013). *Assessing Risks to Endangered and Threatened Species from Pesticides.* Washington, D.C.: National Academies Press. https://doi.org/10.17226/18344.

Neher, D.A. and Barbercheck, M.E. (2019). Soil microarthropods and soil health: Intersection of decomposition and pest suppression in agroecosystems. *Insects* 10(10), 414. https://doi.org/10.3390/insects10120414.

Nelson, R., Wiesner-Hanks, T., Wisser, R. and Balint-Kurti, P. (2018). Navigating complexity to breed disease-resistant crops. *Nature Reviews Genetics* 19, 21-33. https://doi.org/10.1038/nrg.2017.82.

Nesser, G.A.A., Abdelbagi, A.O., Hammad, A.M.A., Tagelseed, M. and Laing, M.D. (2016). Levels of pesticides residues in the White Nile water in the Sudan. *Environmental Monitoring and Assessment* 188, 374. https://doi.org/10.1007/s10661-016-5367-3.

Netherlands Environmental Assessment Agency (2019). *Geïntegreerde gewasbescherming Nader Beschouwd – Tussenevaluatie van de Nota Gezonde Groei, Duurzame Oogst* (Integrated Pest Management Closer Considered – Interim Evaluation of the Policy for Healthy Growth and Sustainable Yield). The Hague, The Netherlands: Netherlands Environmental Assessment Agency (PBL) (in Dutch). https://www.pbl.nl/publicaties/geintegreerde-gewasbescherming-nader-beschouwd.

New Zealand Food Safety (2020). *Food Residues Survey Programme Final Report – Results for 1 July 2017-30 June 2019 for Plant Based Foods*. https://www.mpi.govt.nz/dmsdocument/40274-2017-2019-plant-based-foods-survey-final-report.

New Zealand Food Safety (n.d.). Food monitoring and surveillance. https://www.mpi.govt.nz/food-safety/food-monitoring-and-surveillance/. Accessed 26 December 2020.

North, A.C., Hodgson, D.J., Price, S.J. and Griffiths, A.G.F. (2015). Anthropogenic and ecological drivers of amphibian disease (Ranavirosis). *PLoS ONE* 10(6), e0127037. https://doi.org/10.1371/journal.pone.0127037.

Ntzani, E.E., Chondrogiorgi, M., Ntritsos, G., Evangelou, E. and Tzoulaki, I. (2013). Literature review on epidemiological studies linking exposure to pesticides and health effects. *EFSA Supporting Publications* 10(10), 497e. https://efsa.onlinelibrary.wiley.com/doi/abs/10.2903/sp.efsa.2013.EN-497.

Ntzani, E.E., Markozannes, G., Tagas, C., Rizos, E. and Karalexi, M. (2020). Epidemiological evidence review report: Exposure to pesticides and fertilizers and human health – Part A. Department of Hygiene and Epidemiology, University of Ioannina School of Medicine, Ioannina, Greece. Supporting document to this review (*Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them*).

Nuyttens, D., Devarrewaere, W., Verboven, P. and Foqué, D. (2013). Pesticide-laden dust emission and drift from treated seeds during seed drilling: A review. *Pest Management Science* 69(5), 564-575. https://www.researchgate.net/publication/235777972_Pesticide-laden_dust_emission_and_drift_from_treated_seeds_during_seed_drilling_A_review.

Nygård, T., Sandercock, B.K., Reinsborg, T. and Einvik, K. (2019). Population recovery of peregrine falcons in central Norway in the 4 decades since the DDT-ban. *Ecotoxicology* 28, 1160-1168. https://doi.org/10.1007/s10646-019-02111-4.

Odendaal, C., Seaman, M.T., Kemp, G., Patterton, H.E. and Patterton, H.G. (2015). An LC-MS/MS based survey of contaminants of emerging concern in drinking water in South Africa. *South African Journal of Science* 111(9-10). https://www.researchgate.net/publication/282313442_An_LC-MSMS_based_survey_of_contaminants_of_emerging_concern_in_drinking_water_in_South_Africa.

Ogada, D., Botha, A. and Shaw, P. (2016). Ivory poachers and poison: drivers of Africa's declining vulture populations. *Oryx* 50(4), 593-596. https://doi.org/10.1017/s0030605315001209.

Ogada, D., Shaw, P., Beyers, R.L., Buij, R., Murn, C., Thiollay, J.M. *et al.* (2016). Another continental vulture crisis: Africa's vultures collapsing toward extinction. *Conservation Letters* 9(2), 89-97 https://conbio.onlinelibrary.wiley.com/doi/full/10.1111/conl.12182.

Ogada, D.L. (2014). The power of poison: Pesticide poisoning of Africa's wildlife. *Annals of the New York Academy of Sciences* 1322, 1-20. https://www.researchgate.net/publication/261476784_The_power_of_poison_Pesticide_poisoning_of_Africa%27s_wildlife.

Ohlander, J., Fuhrimann, S., Basinas, I., Cherrie, J.W., Galea, K.S., Povey, A.C. *et al.* (2020). Systematic review of methods used to assess exposure to pesticides in occupational epidemiology studies, 1993-2017. *Occupational and Environmental Medicine* 77(6), 357-367. http://dx.doi.org/10.1136/oemed-2019-105880.

Ojemaye, C.Y., Onwordi, C.T., Pampanin, D.M., Sydnes, M.O. and Petrik, L. (2020). Presence and risk assessment of herbicides in the marine environment of Camps Bay (Cape Town, South Africa). *Science of The Total Environment* 738, 140346. https://doi.org/10.1016/j.scitotenv.2020.140346.

Oliveira Jardim, A.N., Brito, A.P., van Donkersgoed, G., Boon, P.E. and Caldas, E.D. (2018a). Dietary cumulative acute risk assessment of organophosphorus, carbamates and pyrethroids insecticides for the Brazilian population. *Food and Chemical Toxicology* 112, 108-117. https://doi.org/10.1016/j.fct.2017.12.010.

Oliveira Jardim, A.N., Mello, D.C., Brito, A.P., van der Voet, H., Boon, P.E. and Caldas, E.D. (2018b). Probabilistic dietary risk assessment of triazole and dithiocarbamate fungicides for the Brazilian population. *Food and Chemical Toxicology* 118, 317-327. https://doi.org/10.1016/j.fct.2018.05.002.

Organisation for Economic Co-operation and Development (2004). *OECD Guidelines for the Testing of Chemicals - Test No. 312: Leaching in Soil Columns*. https://www.oecd-ilibrary.org/environment/test-no-312-leaching-in-soil-columns_9789264070561-en.

Ortiz-Santaliestra, M.E., Maia, J.P., Egea-Serrano, A. and Lopes, I. (2018). Validity of fish, birds and mammals as surrogates for amphibians and reptiles in pesticide toxicity assessment. *Ecotoxicology* 27, 819-833. https://doi.org/10.1007/s10646-018-1911-y.

Orton, F. and Tyler, C.R. (2015). Do hormone-modulating chemicals impact on reproduction and development of wild amphibians? *Biological Reviews* 90, 1100-1117. https://doi.org.10.1111/brv.12147.

Osei-Fosu, P., Donkor, A.K., Nyarko, S., Nazzah, N.K., Asante, I.K., Kingsford-Adabo, R. and Arkorful, N.A. (2014). Monitoring of pesticide residues of five notable vegetables at Agbogbloshie market in Accra, Ghana. *Environmental Monitoring and Assessment* 186(11), 7157-7163. https://doi.org/10.1007/s10661-014-3917-0.

Osterman, J., Wintermantel, D., Locke, B., Jonsson, O., Semberg, E., Onorati, P. *et al.* (2019). Clothianidin seed-treatment has no detectable negative impact on honeybee colonies and their pathogens. *Nature Communications* 10, 692. https://doi.org/10.1038/s41467-019-08523-4.

Pan, L., Sun, J., Li, Z., Zhan, Y., Xu, S. and Zhu, L. (2018). Organophosphate pesticide in agricultural soils from the Yangtze River Delta of China: Concentration, distribution, and risk assessment. *Environmental Science and Pollution Research* 25(1), 4-11. https://doi.org/10.1007/s11356-016-7664-3.

Pannell, D.J., Tillie, P., Rodriguez-Cerezo, E., Ervin, D. and Frisvold, G. (2016). Herbicide resistance: Economic and environmental challenges. *AgBioForum* 19(2), 136-155. https://www.researchgate.net/publication/316669609_Herbicide_Resistance_Economic_and_Environmental_Challenges_AgBioForum_192_136-155.

Park, M.G., Blitzer, E.J., Gibbs, J., Losey, J.E. and Danforth, B.N. (2015). Negative effects of pesticides on wild bee communities can be buffered by landscape context. *Proceedings of the Royal Society B: Biological* Sciences 282, 20150299. http://dx.doi.org/10.1098/rspb.2015.0299.

Parsons, A.E., Escobar-Lux, R.H., Sævik, P.N., Samuelsen, O.B. and Agnalt, A.L. (2020). The impact of anti-sea lice pesticides, azamethiphos and deltamethrin, on European lobster (*Homarus gammarus*) larvae in the Norwegian marine environment. *Environmental Pollution* 264, 114725. https://doi.org/10.1016/j.envpol.2020.114725.

Parsons, K.C., Mineau, P. and Renfrew, R.B. (2010). Effects of pesticide use in rice fields on birds. *Waterbird*s 33 (Special Publication 1), 193-218. https://doi.org/10.1675/063.033.s115.

Pasiani, J.O., Torres, P., Silva, J.R., Diniz, B.Z. and Caldas, E.D. (2012). Knowledge, attitudes, practices and biomonitoring of farmers and residents exposed to pesticides in Brazil. *International Journal of Environmental Research and Public Health* 24(9), 3051-3068. https://doi.org/10.3390/ijerph9093051.

Peakall, D.B. (1993). DDE-induced eggshell thinning: An environmental detective story. *Environmental Reviews* 1(1),13-20. https://www.jstor.org/stable/envirevi.1.1.13?seq=1.

Peeters, F.M., Qu Mengmeng, Piao Xiuying, Valk, H. van der and Tao Chuanjiang (2014). *Environmental Risk Assessment Handbook for Pesticide Registration in China*. Alterra report 2558. Wageningen: Wageningen University and Research Centre. http://content.alterra.wur.nl/Webdocs/PDFFiles/Alterrarapporten/AlterraRapport2558.pdf.

Pelch, K.E., Beeman, J.M., Niebruegge, B.A., Winkeler, S.R. and Nagel, S.C. (2011). Endocrine-disrupting chemicals (EDCs) in mammals. *Hormones and Reproduction of Vertebrates* 5(5), 329-371. https://doi.org/10.1016/B978-0-12-374928-4.10014-8.

Pelosi, C., Barot, S., Capowiez, Y., Hedde, M. and Vandenbulcke, F. (2014). Pesticides and earthworms. A review. *Agronomy for Sustainable Development* 34(1), 199-228. https://doi.org/10.1007/s13593-013-0151-z.

Pelosi, C., Bertrand, C., Daniele, G., Coeurdassier, M., Benoit, P., Nélieu, S. *et al*. (2021). Residues of currently used pesticides in soils and earthworms: A silent threat? *Agriculture, Ecosystems and Environment* 305, 107167. https://doi.org/10.1016/j.agee.2020.107167.

Pérez-Lucas, G., Vela, N., Aatik, A. El, and Navarro, S. (2018). Environmental risk of groundwater pollution by pesticide leaching through the soil profile. In *Pesticides - Use and Misuse and Their Impact in the Environment*. Larramendy, M.L. and Soloneski, S. (eds). London: IntechOpen. Chapter 3. https://doi.org/10.5772/intechopen.82418.

Perry, L., Adams, R.D., Bennett, A.R., Lupton, D.J., Jackson, G., Good, A.M. *et al*. (2014). National toxicovigilance for pesticide exposures resulting in health care contact – An example from the UK's National Poisons Information Service. *Clinical Toxicology* 52(5), 549-555. https://doi.org/10.3109/15563650.2014.908203.

Persson, H.E., Sjöberg, G.K., Haines, J.A. and Pronczuk de Garbino, J. (1998). Poisoning severity score – Grading of acute poisoning. *Clinical Toxicology* 36(3), 205-213. https://doi.org/10.3109/15563659809028940.

Pesticide Action Network Asia Pacific (2010). *Communities in Peril – Global Report on Health Impacts of Pesticide Use in Agriculture*. http://www.panna.org/sites/default/files/PAN-Global-Report_0.pdf.

Pesticide Action Network UK (2020). *Acute Pesticide Poisoning Among Smallholder Farmers and Farmworkers – A Review of 13 Studies in EECCA and Africa*. https://www.pan-uk.org/acute-pesticide-poisoning-among-smallholder-farmers.

Peters, B., Gao, Z. and Zumkier, U. (2016). Large-scale monitoring of effects of clothianidin-dressed oilseed rape seeds on pollinating insects in Northern Germany: Effects on red mason bees (*Osmia bicornis*). *Ecotoxicology* 25, 1679-1690. https://doi.org/10.1007/s10646-016-1729-4.

Peters, K., Bundschuh, M. and Schäfer, R.B. (2013). Review on the effects of toxicants on freshwater ecosystem functions. *Environmental Pollution* 180. https://doi.org/10.1016/j.envpol.2013.05.025.

Peveling, R. and Demba, S.A. (2003). Toxicity and pathogenicity of *Metarhizium anisopliae* var. acridum (Deuteromycotina, Hyphomycetes) and fipronil to the fringe-toed lizard *Acanthodactylus dumerrili* (Squamata: Lacertidae). *Environmental Toxicology and Chemistry* 22(7), 1437-1447. https://doi.org/10.1002/etc.5620220704.

Peveling, R., McWilliam, A.N., Nagel, P., Rasolomanana, H., Raholijaona, L., Rakotomianina, A. *et al.* (2003). Impact of locust control on harvester termites and endemic vertebrate predators in Madagascar. *Journal of Applied Ecology* 40, 729-741. https://doi.org/10.1046/j.1365-2664.2003.00833.x.

Pietrzak, D., Kania, J., Malina, G., Kmiecik, E., and Wątor, K. (2019). Pesticides from the EU First and Second Watch Lists in the Water Environment. *Clean Soil, Air, Water* 47(7), 1800376. https://doi.org/10.1002/clen.201800376.

Pirsaheb, M., Hossini, H., Asadi, F. and Janjani, H. (2017). A systematic review on organochlorine and organophosphorus pesticides content in water resources. *Toxin Reviews* 36(3), 210-221. https://doi.org/10.1080/15569543.2016.1269810.

Pisa, L.W., Amaral-Rogers, V., Belzunces, L.P., Bonmatin, J.M., Downs, C.A., Goulson, D. *et al*. (2015). Effects of neonicotinoids and fipronil on non-target invertebrates. *Environmental Science and Pollution Research* 22, 68-102. https://doi.org/10.1007/s11356-014-3471-x.

Pitt, W.C., Berentsen, A.R., Shiels, A.B., Volker, S.F., Eisemann, J.D., Wegmann, A.S. and Howald, G.R. (2015). Non-target species mortality and the measurement of brodifacoum rodenticide residues after a rat (*Rattus rattus*) eradication on Palmyra Atoll, tropical Pacific. *Biological Conservation* 185, 36-46. http://dx.doi.org/10.1016/j.biocon.2015.01.008.

Plano de Nacional de Controle de Resíduos e Contaminantes (Brazil) (2018). Results of the national plan for control of residues and contaminants – PNCRC / Animal. Ministério da Agricultura, Pecuária e Abastecimento. https://www.gov.br/agricultura/pt-br/assuntos/inspecao/produtos-animal/plano-de-nacional-de-controle-de-residuos-e-contaminantes/plano-de-nacional-de-controle-de-residuos-e-contaminantes. Accessed 23 January 2020.

Plano de Nacional de Controle de Resíduos e Contaminantes (Brazil) (n.d). PNCRC / Vegetal. Ministério da Agricultura, Pecuária e Abastecimento. http://www.agricultura.gov.br/assuntos/inspecao/produtos-vegetal/copy_of_pncrc-vegetal. Accessed 23 January 2020.

Plaza, P.I., Martínez-López, E. and Lambertucci, S.A. (2019). The perfect threat: Pesticides and vultures. *Science of The Total Environment* 687, 1207-1218. https://doi.org/10.1016/j.scitotenv.2019.06.160.

Postigo, C. and Barceló, D. (2015). Synthetic organic compounds and their transformation products in groundwater: Occurrence, fate and mitigation. *Science of The Total Environment* 503-504, 32-47. https://doi.org/10.1016/j.scitotenv.2014.06.019.

Potts, S.G., Imperatriz-Fonseca, V.L. and Ngo, H.T. (eds.) (2016). *Assessment Report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on Pollinators, Pollination and Food Production*. Bonn: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://ipbes.net/assessment-reports/pollinators.

Prado, J.B., Mulay, P.R., Kasner, E.J., Bojes, H.K. and Calvert, G.M. (2017). Acute pesticide-related illness among farmworkers: Barriers to reporting to public health authorities. *Journal of Agromedicine* 22(4), 395-405. https://www.ncbi.nlm.nih.gov/pmc/articles/PMC5846675/.

Pruden, A., Larsson, D.G.J., Amézquita, A., Collignon, P., Brandt, K.K., Graham, D.W. *et al.* (2013). Management options for reducing the release of antibiotics and antibiotic resistance genes to the environment. *Environmental Health Perspectives* 121, 878-885. http://dx.doi.org/10.1289/ehp.1206446.

Prüss-Ustün, A., Wolf, J., Corvalán, C., Bos, R. and Neira, M. (2016). *Preventing Disease through Healthy Environments: A Global Assessment of the Burden of Disease from Environmental Risks*. Geneva: World Health Organization. https://www.who.int/publications/i/item/9789241565196.

Puglisi, E. (2012). Response of microbial organisms (aquatic and terrestrial) to pesticides. *EFA Supporting Publications* 9(11), 359E. https://doi.org/10.2903/sp.efsa.2012.EN-359.

Quandt, S.A., Hernández-Valero, M.A., Grzywacz, J.G., Hovey, J.D., Gonzales, M. and Arcury, T.A. (2006). Workplace, household, and personal predictors of pesticide exposure for farmworkers. *Environmental Health Perspectives* 114, 943-952. https://doi.org/10.1289/ehp.8529.

Quinn, N. (2019). Assessing individual and population-level effects of anticoagulant rodenticides on wildlife. *Human-Wildlife Interactions* 13(2), 200-211. https://doi.org/10.26077/y9tj-ew86.

Ranson, H. and Lissenden, N. (2016). Insecticide resistance in African Anopheles mosquitos: a worsening situation that needs urgent action to maintain malaria control. *Trends in Parasitology* 32(2), 187-196. http://dx.doi.org/10.1016/j.pt.2015.11.010.

Rauert, C., Harner, T., Schuster, J.K., Eng, A., Fillmann, G., Castillo, L.E. *et al.* (2018). Air monitoring of new and legacy POPs in the Group of Latin America and Caribbean (GRULAC) region. *Environmental Pollution* 243, Part B, 1252-1262. https://doi.org/10.1016/j.envpol.2018.09.048.

Rhalem, N., Chaoui, H., Semlali, I., Aoued, L., El Hattimy, F., Aghandous, R. *et al.* (2012). Exhaustivité des données de toxicovigilance au Maroc – Implication des services de santé publique et de surveillance épidémiologique. *Toxicologie Maroc* 15, 8-15. https://www.researchgate.net/publication/259467113_Exhaustivite_des_donnees_de_Toxicovigilance_au_Maroc_Implication_des_Services_de_Sante_Publique_Et_de_Surveillance_epidemiologique.

Ricci, B., Lavigne, C., Alignier, A., Aviron, S., Biju-Duval, L., Bouvier, J.C. *et al.* (2019). Local pesticide use intensity conditions landscape effects on biological pest control. *Proceedings of the Royal Society B: Biological Sciences*, 286(1904). https://doi.org/10.1098/rspb.2018.2898.

Richards, N.L., Ogada, D., Buij, R. and Botha, A. (2018). The killing fields: the use of pesticides and other contaminants to poison wildlife in africa. *Encyclopedia of the Anthropocene* 1–5, 161–167. https://doi.org/10.1016/b978-0-12-809665-9.09995-x.

Rico, A., Brock, T.C.M. and Daam, M.A. (2019). Is the effect assessment approach for fungicides as laid down in the European Food Safety Authority aquatic guidance document sufficiently protective for freshwater ecosystems? *Environmental Toxicology and Chemistry* 38, 2279-2293. https://doi.org/10.1002/etc.4520.

Rico, A., Dafouz, R., Vighi, M., Rodríguez-Gil, J.L. and Daam, M.A. (2020). Use of post registration monitoring data to evaluate the ecotoxicological risks of pesticides to surface waters: A case study with Chlorpyrifos in the Iberian Peninsula. *Environmental Toxicology and Chemistry* 40(2), 500-512. https://doi.org/10.1002/etc.4927.

Ripper. W.E. (1956). Effect of pesticides on the balance of arthropod populations. *Annual Review of Entomology* 1, 403-438. https://doi.org/10.1146/annurev.en.01.010156.002155.

Roberts, J. and Reigart, J.R. (2013). *Recognition and Management of Pesticide Poisonings.* Sixth edition. Washington, D.C.: United States Environmental Protection Agency. https://www.epa.gov/sites/production/files/2015-01/documents/rmpp_6thed_final_lowresopt.pdf.

Robinson, C., Portier, C.J., Čavoški, A., Mesnage, R., Roger, A., Clausing P. *et al.* (2020). Achieving a high level of protection from pesticides in Europe: Problems with the current risk assessment procedure and solutions. *European Journal of Risk Regulation* 11, 450-480. https://doi.org/10.1017/err.2020.18.

Rohr, J. R. and Mccoy, K. A. (2018). A qualitative meta-analysis reveals consistent effects of atrazine on freshwater fish and amphibians. *Environmental Health Perspectives* 118(1), 20-32. http://www.jstor.com/stable/30249901.

Römbke, J., Schmelz, R.M. and Pélosi, C. (2017). Effects of organic pesticides on enchytraeids (Oligochaeta) in agroecosystems: Laboratory and higher-tier tests. *Frontiers in Environmental Science* 5(20), 1-23. https://doi.org/10.3389/fenvs.2017.00020.

Rosic, N., Bradbury, J., Lee, M., Baltrotsky, K. and Grace, S. (2020). The impact of pesticides on local waterways: A scoping review and method for identifying pesticides in local usage. *Environmental Science and Policy* 106, 12-21. https://doi.org/10.1016/j.envsci.2019.12.005.

Roubos, C.R., Rodriguez-Saona, C. and Isaacs, R. (2014). Mitigating the effects of insecticides on arthropod biological control at field and landscape scales. *Biological Control* 75, 28-38. https://doi.org/10.1016/j.biocontrol.2014.01.006.

Ruddle, N., Elston, C., Klein, O., Hamberger, A. and Thompson, H. (2018). Effects of exposure to winter oilseed rape grown from thiamethoxam-treated seed on the red mason bee *Osmia bicornis*. *Environmental Toxicology and Chemistry* 37(4), 1071-1083. https://doi.org/10.1002/etc.4034.

Rusch, A., Valantin-Morison, M., Sarthou, J.-P. and Roger-Estrade, J. (2010). Biological control of insect pests in agroecosystems: Effects of crop management, farming systems, and seminatural habitats at the landscape scale: A review. In *Advances in Agronomy vol 109*. Sparks, D.L. (ed.). Academic Press. Chapter 6, 219-259. https://www.sciencedirect.com/science/article/pii/B9780123850409000062.

Sackaria, M. and Elango, L. (2020). Organic micropollutants in groundwater of India – A review. *Water Environment Research* 92(4), 504-523. https://doi.org/10.1002/wer.1243.

Sánchez-Bayo, F., Goka, K. and Hayasaka, D. (2016). Contamination of the aquatic environment with neonicotinoids and its implication for ecosystems. *Frontiers in Environmental Science* 4(71), 1-14. https://doi.org/10.3389/fenvs.2016.00071.

Sánchez-Bayo, F. and Hyne, R.V. (2011). Comparison of environmental risks of pesticides between tropical and nontropical regions. *Integrated Environmental Assessment and Management* 7(4), 577-586. https://doi.org/10.1002/ieam.189.

Sánchez-Bayo, F. and Tennekes, H.A. (2020). Time-cumulative toxicity of neonicotinoids: Experimental evidence and implications for environmental risk assessments. *International Journal of Environmental Research and Public Health* 17(5), 1629. https://doi.org/10.3390/ijerph17051629.

Sánchez-Bayo, F. and Wyckhuys, K.A.G. (2019). Worldwide decline of the entomofauna: A review of its drivers. *Biological Conservation* 232, 8-27. https://doi.org/10.1016/j.biocon.2019.01.020.

Sapbamrer, R. and Hongsibsong, S. (2014). Organophosphorus pesticide residues in vegetables from farms, markets, and a supermarket around Kwan Phayao Lake of Northern Thailand. *Archives of Environmental Contamination and Toxicology* 67(1), 60-67. https://doi.org/10.1007/s00244-014-0014-x.

Sapbamrer, R. and Thammachai, A. (2020). Factors affecting use of personal protective equipment and pesticide safety practices: A systematic review. *Environmental Research* 185, 109444. https://doi.org/10.1016/j.envres.2020.109444.

Sattler, C., Gianuca, A.T., Schweiger, O., Franzén, M. and Settele, J. (2020). Pesticides and land cover heterogeneity affect functional group and taxonomic diversity of arthropods in rice agroecosystems. *Agriculture, Ecosystems and Environment* 297, 106927. https://doi.org/10.1016/j.agee.2020.106927.

Schäfer, R.B. (2019). Responses of freshwater macroinvertebrates to pesticides: Insights from field studies. *Current Opinion in Environmental Science and Health* 11, 1-7. https://doi.org/10.1016/j.coesh.2019.06.001.

Schäfer, R.B., Liess, M., Altenburger, R., Filser, J., Hollert, H., Roß-Nickoll, M., Schäffer, A. and Scheringer, M. (2019). Future pesticide risk assessment: Narrowing the gap between intention and reality. *Environmental Sciences Europe* 31(21), 1-5. https://doi.org/10.1186/s12302-019-0203-3.

Schäfer, R.B., van den Brink, P.J. and Liess, M. (2011). Impacts of pesticides on freshwater ecosystems. In *Ecological Impacts of Toxic Chemicals*. Sánchez Bayo, F., van den Brink, P.J. and Mann, R.M. (eds.). Bentham Science. Chapter 6, 111-137. https://www.eurekaselect.com/51450/chapter/impacts-of-pesticides-on-freshwater-ecosystems.

Schäfer, R.B., von der Ohe, P.C., Rasmussen, J., Kefford, B.J., Beketov, M.A., Schulz, R. and Liess, M. (2012). Thresholds for the effects of pesticides on invertebrate communities and leaf breakdown in stream ecosystems. *Environmental Science and Technology* 46(9), 5134-5142. https://doi.org/10.1021/es2039882.

Schepker, T.J., Webb, E.B., Tillitt, D. and LaGrange, T. (2020). Neonicotinoid insecticide concentrations in agricultural wetlands and associations with aquatic invertebrate communities. *Agriculture, Ecosystems and Environment* 287, 106678. https://doi.org/10.1016/j.agee.2019.106678.

Schiesari, L. and Corrêa, D.T. (2016). Consequences of agroindustrial sugarcane production to freshwater biodiversity. *GCB-Bioenergy* 8(3), 644-657. https://doi.org/10.1111/gcbb.12279.

Schreiner, V.C., Link, M., Kunz, S., Szöcs, E., Scharmüller, A., Vogler, B. *et al.* (2021). Paradise lost? Pesticide pollution in a European region with considerable amount of traditional agriculture. *Water Research* 188(116528), 1-10. https://doi.org/10.1016/j.watres.2020.116528.

Science Advice for Policy by European Academies (SAPEA) (2018). *Improving Authorisation Processes for Plant Protection Products in Europe: A Scientific Perspective on the Potential Risks to Human Health*. https://allea.org/portfolio-item/improving-authorisation-processes-for-plant-protection-products-in-europe-a-scientific-perspective-on-the-assessment-of-potential-risk-to-human-health/?cn-reloaded=1.

Secretariat of the Convention on Biological Diversity (2020). *Global Biodiversity Outlook 5*. https://www.cbd.int/gbo5.

Settimi, L., Orford, R., Davanzo, F., Hague, C., Desel, H., Pelclova, D. *et al*. (2016). Development of a new categorization system for pesticides exposure to support harmonized reporting between EU Member States. *Environment International* 91, 332-340. https://doi.org/10.1016/j.envint.2016.02.034.

Settle, W.H., Ariawan, H., Astuti, E.T., Cahyana, W., Hakim, A.L., Hindayana, D. *et al*. (1996). Managing tropical rice pests through conservation of generalist natural enemies and alternative prey. *Ecology* 77(7), 1975-1988. https://doi.org/10.2307/2265694.

Shore, R.F. and Coeurdassier, M. (2018). Primary exposure and effects in non-target animals. In *Anticoagulant Rodenticides and Wildlife. Emerging Topics in Ecotoxicology (Principles, Approaches and Perspectives) vol 5*. van den Brink, N.W., Elliott, J.E., Shore, R.F. and Rattner, B.A. (eds.). Cham: Springer. Chapter 6, 135-157. https://link.springer.com/chapter/10.1007/978-3-319-64377-9_7.

Shunthirasingham, V., Oyiliagu, C.E., Cao, Z., Gouin, T., Wania, F., Lee, S.-C. *et al*. (2010). Spatial and temporal pattern of pesticides in the global atmosphere. *Journal of Environmental Monitoring* 12, 1650-1657. https://doi.org/10.1039/C0EM00134A.

Siddique, A., Liess, M., Shahid, N. and Becker, J.M. (2020). Insecticides in agricultural streams exert pressure for adaptation but impair performance in *Gammarus pulex* at regulatory acceptable concentrations. *Science of The Total Environment* 722, 137750. https://doi.org/10.1016/j.scitotenv.2020.137750.

Sieke, C., Michalski, B. and Kuhl, T. (2018). Probabilistic dietary risk assessment of pesticide residues in foods for the German population based on food monitoring data from 2009 to 2014. *Journal of Exposure Science and Environmental Epidemiology* 28(1), 46-54. https://doi.org/10.1038/jes.2017.7.

Silva, V., Mol, H.G.J., Zomer, P., Tienstra, M., Ritsema, C.J. and Geissen, V. (2019). Pesticide residues in European agricultural soils – A hidden reality unfolded. *Science of The Total Environment* 653, 1532-1545. https://doi.org/10.1016/j.scitotenv.2018.10.441.

Sjerps, R.M.A., Kooij, P.J.F., van Loon, A. and Van Wezel, A.P. (2019). Occurrence of pesticides in Dutch drinking water sources. *Chemosphere* 235, 510-518. https://doi.org/10.1016/j.chemosphere.2019.06.207.

Skovgaard, M., Renjel Encinas, S., Jensen, O.C., Andersen, J.H., Condarco, G. and Jørs, E. (2017). Pesticide residues in commercial lettuce, onion, and potato samples from Bolivia – A threat to public health? *Environmental Health Insights* 11, 1-8. https://doi.org/10.1177/1178630217704194.

Slama, R., Bourguignon, J.-P., Demeneix, B., Ivell, R., Panzica, G., Kortenkamp, A. and Zoeller, R.T. (2016). Scientific issues relevant to setting regulatory criteria to identify endocrine-disrupting substances in the European Union. *Environmental Health Perspectives* 124(10), 1497-1503. http://dx.doi.org/10.1289/EHP217.

Sonne, C., Siebert, U., Gonnsen, K., Desforges, J.-P., Eulaers, I., Persson, S. *et al.* (2020). Health effects from contaminant exposure in Baltic Sea birds and marine mammals: A review. *Environment International* 139, 105725. https://doi.org/10.1016/j.envint.2020.105725.

Spiller, K.J. and Dettmers, R. (2019). Evidence for multiple drivers of aerial insectivore declines in North America. *The Condor: Ornithological Applications* 121(2), 1-13. https://doi.org/10.1093/condor/duz010.

Spilsbury, F.D., Warne, M.S.J. and Backhaus, T. (2020). Risk assessment of pesticide mixtures in Australian rivers discharging to the Great Barrier Reef. *Environmental Science and Technology* 54 (22), 14361-14371. https://doi.org/10.1021/acs.est.0c04066.

Staley, Z.R., Harwood, V.J. and Rohr, J.R. (2015). A synthesis of the effects of pesticides on microbial persistence in aquatic ecosystems. *Critical Reviews in Toxicology* 45(10), 813-836. https://doi.org/10.3109/10408444.2015.1065471.

Stanton, R.L., Morrissey, C.A. and Clark, R.G. (2018). Analysis of trends and agricultural drivers of farmland bird declines in North America: A review. *Agriculture, Ecosystems and Environment* 254, 244-254. https://doi.org/10.1016/j.agee.2017.11.028.

Stehle, S. and Schulz, R. (2015). Agricultural insecticides threaten surface waters at the global scale. *Proceedings of the National Academy of Sciences of the United States of America* 112(18), 5750-5755. https://doi.org/10.1073/pnas.1500232112.

Stehle, S., Bub, S. and Schulz, R. (2018). Compilation and analysis of global surface water concentrations for individual insecticide compounds. *Science of The Total Environment* 639, 516-525. https://doi.org/10.1016/j.scitotenv.2018.05.158.

Stehle, S., Knäbel, A. and Schulz, R. (2013). Probabilistic risk assessment of insecticide concentrations in agricultural surface waters: A critical appraisal. *Environmental Monitoring and Assessment* 185(8), 6295-6310. https://doi.org/10.1007/s10661-012-3026-x.

Stockwell, V.O. and Duffy, B. (2012). Use of antibiotics in plant agriculture. *Revue Scientifique et Technique,* World Organization for Animal Health (OIE). 31(1), 199-210. https://doc.oie.int/seam/resource/directMedia/7KydABGWigpmB6PIAoDoAhshrFrZIn-u;jsessionid=1d60b632a935cedf58f7088cbd43?binaryFileId=10947&cid=1990.

Stokstad, R. (2013). The war against weeds down under. *Science* 341(6147), 734-736 https://doi.org/10.1126/science.341.6147.734. In Special Issue: Smarter Pest Control. https://www.sciencemag.org/site/special/pesticides/index.xhtml.

Stone, W.W., Gilliom, R.J. and Ryberg, K.R. (2014). Pesticides in U.S. streams and rivers: Occurrence and trends during 1992-2011. *Environmental Science and Technology* 48 (19), 11025-11030. https://doi.org/10.1021/es5025367.

Storck, V., Karpouzas, D.G. and Martin-Laurent, F. (2017). Towards a better pesticide policy for the European Union. *Science of the Total Environment* 575, 1027–1033. http://dx.doi.org/10.1016/j.scitotenv.2016.09.167.

Sud, M. (2020). *Managing the Biodiversity Impacts of Fertiliser and Pesticide use – Overview and Insights from Trends and Policies across Selected OECD Countries*. OECD Environment Working Paper No. 155. Paris: Organisation of Economic Co-operation and Development. https://dx.doi.org/10.1787/63942249-en.

Sun, J., Pan, L., Tsang, D.C.W., Zhan, Y., Zhu, L. and Li, X. (2018). Organic contamination and remediation in the agricultural soils of China: A critical review. *Science of The Total Environment* 615, 724-740. https://doi.org/10.1016/j.scitotenv.2017.09.271.

Sun, Y., Cao, M., Wan, Y., Wang, H., Liu, J., Pan, F. *et al.* (2020). Spatial variation of 2,4-D and MCPA in tap water and groundwater from China and their fate in source, treated, and tap water from Wuhan, Central China. *Science of The Total Environment* 727, 138691. https://doi.org/10.1016/j.scitotenv.2020.138691.

Székács, A., Mörtl, M. and Darvas, B. (2015). Monitoring pesticide residues in surface and ground water in Hungary: Surveys in 1990-2015. *Journal of Chemistry* 2015, 717948. https://doi.org/10.1155/2015/717948.

Tabashnik, B.E. and Carrière, Y. (2017). Surge in insect resistance to transgenic crops and prospects for sustainability. *Nature Biotechnology* 35(10), 926-935. https://doi.org/10.1038/nbt.3974.

Tabashnik, B.E. and Carrière, Y. (2019). Global patterns of resistance to Bt crops highlighting pink bollworm in the United States, China, and India. *Journal of Economic Entomology* 112(6), 2513-2523. https://doi.org/10.1093/jee/toz173.

Taiwo, A.M. (2019). A review of environmental and health effects of organochlorine pesticide residues in Africa. *Chemosphere* 220, 1126-1140. https://doi.org/10.1016/j.chemosphere.2019.01.001.

Tang, W., Wang, D., Wang, J., Wu, Z., Li, L., Huang, M. *et al.* (2018). Pyrethroid pesticide residues in the global environment: An overview. *Chemosphere* 191 (308), 990-1007. https://doi.org/10.1016/j.chemosphere.2017.10.115.

Tassin de Montaigu, C. and Goulson, D. (2020). Identifying agricultural pesticides that may pose a risk for birds. *PeerJ* 8, e9526. https://doi.org/10.7717/peerj.9526.

Taylor, P. and Reeder, R. (2020). Antibiotic use on crops in low and middleincome countries based on recommendations made by agricultural advisors. *CABI Agriculture and Bioscience* 1, 1. https://doi.org/10.1186/s43170-020-00001-y.

Teysseire, R., Manangama, G., Baldi, I., Carles, C., Brochard, P., Bedos, C. and Delva, F. (2020). Assessment of residential exposures to agricultural pesticides: A scoping review. *PLoS ONE* 15(4), e0232258. https://doi.org/10.1371/journal.pone.0232258.

Thundiyil, J.G., Stober, J., Besbelli, N. and Pronczuk, J. (2008). Acute pesticide poisoning: A proposed classification tool. *Bulletin of the World Health Organization* 86, 205-209. https://apps.who.int/iris/handle/10665/270154.

Toccalino, P.L., Gilliom, R.J., Lindsey, B.D. and Rupert, M.G. (2014). Pesticides in groundwater of the United States: Decadal-scale changes, 1993- 2011. *Ground Water* 52, 112-125. https://doi.org/10.1111/gwat.12176.

Todd, B.D., Willson, J.D. and Gibbons, J.W. (2010). The global status of reptiles and causes of their decline. In *Ecotoxicology of Amphibians and Reptiles*. Third edition. Sparling, D.W., Bishop, C.A. and Krest, S. (eds.). Pensacola, FL: CRC Press. 47-67. https://www.routledge.com/Ecotoxicology-of-Amphibians-and-Reptiles/Sparling-Linder-Bishop-Krest/p/book/9781420064162.

Tomenson, J.A. and Matthews, G.A. (2009). Causes and types of health effects during the use of crop protection chemicals: Data from a survey of over 6,300 smallholder applicators in 24 different countries. *International Archives of Occupational and Environmental Health* 89, 935-949. https://doi.org/10.1007/s00420-009-0399-4.

Topping, C.J., Aldrich, A. and Berny, P. (2020). Overhaul environmental risk assessment for pesticides. *Science* 367(6476), 360-363. https://doi.org/10.1126/science.aay1144.

Trudeau, V.L., Thomson, P., Zhang, W.S., Reynaud, S., Navarro-Martin, L. and Langlois, V.S. (2020). Agrochemicals disrupt multiple endocrine axes in amphibians. *Molecular and Cellular Endocrinology* 513, 110861. https://doi.org/10.1016/j.mce.2020.110861.

Tuck, S.L., Winqvist, C., Mota, F., Ahnström. J., Turnbull, L.A. and Bengtsson, J. (2013). Land-use intensity and the effects of organic farming on biodiversity: A hierarchical meta-analysis. *Journal of Applied Ecology* 51, 746-755. https://doi.org/10.1111/1365-2664.12219.

Tyrell, K., Willis, S., Williamson, S., Vodouhe, D.S. and Youdeowei, A. (2019). Monitoring and minimizing health risks related to pesticides. In *Integrated Management of Insect Pests: Current and Future Developments*. Kogan, M. and Heinrichs, E.A. (eds.). Cambridge, UK: Burleigh Dodds Science Publishing. Chapter 21 https://shop.bdspublishing.com/store/bds/detail/product/3-190-9781838799564.

Uman, L.S. (2011). Systematic Reviews and Meta-Analyses. *Journal of the American Academy of Child and Adolescent Psychiatry* 20(1), 57-59. https://www.ncbi.nlm.nih.gov/pmc/articles/PMC3024725/.

United Nations Environment Programme (2017a). *Second Global Monitoring Report: Global Monitoring Plan for Persistent Organic Pollutants.* UNEP/POPS/COP.8/INF/38. http://chm.pops.int/Implementation/GlobalMonitoringPlan/MonitoringReports/tabid/525/Default.aspx.

United Nations Environment Programme (2017b). *Overview Report I: Worldwide initiatives to identify endocrine disrupting chemicals (EDCs) and potential EDCs.* Prepared by: The International Panel on Chemical Pollution (IPCP). https://www.unenvironment.org/explore-topics/chemicals-waste/what-we-do/emerging-issues/scientific-knowledge-endocrine-disrupting.

United Nations Environment Programme (2018). *Resolution 3/4 on Environment and Health*. UNEP/EA.3/Res.4. https://wedocs.unep.org/bitstream/handle/20.500.11822/30795/UNEA3_4EN.pdf?sequence=1&isAllowed=y.

United Nations Environment Programme (2020). Pesticide poisoning – Data from poison centres and national poisoning statistics. Working document for the UNEP report, *Environmental and Health Effects of Pesticides and Fertilizers and Ways to Minimize Them*; version: 25 January 2021. Supporting document to *Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them.* https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

United Nations Environment Programme (2022). *Synthesis Report on Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them*. https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

United Nations Environment Programme and World Health Organization (2012). *State of the Science of Endocrine Disrupting Chemicals – An Assessment of the State of the Science of Endocrine Disruptors Prepared by a Group of experts for the United Nations Environment Programme and World Health Organization*. Bergman, Å, Heindel, J.J., Jobling, S., Kidd, K.A. and Zoeller, R.T. (eds.) Nairobi and Geneva. https://www.who.int/ceh/publications/endocrine/en/.

United States Department of Agriculture (2019). *Pesticide Data Program Annual Summary, Calendar Year 2018*. https://www.ams.usda.gov/reports/pdp-annual-summary-reports.

United States Environmental Protection Agency (2016). *Generic Ecological Assessment Endpoints (GEAEs) for Ecological Risk Assessment: Second Edition with Generic Ecosystem Services Endpoints Added.* https://www.epa.gov/sites/production/files/2016-08/documents/geae_2nd_edition.pdf.

United States Environmental Protection Agency (2017). DEEM-FCID/Calendex Software Installer. https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/deem-fcidcalendex-software-installer. Accessed 26 December 2020.

United States Environmental Protection Agency (2020a). Pesticide chemical search: conventional, antimicrobial and biopesticide active ingredients. https://ofmpub.epa.gov/apex/pesticides/f?p=chemicalsearch:1. Accessed 1 December 2020.

United States Environmental Protection Agency (2020b). Ecological risk assessment for pesticides: Technical overview. https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/ecological-risk-assessment-pesticides-technical. Accessed 29 December 2020.

United States Environmental Protection Agency (2020c). New approach methods work plan: reducing use of animals in chemical testing. https://www.epa.gov/chemical-research/epa-new-approach-methods-work-plan-reducing-use-animals-chemical-testing. Accessed 26 December 2020.

United States Environmental Protection Agency (2020d). *Chemical Safety for Sustainability. National Research Program Strategic Research Action Plan 2019-2022.* https://www.epa.gov/sites/production/files/2020-10/documents/css_fy19-22_strap-final_2020.pdf.

United States Environmental Protection Agency, Heath Canada Pest Management Agency, and California Department of Pesticide Regulation (2014). *Guidance for Assessing Pesticide Risks to Bees.* https://www.epa.gov/sites/production/files/2014-06/documents/pollinator_risk_assessment_guidance_06_19_14.pdf.

United States Food and Drug Administration (2019). *Pesticide Residue Monitoring Program – Fiscal Year 2017 Pesticide Report.* https://www.fda.gov/food/pesticides/pesticide-residue-monitoring-2017-report-and-data.

United States Geological Survey (2020). National Water-Quality Assessment (NAWQA). https://www.usgs.gov/mission-areas/water-resources/science/national-water-quality-assessment-nawqa?qt-science_center_objects=0#qt-science_center_objects. Accessed 29 December 2020.

United States National Institute for Safety and Health (2015). Hierarchy of controls. *Centers for Disease Control and Prevention*. https://www.cdc.gov/niosh/topics/hierarchy/. Accessed 28 December 2020.

van den Brink, N.W., Elliott, J.E., Shore, R.F. and Rattner, B.A. (2018a). Anticoagulant rodenticides and wildlife: Introduction. In *Anticoagulant Rodenticides and Wildlife. Emerging Topics in Ecotoxicology (Principles, Approaches and Perspectives) vol 5*. van den Brink, N.W., Elliott, J.E., Shore, R.F. and Rattner, B.A. (eds.). Cham: Springer. 1-9. https://doi.org/10.1007/978-3-319-64377-9_1.

van den Brink, N.W., Elliott, J.E., Shore, R.F. and Rattner, B.A. (eds.) (2018b). *Anticoagulant Rodenticides and Wildlife. Emerging Topics in Ecotoxicology (Principles, Approaches and Perspectives) vol 5*. Cham: Springer. https://www.springer.com/gp/book/9783319643755.

van den Brink, P.J. and Daam, M.A. (2014). Multispecies environmental testing designs, In *Encyclopedia of Toxicology.* Third edition. Elsevier. 399-402. https://doi.org/10.1016/B978-0-12-386454-3.00569-8.

van der Sluijs, J.P., Amaral-Rogers, V., Belzunces, L.P., Bijleveld van Lexmond, M.F.I.J., Bonmatin, J.-M., Chagnon, M. *et al*. (2015). Conclusions of the Worldwide Integrated Assessment on the risks of neonicotinoids and fipronil to biodiversity and ecosystem functioning. *Environmental Science and Pollution Research* 22, 148-154. https://doi.org/10.1007/s11356-014-3229-5.

Van der Valk, H.C.H.G., Niassy, A. and Bèye, A.B. (1999). Does grasshopper control create grasshopper problems? – Monitoring side-effects of fenitrothion applications in the western Sahel. *Crop Protection* 18(2), 139–149. https://doi.org/10.1016/S0261-2194(99)00012-5.

van Klink, R., Bowler, D.E., Gongalsky, K.B., Swengel, A.B., Gentile, A. and Chase, J.M. (2020). Meta-analysis reveals declines in terrestrial but increases in freshwater insect abundances. *Science* 368(6489), 417-420. https://doi.org/10.1126/science.aax9931 and Erratum: https://doi.org/10.1126/science.abf1915.

van Wijngaarden, R.P., Cuppen, J.G., Arts, G.H., Crum, S.J., Van Den Hoorn, M.W., Van Den Brink, P.J. and Brock T.C. (2004). Aquatic risk assessment of a realistic exposure to pesticides used in bulb crops: A microcosm study. *Environmental Toxicology and Chemistry* 23, 1479-1498. https://doi.org/10.1897/03-80.

Varea, R., Piovano, S. and Ferreira, M. (2020). Knowledge gaps in ecotoxicology studies of marine environments in Pacific Island Countries and Territories – A systematic review. *Marine Pollution Bulletin* 156, 111264. https://doi.org/10.1016/j.marpolbul.2020.111264.

Vašíčková, J., Hvězdová, M., Kosubová, P. and Hofman, J. (2019). Ecological risk assessment of pesticide residues in arable soils of the Czech Republic. *Chemosphere* 216, 479-487. https://doi.org/10.1016/j.chemosphere.2018.10.158.

Vijver, M.G., Hunting, E.R., Nederstigt, T.A.P., Tamis, W.L.M., van den Brink, P.J. and van Bodegom, P.M. (2017). Postregistration monitoring of pesticides is urgently required to protect ecosystems. *Environmental Toxicology and Chemistry* 36(4), 860-865. https://doi.org/10.1002/etc.3721.

Vorkamp, K. and Rigét, F.F. (2014). A review of new and current-use contaminants in the Arctic environment: Evidence of long-range transport and indications of bioaccumulation. *Chemosphere* 111, 379-395. https://doi.org/10.1016/j.chemosphere.2014.04.019.

Vulliet, E., Tournier, M., Vauchez, A., Wiest, L., Baudot, R., Lafay, F. *et al.* (2014). Survey regarding the occurrence of selected organic micropollutants in the groundwaters of overseas departments. *Environmental Science and Pollution Research* 21(12), 7512-7521. https://doi.org/10.1007/s11356-014-2619-z.

Wagner, D.L. (2020). Insect declines in the anthropocene. *Annual Review of Entomology* 65, 457-480. https://doi.org/10.1146/annurev-ento-011019-025151.

Wagner, N., Mingo, V., Schulte, U. and Lötters, S. (2015). Risk evaluation of pesticide use to protected European reptile species. *Biological Conservation* 191, 667-673. http://dx.doi.org/10.1016/j.biocon.2015.08.002.

Wang, A., Hu, X., Wan, Y., Mahai, G., Jiang, Y., Huo, W. *et al*. (2020). A nationwide study of the occurrence and distribution of atrazine and its degradates in tap water and groundwater in China: Assessment of human exposure potential. *Chemosphere* 252. https://doi.org/10.1016/j.chemosphere.2020.126533.

Wang, X., Wang, C., Zhu, T., Gong. P., Fu, J. and Cong, Z. (2019). Persistent organic pollutants in the polar regions and the Tibetan Plateau: A review of current knowledge and future prospects. *Environmental Pollution* 248, 191-208. https://doi.org/10.1016/j.envpol.2019.01.093.

Warner, G.R., Mourikes, V.E., Neff, A.M., Brehm, E. and Flaws, J.A. (2020). Mechanisms of action of agrochemicals acting as endocrine disrupting chemicals. *Molecular and Cellular Endocrinology* 502, 110680. https://doi.org/10.1016/j.mce.2019.110680.

Weir, S.M., Talent, L.G., Anderson, T.A. and Salice, C.J. (2014). Unraveling the relative importance of oral and dermal contaminant exposure in reptiles: Insights from studies using the western fence lizard (*Sceloporus occidentalis*). *PLoS ONE* 9(6), e99666. https://doi.org/10.1371/journal.pone.0099666.

Wellcome (2018). *Initiatives for Addressing Antimicrobial Resistance in the Environment: Current Situation and Challenges: Executive Summary.* Wellcome, United States Centers for Disease Control and Prevention, and UK Research and Innovation Network. London: Wellcome Trust. https://wellcome.org/sites/default/files/antimicrobial-resistance-environment-summary.pdf.

Wellington, E.M.H., Boxall, A.B.A., Cross, P., Feil, E.J., Gaze, W.H., Hawkey, P.M. *et al.* (2013). The role of the natural environment in the emergence of antibiotic resistance in Gram-negative bacteria. *The Lancet Infectious Diseases* 13, 155-65. https://doi.org/10.1016/S1473-3099(12)70317-1.

Wesseling, C., Corriols, M. and Bravo, V. (2005). Acute pesticide poisoning and pesticide registration in Central America. *Toxicology and Applied Pharmacology* 207(2), 697-705. https://doi.org/10.1016/j.taap.2005.03.033.

Whalon, M.E. (2008). Introduction. In *Global Pesticide Resistance in Arthropods*. Whalon, M.E., Mota-Sanchez, D. and Hollingworth, R.M. (eds.). Wallingford, UK: CAB International. 1-4. https://www.cabi.org/bookshop/book/9781845933531/.

Whitfield, S.M., Lips, K.R. and Donnelly, M.A. (2016). Amphibian decline and conservation in Central America. *Copeia* 104, 351-379. https://doi.org/10.1643/CH-15-300.

Wolfram, J., Stehle, S., Bub, S., Petschick, L.L. and Schulz, R. (2018). Meta-analysis of insecticides in United States surface waters: Status and future implications. *Environmental Science and Technology* 52(24), 14452-14460. https://doi.org/10.1021/acs.est.8b04651.

Wood, T.J. and Goulson, D. (2017). The environmental risks of neonicotinoid pesticides: A review of the evidence post 2013. *Environmental Science and Pollution Research* 24(21), 17285-17325. https://doi.org/10.1007/s11356-017-9240-x.

Woodcock, B.A., Bullock, J.M., Shore, R.F., Heard, M.S., Pereira, M.G., Redhead, J. *et al.* (2017). Country-specific effects of neonicotinoid pesticides on honey bees and wild bees. *Science* 356(6345), 1393-1395. https://doi.org/10.1126/science.aaa1190.

Woodcock, B.A., Isaac, N.J.B., Bullock, J.M., Roy, D.B., Garthwaite, D.G., Crowe, A. and Pywell, R.F. (2016). Impacts of neonicotinoid use on long-term population changes in wild bees in England. *Nature Communications* 7, 12459. https://doi.org/10.1038/ncomms12459.

World Health Organization (1990). *Public Health Impact of Pesticides Used in Agriculture*. https://apps.who.int/iris/handle/10665/39772.

World Health Organization (2004). *IPCS Risk Assessment Terminology*. https://apps.who.int/iris/handle/10665/42908.

World Health Organization (2014). *Preventing Suicide: A Global Imperative*. https://www.who.int/publications/i/item/9789241564779.

World Health Organization (2015). *Global Action Plan on Antimicrobial Resistance*. https://apps.who.int/iris/handle/10665/193736.

World Health Organization (2019). *Critically Important Antimicrobials for Human Medicine. 6th revision 2018. Ranking of Medically Important Antimicrobials for Risk Management of Antimicrobial Resistance due to Non-human use*. Geneva. https://apps.who.int/iris/handle/10665/312266.

World Health Organization (2020a). *World Malaria Report 2020: 20 Years of Global Progress and Challenges*. Geneva. https://www.who.int/publications/i/item/9789240015791.

World Health Organization (2020b). INTOX tools for harmonized data collection: Poisoning severity score. Appendix 1. In *Poisons Centre Training Manual*. https://apps.who.int/iris/bitstream/handle/10665/329502/WHO-FWC-PHE-EPE-13.01-eng.pdf?ua=1.

World Health Organization (2020c). World directory of poison centres. https://www.who.int/data/gho/data/themes/topics/indicator-groups/poison-control-and-unintentional-poisoning. Accessed 26 December 2020.

World Health Organization (2020d). Global Environment Monitoring System (GEMS) / Food Contamination Monitoring and Assessment Programme. https://www.who.int/teams/nutrition-and-food-safety/databases/global-environment-monitoring-system-food-contamination. Accessed 26 December 2020.

World Health Organization (2020e). Antimicrobial resistance. https://www.who.int/health-topics/antimicrobial-resistance. Accessed 26 December 2020.

World Health Organization and Food and Agriculture Organization of the United Nations (2019a). *Global Situation of Pesticide Management in Agriculture and Public Health: Report of a 2018 WHO-FAO Survey*. Geneva and Rome. https://apps.who.int/iris/bitstream/handle/10665/329971/9789241516884-eng.pdf?sequence=1&isAllowed=y.

World Health Organization and Food and Agriculture Organization of the United Nations (2019b). *Preventing Suicide: A Resource for Pesticide Registrars and Regulators*. Geneva and Rome. https://www.who.int/publications/i/item/9789241516389.

World Health Organization, Food and Agriculture Organization of the United Nations and World Organisation for Animal Health (2020). *International Instruments on the Use of Antimicrobials across the Human, Animal and Plant Sectors*. https://www.who.int/publications/i/item/9789240013964.

Wu, J., Ge, L., Liu, F., Song, Q. and Stanley, D. (2020). Pesticide-induced planthopper population resurgence in rice cropping systems. *Annual Review of Entomology* 65, 409-429. https://doi.org/10.1146/annurev-ento-011019-025215.

Xie, H., Wang, X., Chen, J., Li, X., Jia, G., Zou, Y. *et al.* (2019). Occurrence, distribution and ecological risks of antibiotics and pesticides in coastal waters around Liaodong Peninsula, China. *Science of The Total Environment* 656, 946-951. https://doi.org/10.1016/j.scitotenv.2018.11.449.

Yadav, I.C., Devi, N. L., Syed, J. H., Cheng, Z., Li, J., Zhang, G., and Jones, K.C. (2015). Current status of persistent organic pesticides residues in air, water, and soil, and their possible effect on neighboring countries: A comprehensive review of India. *Science of The Total Environment* 511, 123-137. https://doi.org/10.1016/j.scitotenv.2014.12.041.

Yeung, M., Kerr, W.A., Coomber, B., Lantz, M. and McConnell, A. (2018). Declining harmonization in maximum residue levels for pesticides. *British Food Journal* 120(4), 901-913. https://doi.org/10.1108/BFJ-05-2017-0291.

Zubrod, J.P., Bundschuh, M., Arts, G., Brühl, C.A., Imfeld, G., Knäbel, A. *et al.* (2019). Fungicides: An overlooked pesticide class? *Environmental Science and Technology* 53(7), 3347-3365. https://doi.org/10.1021/acs.est.8b0439.

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them: Envisioning a chemical-safe world

Annexes to Chapter 4.3 – Adverse environmental effects of pesticide use

Contents

Annex 4.3-1 Recent reviews of pesticide concentrations in surface water bodies and their risks to aquatic life 2

Annex 4.3-2 Recent reviews and studies of pesticide concentrations in ambient air. 8

Annex 4.3-3 Recent reviews of pesticide concentrations in soils and their risks to soil organisms 13

Annex 4.3-4 Recent reviews of pesticide concentrations in groundwater and drinking water. 18

Annex 4.3-5 Recent reviews and large scale studies on the effects of pesticide use on vertebrate wildlife 26

Annex 4.3-6 Recent large-scale studies and partial reviews of the effects of pesticide use on biodiversity 33

Annex 4.3-1 Recent reviews of pesticide concentrations in surface water bodies and their risks to aquatic life

Included are reviews published between 2010 and 2020

Region (coverage)	**Monitoring (or publication) period**	**Extent of review**	**Pesticides**	**Outcomes**	**Reference**
Global					
Global	2012 – 2019 (publication)	Not specified	No specified	– Most detected herbicides in surface water were atrazine and its metabolites, metolachlor – Most detected insecticide was chlorpyrifos – Mist detected fungicides were carbendazim, tebuconazole	De Souza *et al.* 2020
Global (73 countries)	1960 – 2011	838 studies with 11,300 measured concentrations	28 insecticides	– in 97% of the cases, no insecticides were measured in the samples – 52% of measured insecticide concentrations exceeded regulatory thresholds from the USA and/or the EU, and on 69% of the sampled sites – Median concentrations of neonicotinoids exceeded those of organochlorines and organophosphates by a factor of about 3, and of pyrethroids by a factor 10 – Most insecticide concentration monitoring data for surface waters were available from North America, Asia and Europe – No monitoring of surface water exposure by insecticides exist for about 90% of high-intensity agricultural areas of the world – Highest insecticide concentrations were detected in Africa, Asia and South America	Stehle & Schulz 2015 Stehle *et al.* 2018
Global (11 countries)	2010 – 2016 (publication)	31 studies	Neonicotinoids	– Geometric mean for average concentrations = 0.13 µg/L – Geometric mean for peak concentrations = 0.63 µg/L – 74% of measured average concentrations exceeded ecological thresholds – 81% of measured peak concentrations exceeded ecological thresholds – Average neonicotinoid residues in surface waters increased over past 15 years	Morrissey *et al.* (2015) Sánchez-Bayo *et al.* (2016)
Global (15 countries)	1986 – 2017 (publication)	176 papers	Pyrethroids	– Pyrethroid concentrations in surface water worldwide ranged from not detected to 13,000 µg/L – High concentrations of pyrethroids are found across all global regions; no geographical hot spots – No risk assessment conducted	Tang *et al.* 2018

Region (coverage)	**Monitoring (or publication) period**	**Extent of review**	**Pesticides**	**Outcomes**	**Reference**
Regional					
Europe (34 countries)	2007 – 2017	Monitoring data from the Waterbase – Water Quality Database	180 pesticides	– Assessed were the exceedance rates of European aquatic environmental quality standards – 5–15% of European surface water monitoring stations showed exceedances by herbicides; 3–8 % of stations by insecticides; exceedances of fungicides were less prevalent	Mohaupt *et al.* 2020
Africa	2000–2018 (publication)	Not reported	Organochlorines Current use pesticides	– Concentrations of pesticides reported in African surface waters range from 0.06 ng/L to 69 µg/L (insecticides), 0.2 ng/L to 14 µg/L (herbicides), and 0.1 ng/L to 9 µg/L (fungicides) – Highest concentrations reported for chlorpyrifos, chlorpyrifos-ethyl, malathion, diazinon, endosulfan, DDT and spiroxamine	K'oroje *et al.* 2020
Africa	2000 – 2018 (publication)	22 studies from 7 African countries	Organochlorines	– Average values in surface waters: Σ-DDT = 1271 ± 1721 µg/L; dieldrin = 793 ± 1760 µg/L; aldrin = 384 ± 1043 µg/L; Σ-HCH = 159 ± 368 µg/L; HBC = 77± 104 µg/L; methoxychlor = 49 ± 85 µg/L – OCP concentration were many times higher than levels in other parts of the world where OCPs were still used during the same period (e.g. Japan, China, Iran, India and Mexico)	Taiwo 2019
Caribbean and Pacific	1997 – 2001	33 samples from marine surface slicks	Organochlorine pesticides (OCPs)	– OCPs detected in 31 out of 33 samples – Concentrations observed were below commonly accepted toxic levels to animals and humans – Traces of OCPs were observed in areas far away from human habitation	Menzies *et al.* 2013
Europe (91 river basins across the continent)	2006 – 2010	4,001 monitoring sites	223 organic chemicals, including 98 pesticides	– 14% of the monitoring sites were likely to be acutely affected by organic chemicals and 42% likely to be chronically affected – Pesticides were responsible for 81%, 97% and 96% of the observed exceedances of acute risk thresholds for fish, invertebrates and algae respectively	Malaj *et al.* 2014
South Asia	1988 – 2011	13 studies from 3 countries	Organochlorine pesticides (OCPs)	– All studies reported 1 or more OCPs in surface water at concentrations above detection levels – Ranges of concentrations detected : Σ-DDT = non detectable (ND) – 400 µg/L; Σ-HCH = ND – 75 µg/L; endosulfan = ND – 66 µg/L	Ali *et al.* 2014

Region (coverage)	**Monitoring (or publication) period**	**Extent of review**	**Pesticides**	**Outcomes**	**Reference**
National (selected country-wide or review studies)					
Brazil	Up to December 2015 (publication)	17 studies	43 current use pesticides	– Pesticides were detected in 14% of the samples in concentrations above the LOQ of the analytical method used – From the 43 pesticides investigated, 34 were quantified at least once – Freshwater water quality criteria (WQC) for the protection aquatic life were compiled from the literature for 31 of the 34 pesticides – A potential risk to aquatic life was identified for 59% of the pesticides with the occurrence data in Brazil – This review only encompasses 43 compounds, which are 11% of the pesticides approved in Brazil for agricultural use	Albuquerque *et al.* 2016
China	1995 – 2013 (publication)	National review: 30 studies for surface water 51 studies for sediment	Mainly organochlorine pesticides (OCPs); some current use pesticides	– Risk assessments only conducted for Σ-DDT and lindane, based on Norwegian quality criteria for sediment and surface water – Quality criteria for Σ-DDT in surface water: 13% of all cases = good, 22% = bad, 4% = very bad – Quality criteria for Σ-DDT in sediment: 58% of all cases = good, 3% = bad, 3% = very bad – Quality criteria for lindane in surface water: 73% of all cases = good, 16% = bad, 5% = very bad – Quality criteria for lindane in sediment: 54% of all cases = good, 10% = bad, 10% = very bad	Grung *et al*. 2015
China	2013	26 surface water and sediment sampling locations in the Yellow River estuary	Organochlorine pesticides (OCPs)	– Σ-HCH was detected in all surface water and sediment samples; and Σ-DDT in all surface water and 96% of sediment samples – Mean concentration of Σ-HCH in surface water = 19 µg/L and in sediment = 20 µg/L; Σ-DDT in surface water = 10 µg/L and in sediment = 7 µg/L – No environmental risk assessment	Li *et al.* 2015
China	2016	96 samples from 16 rivers in eastern China	9 neonicotinoid insecticides	– Average concentrations of the 9 neonicotinoids combined were 343 ng/L during the dry season, and 174 ng/L during the wet season – 27% of river water samples exceeded the thresholds for acute ecological risks – 84% of river water samples exceeded the thresholds for chronic ecological risks – Contamination levels were higher than those found in the USA and Japan	Chen *et al.* 2019

Region (coverage)	**Monitoring (or publication) period**	**Extent of review**	**Pesticides**	**Outcomes**	**Reference**
Egypt	2013	60 samples from 20 sampling sites along the River Nile	Organochlorine pesticides (OCPs) & organophosphate (OP) pesticides	– Organochlorine pesticides detected were dieldrin, endrin, p,p'-DDE, p,p'- DDD and p,p'-DDT – organophosphorus pesticides detected were triazophos, ethoprophos, quinalphos, chlorpyrifos, fenitrothion, ethion, fenamiphos and pirimiphos-methyl – dieldrin and p,p'-DDT residues in some sampling sites exceeded drinking water guidelines of WHO	Dashan *et al.* 2016
Ethiopia	2015	46 sediment samples from 1 river basin	16 organochlorine pesticides (OCPs)	– All samples contained measured concentrations of at least 1 OCP – Total concentrations of OCPs, ranged from 6.63 to 206.13 ng/g dry weight (dw) – Heptachlor, heptachlor epoxide, p,p'-DDT and β-HCH) had highest concentrations – Adverse ecological effects were expected to occur mainly due to p,p'-DDT and γ-HCH	Dirbaba *et al*. 2018
Ghana	2001 – 2016 (publication)	8 studies	Organochlorine pesticides (OCPs)	– Range of OCP concentrations in surface water = from below limit of detection – 6350 ng/L – Concentrations in surface water of Σ-DDT and Σ-endosulfan showed significant increase from 2004 – 2012; concentrations of Σ-HCH did not – No aquatic risk assessment conducted	Bruce-Vanderpuije *et al.* 2019
Hungary	1990 – 2015	National monitoring programme	49 pesticides and degradation products	– Pesticide residues detected in 32 – 61% of samples – Main surface water contaminants were triazines (atrazine, propisochlor), chloroacetamides (acetochlor, metolachlor), and phenoxycarboxylic acids (2,4-D, MCPA) during the late 90s, followed by triazines (atrazine, prometryn, and diazinon) and chloroacetamides (acetochlor) after the turn of the millennium, while glyphosate and neonicotinoids were more frequently detected in latter years of the programme	Székács *et al.* 2015
India	1990 – 2014	14 studies	organochlorine pesticides (OCPs)	– Surface water bodies throughout India – Σ-DDT = not detected (ND) – 6121 ng/L; Σ-HCH = ND – 4911; endosulfan = ND – 130 ng/L	Yadav *et al.* 2015
Italy	1997 – 2013 (publication)	10 studies	137 pesticides	– 54.3% of reported maximum pesticides concentrations in surface water were above the environmental quality standards – Pesticides considered to represent the most severe threat for the Italian water resources were AMPA (glyphosate metabolite), terbuthylazine, diuron, 2,4-DB, 2,4-D, terbutryne, malathion, metolachlor, pendimethalin and linuron.	Meffe & de Bustamente 2014

Region (coverage)	**Monitoring (or publication) period**	**Extent of review**	**Pesticides**	**Outcomes**	**Reference**
Japan	1987 – 2016	34 samples from 11 rivers	9 organophosporous pesticides	– High risks for aquatic organisms = 56% of cases – Low risk for aquatic organisms = 18% of cases – Diazinon and fenitrothion pose highest risks to aquatic environments	Derbalah *et al.* 2019
The Netherlands	2013 – 2018	96 sampling locations	Plant protection products	– Exceedance of national surface water quality criteria (chronic exposure) declined between 2013 to 2015 with 15% for chronic exposure and with 30% for acute exposure – Number of sampling locations with exceedances by at least one pesticide did not significantly decline	PBL 2019
Romania	2016	19 streams	244 pesticides and metabolites	– Up to 50 pesticides were detected simultaneously, resulting in sum concentrations between 0.02 and 37 µg/L – Streams in low intensity agriculture, despite a minor reported use of agrochemicals, exhibited similar levels of pesticide pollution as in regions of high intensity agriculture – The sum concentration as well as the toxicities were in a similar range as in high intensity agricultural streams of Western Europe	Schreiner *et al.* 2021
South Africa	1977 – 2011	41 studies	OCPs and current use pesticides	– Even more recent studies focus primarily on OCPs – Widespread occurrences of OCPs with some banned organochlorine residues still present in the aquatic environment – Atrazine frequently detected in surface waters	Ansara-Ross *et al.* 2012
Sudan	2004 – 2005	27 samples from 3 locations	22 pesticides	– Organochlorines detected in 70% of samples, pyrethroids in in 15% of samples, herbicides in 4% of samples, organophosphates not detected – Highest pesticide concentrations ranges for: Σ-heptachlor = non-detected (ND) – 1123 ng/L; Σ-DDT = ND – 427 ng/L; cypermethrin = ND – 630 ng/L; fenvalerate = ND – 600 ng/L; pendimethalin = ND – 1060 ng/L – Pesticide concentration appear to be lower and less frequently detected than in earlier studies in the Nile	Nesser *et al.* 2016
Switzerland	2005 – 2012	345,000 samples	60 pesticides	– 27% of evaluated pesticides showed exceedance of national regulatory acceptable concentrations (RACs) at 1 or more sampling sites – 95% of measured concentrations (including non-detections) of all pesticides were below their respective RACs – Exceedance of RACs occurred in small to medium surface waters, surrounded by vineyards – Mainly herbicides and fungicides exceeded RACs	Knauer 2016

Region (coverage)	**Monitoring (or publication) period**	**Extent of review**	**Pesticides**	**Outcomes**	**Reference**
Tanzania	2003 – 2014 (publication)	5 studies	Mainly organochlorine pesticides (OCPs)	– Up to 7 OCPs measured in surface waters above detection levels in individual studies – No risk assessment	Elibariki *et al*. 2017
USA	1962 – 2017	259 studies with 5817 measured concentration covering 644 sampling sites	32 insecticides and 6 degradation products	– 49% of measured pesticide concentrations exceeded US regulatory threshold levels – 70% of sampling sites experienced at least one exceedance of a US regulatory threshold level – Exceedance rate of pyrethroids = 81%; organophosphates & carbamates = 42%, organochlorines = 33%; neonicotinoids = 22%; phenylpyrazoles = 25% – Overall exceedance rate 1960–2001 = 49%; 2002–2015 = 49%	Wolfram *et al.* 2018
USA	1992 – 2011	USGS – National Water Quality Assessment database, covering 182 sites (1992–2001) and 96 sites (2002–2011)	47 pesticides (1992–2001) 123 pesticides (USA2002–2011)	– Exceedance rates of US chronic aquatic-life benchmark levels (2002–2011) – Agricultural streams/rivers: 69% – Urban streams/rivers: 90% – Mixed streams/rivers: 46% – Exceedance rates did not significantly change from 1992–2001 to 2002–2011 for agricultural and mixed streams; they increased considerably for urban streams – Upward and downward trends in pesticide concentrations to large extent reflect changes in their use patterns	Stone *et al.* 2014
USA	2012 – 2014	38 streams in 24 US states; 149 samples	6 neonicotinoids insecticides	– At least 1 neonicotinoid detected in 53% of the sampled sites – 2 or more neonicotinoids in 26% of samples – 8% of samples with detectable concentrations were >100 ng/L – No risk assessment conducted	Hladik & Kolpin 2016

Annex 4.3-2 Recent reviews and studies of pesticide concentrations in ambient air.

Included are reviews published between 2010 and 2020

Region (coverage)	**Monitoring (or publication) period**	**Extent of review/study**	**Pesticides**	**Outcomes**	**Reference**
Global					
Global	2000 – 2015	Baseline data and time trends for 5 regions of the world.	12 organochlorine pesticides and their metabolites	– Long time series monitoring data of legacy POPs in air are available from Asia, Central and Eastern Europe and Western Europe and Others Group, while information on changes in concentrations over time is very limited in Africa, the Pacific and in Latin American and Caribbean Group – Information on changes over time in concentrations of the newly listed POPs is still limited – Overall, concentrations of POPs measured in air have largely decreased. The steep decrease in air concentrations of legacy POPs such as organochlorine pesticide) seems to have followed their early regulation in the 1980s – By 2000, the majority of primary sources had been controlled and the relatively low levels that are currently measured do not show significant changes and are driven by secondary sources and, for some compounds, emissions from stockpiles/waste management practices	UNEP 2017
Global (9 countries)	2011 – 2014	34 studies covering 475 cases (=pesticide-study combinations)	150 pesticide active ingredients and metabolites	– Most studies conducted on herbicides and insecticides, in Canada, USA, France, Spain and China; no studies conducted in low or lower-middle income countries – Organochlorine pesticides and metabolites represented 27% of cases – The more frequently detected CUPs worldwide are the fungicides chlorothalonil and folpet, and the insecticides chlorpyrifos, dimethoate, malathion and phosmet, with average concentrations ranging from 0.01 to 11.6 ng/m^3 – Inhalation exposure of CUPs in the general population does not represent a significant concern, except in some cases for chlorpyrifos. The combined risk resulting from exposure to OP, pyrethroid and carbamate pesticides was also considered to be acceptable	Coscollà & Yusà 2016

Region (coverage)	**Monitoring (or publication) period**	**Extent of review/study**	**Pesticides**	**Outcomes**	**Reference**
Global	1988 – 2008	16 studies on air concentrations in developing and developed countries	DDTs	– Most data from the 1990s – DDTs in ambient air is compiled from Congo, Malawi and Zimbabwe in Africa; southern Mexico, the Brazilian mountains and Costa Rica in Latin America; and India, Vietnam, Thailand and the Solomon Islands in S-SE Asia. – Concentrations of DDTs a factor 10-100 higher in tropical (generally developing) than in temperate (generally industrialised) countries – Most of the developing countries described in this review are located in the tropical regions where climatic conditions favour dissipation of DDT and other chlorinated pesticides – Continuing use of DDT in indoor residual spraying programs in some parts of the region and emissions from old pesticide storage areas may be attributed to the higher DDTs levels in the developing regions	Mochungong & Zhu 2015
Global	2005 - 2008	34 – 46 sampling sites on all continents	OCPs: HCHs, HCB, chlordanes, heptachlor, aldrin, dieldrin, endrin, DDTs, and metabolites CUPs: endosulfans, trifluralin, chlorothalonil, dacthal, pendimethalin	– Organochlorine pesticides such as a- and g-hexachlorocyclohexane, endosulfans, DDT and its metabolites, and chlordane-related compounds tend to be more prevalent in developing countries, especially in Asia – Current use pesticides such as trifluralin and chlorothalonil showed higher levels in Europe and North America – Levels of hexachlorobenzene, hexachlorocyclohexanes and chlordanes decline in most world regions; levels of organochlorine pesticides in India, however, remained exceptionally high – The CUPs, including endosulfans, did not display clear trends over the four years of sampling, except that the levels of α-endosulfan and chlorothalonil were decreasing in the European atmosphere	Shunthirasingham *et al.* 2010
Regional					
East Asia to the High Arctic Ocean	2010	17 air samples on a latitudinal transect from the East China Sea to the high Arctic (33.23–84.5° N).	6 current-use pesticides (CUPs): trifluralin, endosulfan, chlorothalonil, chlorpyrifos, dacthal (DCPA) and dicofol.	– In all oceanic air samples, the six CUPs were detected, showing highest level (>100 pg/m3) in the Sea of Japan – Gaseous CUPs decreased from East Asia (between 36.6 and 45.1° N) toward Bering and Chukchi Seas	Zhong *et al.* 2012

Region (coverage)	**Monitoring (or publication) period**	**Extent of review/study**	**Pesticides**	**Outcomes**	**Reference**
Arctic, Antarctica, Tibetan Plateau	1990 – 2014 (review)	66 reviews for the Arctic, 2 reviews for the Antarctic, 1 review Tibetan Plateau	Organochlorine pesticides (OCPs)	– Concentrations of chemicals in the Antarctic are generally lower than those of the Arctic; similar concentration ranges of HCB, α-HCH, γ-HCH and α-endosulfan are apparent between the Arctic and Tibetan Plateau; – Atmospheric concentrations of most OCPs in the Arctic, Antarctic and Tibetan Plateau do not show consistent downward trends over the review period (Tables S1, S2 & S3 of the supplementary materials)	Wang *et al.* 2019
Alps	2005 – 2013	3 high Alpine stations in Austria, Germany & Switzerland	Organochlorine pesticides (OCPs)	– OCP concentrations were significantly lower than those in regions adjacent to greater emission sources in other continents (e.g., China, South America) but higher than those in the Arctic – Temporal trend of the single compounds was evaluated: The majority of OCPs did not increase during 2005–2013, but a statistically significant decline was detected during 2006–2013 only for endosulfan and α-HCH	Kirchner *et al.* 2016
West Africa (Togo, Benin, Nigeria, and Cameroon)	2012	25 OCPs 7 sampling sites	Organochlorine pesticides (OCPs)	– High atmospheric concentrations of HCHs and DDTs were found, at average concentrations of 441 pg m^{-3} (range 23–2718) and 403 pg m^{-3} (range 91–1880), respectively – Mirex had the lowest concentrations, ranging between 0.1 and 3.3 pg m^{-3} – There was significant evidence, based on chemical signatures of the contamination that DDT, aldrin, chlordane and endosulfan were recently applied at certain sites	Isogai *et al.* 2016
South Asia	(1987 – 2013; review)	7 research papers	Organochlorine pesticides (OCPs)	– Ambient air concentrations of HCHs ranged from 0.3 to 1690 pg m^{-3}, and from 0.3 – 2900 pg m^{-3} for DDTs – Highly populated and agricultural areas along the India coast were found contaminated with higher levels of OCPs, possibly originating from fresh use of DDT for vector control and agricultural purposes – Elevated concentrations of γ-HCH and HCB also indicated recent use of these pesticides	Ali *et al.* 2014

Region (coverage)	**Monitoring (or publication) period**	**Extent of review/study**	**Pesticides**	**Outcomes**	**Reference**
Arctic	(2008 – 2013; review)	Air and ice cores	Current-use pesticides (CUPs) Organochlorine pesticides (OCPs)	– Ambient air concentrations compiled from reviewed studies: ○ Dacthal: 0.02 – 0.1 pg/m^3 ○ Trifluralin: < LOD – 1.95 pg/m^3 ○ Chlorothalonil: 0.1 – 2.1 pg/m^3 ○ Chlorpyrifos: 0.5 – 2.0 pg/m^3 ○ Endosulfan: α-endosulfan = 0.4 - 130 pg/m^3 ; β-endosulfan: < LOD ○ Methoxychlor: < 0.003 pg/m3 ○ Dicofol: 0.9 – 2.5 pg/m3 – The time trends derived from ice cores show increasing concentrations for chlorpyrifos, endosulfan and trifluralin	Vorkamp & Rigét 2014
Latin America and Caribbean	2014 – 2015 for emerging POPs 2005 – 2015 for other OCPs	9 sampling sites in 7 countries	27 OCPs, including emerging POPs: pentachloroanisole (PCA) and dicofol	– PCA had elevated concentrations at an urban site in Chile (49-222 pg/m^3) and concentrations ranging <1-8.5 pg/m^3 at the other sites in this study – Dicofol indicators were detected at the agricultural site of Sonora (Mexico) at concentrations ranging 30-117 pg/m^3 – Atmospheric concentrations of ɣ-HCH and the endosulfans significantly decreased in this region from 2005 to 2015 – Concentrations of the chlordanes, DDT and PCBs remained similar from 2005 to 2015, suggesting implemented restrictions have not yet led to decreases in atmospheric levels in this region after	Rauert *et al.* 2018
National (selected studies)					
West Antarctica	2010 – 2018	Seven sampling sites in one location	25 organochlorine pesticides (OCPs), congeners and metabolites	– HCB, HCHs, DDTs and endosulfans were detected in most air samples, while chlordane (cis- and trans-), nonachlor (cis- and trans-), oxychlordane, heptachlor, heptachlor epoxide (cis- and trans-), aldrin, dieldrin, endrin, and mirex were not detectable in over 90% atmospheric samples in this study – Relatively low concentrations organochlorine pesticides (OCPs) (Σ13OCPs: 101-278 pg/m3) were found in the atmosphere of West Antarctica – The concentrations of HCHs, DDTs and endosulfans were found to show decreasing temporal trends over the period 2011 – 2017; HCB concentrations remained stable – The atmospheric half-life values for HCHs, DDTs and endosulfans in Antarctic air were estimated as 2.0, 2.4 and 1.2 years, respectively – Long-range atmospheric transport was considered to be the main contributing factor to the atmospheric levels of the POPs in West Antarctica	Hao *et al.* 2019

Region (coverage)	**Monitoring (or publication) period**	**Extent of review/study**	**Pesticides**	**Outcomes**	**Reference**
Brazil	2013-2015 (and comparison with 2008 data)	Two Conservation Unit sites in the Rio de Janeiro state	Current use pesticides (CUPs) and Organochlorine pesticides (OCPs)	– Atmospheric concentrations of legacy OCPs showed background air levels – Endosulfan concentrations decreased between 2008 and 2015 (max concentrations 5500 pg m^{-3} and 2000 pg m^{-3} respectively) – Chlorpyrifos concentrations increased over the same period (max concentrations 40 and 165 pg m^{-3} ,respectively)	Guida *et al.* 2018

Annex 4.3-3 Recent reviews of pesticide concentrations in soils and their risks to soil organisms

Included are reviews published between 2010 and 2021

Region (coverage)	**Monitoring (or publication) period**	**Extent of review**	**Pesticides**	**Outcomes**	**Reference**
Global					
Global	1993 – 2012 (publication)	73 journal publications, globally	p,p'-DDT, p,p'-DDE, HCB, α-, β-, and γ-HCH	– OCP concentrations from agricultural and background land were evaluated – For the period 2003 – 2012 and for all chemicals, the mean concentration in agricultural soil is significantly higher than the concentration in background soil – A decrease in the global mean concentration is discernible for p,p'-DDT and p,p'-DDE from the first to the second decade, but it is statistically significant only for p,p'-DDT in agricultural soil – The mean concentration of HCB in agricultural soil and in background soil does not significantly decrease from the first to the second decade – The 3 HCH isomers decrease in background soils, but not in agricultural soils, from decade 1 to decade 2 – The expected decrease in concentrations of p,p'-DDT and a-HCH in both agricultural and background soil, based on emission modelling, is not supported by measurements. The possibility of new and/or illegal emissions must be kept in mind	Camenzuli *et al.* 2016
Global	2013 – 2017 (publication)	7 studies in 4 countries	Neonicotinoid insecticides	– The current body of evidence shows that detectable levels of neonicotinoids are found in agricultural soils over a year after treated seeds were planted – Neonicotinoids known not to have been recently used can still be present in soils several years after the last application date – Neonicotinoids do not continue to accumulate indefinitely but instead plateau after 2–6 years of repeated application – The studies show that overall, annual sowing of neonicotinoid-treated seed results in chronic levels of soil contamination in the range of 3.5–13.3 ng/g for clothianidin and 0.4–4.0 ng/g for thiamethoxam. This which will act as a constant source of exposure for soil-dwelling organisms, and for neonicotinoid transport into the wider environment	Wood & Goulson 2017

Region (coverage)	**Monitoring (or publication) period**	**Extent of review**	**Pesticides**	**Outcomes**	**Reference**
Global	n.a.	Global review	Fungicides in vineyard soils	– Copper originating from the intensive application of Cu-based fungicides belongs to the most important contaminants of vineyard soils – Concentrations exceed European legislative limits in the majority of the studied vineyards – The intensive application of synthetic organic fungicides has led to increased concentrations of these chemicals in vineyard soils	Komárek *et al.* 2010
Polar regions	1998 – 2015	Review of publications about the Arctic, Antarctic, Tibetan Plateau	DDTs, HCHs, HCB	– Recent ΣHCH concentrations range from n.d. to 5700 pg/g – Recent ΣDDT concentrations range from 20 to 1200 pg/g – Recent HCB concentrations range from 20 – 1500 pg/g – Soil in cold regions are considered as sinks of POPs	Wang *et al.* 2019
Regional					
Europe	2015	317 agricultural topsoil samples from 11 EU Member States and 6 main cropping systems	76 pesticides	– Pesticide residues were present in 83% of the tested soils and 58% of the soils contained multiple residues – Glyphosate and its metabolite AMPA, DDTs and the broad-spectrum fungicides boscalid, epoxiconazole and tebuconazole were most frequently found and the at the highest concentrations – Maximum individual pesticide content was 2.05 mg/kg while maximum total pesticide content was 2.87 mg/kg – Risks to standard soil organisms did generally not exceed acceptable levels. – The presence of mixtures of pesticide residues in soils are the rule rather than the exception	Silva *et al.* 2019
National (selected studies)					
China	Review of papers published since 2000	agricultural soils	OCPs	– OCPs have been widely detected in agricultural soils throughout China – Total concentrations of OCPs ranged from <LOD (limit of detection) to 3520 ng/g, with a mean of 59±53 ng/g – p,p'-DDE exhibited the highest concentration (mean of 15 ± 21 ng/g, n = 66), followed by p,p'-DDT (mean: 13 ± 16 ng/g, n = 66) and chlordane (mean: 8.4 ± 27 ng/g, n = 15). – Most of the concentrations were considered safe under Chinese environmental legislation, though some exceeded acceptable levels – Concentrations of OCPs in almost all recent studies are lower than those that were identified decades ago, indicating that the residues of OCPs have decreased significantly after they were banned for agricultural uses	Sun *et al.* 2018

Region (coverage)	**Monitoring (or publication) period**	**Extent of review**	**Pesticides**	**Outcomes**	**Reference**
China	n.a.	241 agricultural soil sites, across the Yangtze River Delta	9 organo-phosphates	– OPs were present in 93% of samples – Total concentrations of the 9 OPs ranged from <3.0 to 521 ng/g dry weight, with a mean of 64.7 ng/g dry weight – Dimethoate was most detected, followed by methyl-parathion and parathion – There was no significant difference in microbial communities among the sample sites, indicating that OPPs in these soils do not cause significant effects on microbiota	Pan *et al.* 2018
Czech republic	2015	75 arable soils	53 CUPs and 15 transformation products	– Arable soils were found to frequently contain multiple residues of pesticides at noticeable levels several months following the last possible application – In 99% of the soil samples, at least one pesticide was present > LOQ – In 81%, at least one pesticide was present at ≥0.01 mg/kg, the generic soil limit for individual non-chlorinated pesticides in the Czech Republic – The soils contained multiple pesticide residues frequently (51% soils with ≥5 pesticides) – Most frequent detections were of triazine herbicides (89% of soils) and conazole fungicides (73% of soils)	Hvězdová *et al.* 2018
Ghana	2014 – 2015	32 samples from cocoa farms	15 OCPs and metabolites	– 4 OCPs were detected: ○ Lindane in 31.3 % of the soil samples, at a mean concentration of 0.04 ± 0.01 mg/kg ○ beta-HCH in 26.6 % of samples, at a mean concentration of 0.03 ± 0.01 mg/kg ○ dieldrin in 62.5 % of samples, at a mean concentration of 0.02 ± 0.00 mg/kg ○ p,p'-DDT in 50 % of samples, at a mean concentration of 0.03 ± 0.01 mg/kg – Concentrations of OCPs recorded in the soil samples were generally below their respective US MRLs for agricultural soils – The authors conclude that the occurrence of OCP residues in soil may be attributed to the illegal use of the pesticides by farmers or due to their historic use, since OCPs are prohibited from agricultural use in Ghana	Fosu-Mensah *et al.* 2016

Region (coverage)	**Monitoring (or publication) period**	**Extent of review**	**Pesticides**	**Outcomes**	**Reference**
France	2016	180 soil samples from winter cereal crops, grasslands, hedgerows, some of which were untreated or under organic farming. 155 samples of earth-worms (*Allolobophora Chlorotica*) from the same locations.	31 CUPs analyzed (9 insecticides, 10 fungicides, 12 herbicides	– 27 pesticides detected – All the soils contained at least one CUP; 83% ≥ 5 CUPs; and 38% ≥ 10 – The herbicide diflufenican, the insecticide imidacloprid, and the fungicides boscalid and epoxiconazole were most frequently detected – 92% of earthworms contained at least one of the pesticides, and 34% ≥ 5 pesticides. – The long-term, plateau, predicted environmental concentrations in soils (PECs), as defined in EU risk assessment, were exceeded for 11 pesticides (in 94% of samples) – A high risk of chronic toxicity to earthworms was found in 46 % of samples, both in treated cereals and nontreated habitats considered as refuges	Pelosi *et al.* 2021
French West Indies (Guadeloupe)	2003 – 2009	Soil samples from 1045 plots from (former) banana plantations.	Chlordecone; applied between 1972 and 1993	– Chlordecone concentrations ranged from 0.011–52.1 mg/kg of dry soil – 8% of plots contained <0.1 mg chlordecone/kg soil; i.e. the threshold below which no MRL exceedance in subsequent root crops is expected – 34% of plots contained > 1.8 mg chlordecone /kg soil; i.e. the threshold above which almost all subsequent root crops are expected to exceed the MRL – High chlordecone residues in soil were thus present 10 to 35 years after its application	Levillain *et al.* 2012
India (Assam region)	2009	175 surface soil samples from agricultural fields, fallow and urban lands	14 OCPs and metabolites	– Mean concentrations of total HCHs ranged from 705 – 825 ng/g and of total DDTs from 757 – 903 ng/g – Residue levels were among the highest found so far in India; soils from paddy fields contained most HCH and DDT residues – DDT and HCH soil residues originated both from the long past as well as from recent sources (vector control and agricultural use)	Mishra *et al.* 2012
Kazakhstan, Poland	2012 - 2014	Poland (89 samples), Kazakhstan (32 samples)	OCPs	– DDT, DDE and HCHs were detected in 20 – 50% of soil samples – Average total concentrations of DDT in Polish and Kazakh soil samples were 0.104 and 0.097 mg/kg respectively – The authors conclude that, even though concentrations were relatively low, OCPs continue to be a ubiquitous contaminant in soils	Łozowicka *et al.* 2016

Region (coverage)	**Monitoring (or publication) period**	**Extent of review**	**Pesticides**	**Outcomes**	**Reference**
Nepal	2017	147 soil samples from agricultural land	12 CUPs and degradation products & 8 OCPs	– Pesticides were detected in 60% of the soil samples (25% had a single residue, 35% had multiple residues) – Overall, the concentration of pesticides ranged from 1.0 to 251 µg/kg, with a mean of 16 µg/kg – Residues of dichlorvos, dimethoate, omethoate, phorate, α-γ-HCH and α-β-endosulfan were below LOD. Chlorantraniliprole and imidacloprid residues were the most frequent; chlorpyrifos and p,p'-DDT were detected in the highest concdentrations – The ionic ratio of DDT and its degradation products suggested a continuing use of DDT in the area – Low/no pesticide concentrations were detected in soils from IPM fields	Bhandari *et al.* 2020
Switzerland	2015	702 soil in 169 cultivated fields over the entire lowland of Switzerland	5 neonicotinoid insecticides (imidacloprid, clothianidin, thiamethoxam, thiacloprid, acetamiprid)	– Neonicotinoids were detected in 93% of organic soils and more than 80% of soils in "ecological focus areas" (EFAs)—two types of arable land supposedly free of insecticides – Between 5%–9% of above-ground invertebrate species may be exposed to lethal concentrations of clothianidin, and 32%–41% to sublethal concentrations, in "integrated production" and conventional fields – Between 1%–7% of the beneficial invertebrate species may be exposed to sublethal concentrations of neonicotinoids in EFAs and organic fields – The authors conclude that that diffuse contamination by neonicotinoids may harm a significant fraction of non-target beneficial species. The use of neonicotinoids on crops may threaten biodiversity in refuge areas, while also potentially jeopardizing the practice of organic farming by impeding the biological control of pests	Humann-Guilleminot *et al.* 2019
Tanzania	no period	Literature review	Mainly OCPs	– Most studies analyzed OCPs – In virtually all samples, OCPs were detected, in spite of the fact that the use of most of these have been prohibited for a long time – Concentrations exceeding clean-up action levels occurred in former pesticide storage locations	Elibariki & Maguta 2017

Annex 4.3-4 Recent reviews of pesticide concentrations in groundwater and drinking water.

The studies were selected from a literature search conducted using Web of Science for papers from 2010 until June 2020 using the following search criteria: TS=(pesticide*) AND TS=(groundwater* OR subterranean OR drinking). The title and summary of the resulting 3095 hits were evaluated, of which 175 papers were downloaded and analysed in more detail.

Region (coverage)	**Monitoring (or publication) period**	**Extent of review** (GW = groundwater; DW = drinking water)	**Pesticides**	**Outcomes**	**Reference**
Global					
Global (21 countries)	Studies published between 2001 and 2017	73 studies (GW)	Pesticides form the 1st and 2nd EU watchlist (5 neonicotinoids; 2 other insecticides; 2 herbicides)	– Most studies from USA, followed by EU. Few data for other regions – Among the 5 neonicotinoids, imidacloprid was the most frequently studied and detected – Average imidacloprid concentrations ± 8 ng/L to 3 µg/L (10 studies) – Other neonicotinoids were only observed in 1 or 2 water resources – Only one study available for the other pesticides from the watchlists (8 µg oxadiazon/L in Italy)	Pietrzak *et al.* 2019
Global (6 countries)	2000 - 2015	13 studies (GW)	Organochlorines and organophosphates	– Most studies conducted in Asian countries – Diversity and concentrations lower than in surface waters – Concentrations of organophosphates were generally higher than organochlorines due to their widespread use	Pirsaheb *et al.* 2017
Global (not reported)	Studies published between 2004 and 2014	12 studies (GW)	Pesticide transformation products (TPs)	– Few pesticide TPs are usually included in monitoring studies – TPs of atrazine and terbutylazine are the most studied TPs – The most abundant pesticide TPs in groundwater derive from the chloroacetanilide herbicides acetochlor, alachlor and metolachlor, the s-triazine herbicides atrazine and terbutylazine, and the herbicides chloridazon and dichlobenil – TPs derived from banned pesticides have consistently been detected in groundwater due to their their long residence time in the subsurface or the slow release of their precursors from the soil – Pesticide TPs are occasionally more ubiquitous and abundant than their parent molecules – Pesticide TPs have occasionally been quantified at levels above the EU groundwater quality standard of 0.1 µg/L	Postigo & Barceló 2015

Region (coverage)	**Monitoring (or publication) period**	**Extent of review** (GW = groundwater; DW = drinking water)	**Pesticides**	**Outcomes**	**Reference**
Regional					
Africa	Studies published between 2000 and 2018	NR (130 pesticide detections in freshwater resources combined for 13 countries) (GW + DW)	Review on contaminants of emerging concern including pesticides. Data available for 11 insecticides and 8 herbicides	– Despite an increase in the last decade, information is available for only few compounds and countries – In many African countries, groundwater is mostly used for drinking without any treatment – Pesticide concentrations detected in groundwater and drinking water ranging 0.1 ng/L to 18.4 µg/L and 0.02 ng/L to 34 µg/L, respectively	K'oreje *et al.* 2020
6 countries in Europe, Africa and Oceania (Germany, Australia, France, South Africa, the Netherlands and Spain)	November 2015	One (duplicated) drinking water sample per country (DW)	11 pesticides (8 herbicides, 2 insecticides, 1 fungicide)	– None of the pesticides were detected in Germany, Australia and South Africa – Atrazine was detected in France and Spain; Simazine in the Netherlands and Spain; Diuron in Spain – The maximum pesticide concentration detected (37 ng atrazine/L in Spain) was below the EU standard	Leusch *et al.* 2018
Overseas departments of France (Guadeloupe, Martinique, Reunion, Mayotte and French Guiana)	April - June 2012 (1st campaign); September 2012 - January 2013 (2nd campaign)	80 samples during two campaigns (40 sampling points/campaign) (GW)	Review on 66 target organic micropollutants (including 1 synergist and 16 pesticides: 8 herbicides, 4 insecticides, 3 fungicides and 1 metabolite)	– 11 pesticides detected, with greatest detection frequencies for the herbicides pentachlorophenol (55%) and atrazine (38%) – Several of the pesticide concentrations exceed the EU groundwater quality standard: pentachlorophenol (9 samples; max. 418 ng/L), atrazine (3 samples; max. 420 ng/L), diuron (1 sample; 258 ng/L) and fenarimol (1 sample; 121 ng/L)	Vulliet *et al.* 2014
National (selected country-wide or review studies)					
Sweden	Winter of 2018 (Jan-Feb)	Nationwide: 90 water treatment plants geographically distributed all over Sweden (DW)	3 herbicides (bentazon, atrazine, quinmerac)	– Concentrations of atrazine and bentazon were higher than in surface water – Atrazine was detected in all samples, whereas quinmerac and bentazon were not or rarely detected, respectively	Karki *et al.* 2020

Region (coverage)	**Monitoring (or publication) period**	**Extent of review** (GW = groundwater; DW = drinking water)	**Pesticides**	**Outcomes**	**Reference**
China	June 2019	Nationwide: 789 tap water samples and 95 groundwater samples (GW + DW)	Herbicides 2,4-D and MCPA	– MCPA concentrations were generally higher than those of 2,4-D – The cumulative concentration of the 2 herbicides in tap water of China was up to 125 ng/L (median: 1.38 ng/L) – Higher concentrations were found in surface water derived tap water than in groundwater – The median cumulative concentration in groundwater was approximately five times lower than that in tap water. – The highest concentrations were observed in July, and the lowest in October (related with pesticide applications) – The highest concentrations were found in Northeast China and were related with intensive herbicide use	Sun *et al.* 2020
China	June 2019	Nationwide: 789 tap water and 95 groundwater samples (GW + DW)	Atrazine and its metabolites (ATZs)	– At least one of the analytes was found in all the water samples – The median sum concentrations of ATZs (sATZs) was 21.0 ng/L (range: 0.02 ng/L - 3.04 mg/L) – The level of ATZs in groundwater from rural area of China was about 1/3 of that found in tap water	Wang *et al.* 2020
Saudi Arabia	NR	Nationwide: 993 well water samples from all 13 administrative regions (GW)	22 organochlorine insecticides	– Pesticides were not detected in any of the well water samples – However, previous studies reported pesticide residues in groundwater of the country	Alquwaizany *et al.* 2019
Martinique	2009–2014	Nationwide: 282 samples from 21 sites (twice a year at 18 sites; monthly in some years at 3 sites) (GW)	Chlordecone and its metabolite chlordecone-5b-hydro (5bCLD)	– Chlordecone decreased or remained stable during the time period – 5bCLD showed an increasing, although rather erratic, evolution, probably due to the greater influence of hydrological conditions – Most concentrations were < 0.1 µg/L with higher concentrations in banana cropping areas	Cattan *et al.* 2019
Bangladesh	not limited	National review: 3 studies (GW)	7 insecticides (organochlorines and organophosphates)	– DDT (and metabolites) encountered in concentrations up to 1.2 mg/L – High DDT concentrations encountered years after ban, which may also be partly due to an apparent illegal use – Little information available, but groundwater pollution is likely since groundwater levels in Bangladesh are high, and soil is mainly coarse	Hasan *et al.* 2019

Region (coverage)	**Monitoring (or publication) period**	**Extent of review** (GW = groundwater; DW = drinking water)	**Pesticides**	**Outcomes**	**Reference**
Spain	Studies including monitoring data from 1993-2012	National review: 12 studies (GW)	EU Watchlist pesticides (5 neonicotinoids, 2 carbamates, 1 oxadiazole and 1 semicarbazone)	– Only imidacloprid and methiocarb were detected in the GW bodies of Spain. However, this may also be at least partly due since the other pesticides were only studies in 1 or 2 cases. – Mean imidacloprid and methiocarb concentrations generally below, and maximum concentrations above 0.1 µg/L	Jurado *et al.* 2019
India	Studies between 2001 and 2015	National review: 12 studies (GW)	10 insecticides (organochlorines and organophosphates)	– Average concentrations of almost all insecticides and in almost all studies > 0.1 µg/L – Since pesticide use is expended to increase, groundwater pollution is likely to increase even further	Malyan *et al.* 2019
India	Studies since 1995	National review (45 studies including other micropollutants) (GW)	Insecticides (organochlorines (OCP), organophosphates (OPP) and synthetic pyrethroids (SP))	– Vast majority of studies for OCP, followed by OPP. Only few studies with SP – Many (maximum) concentrations > 0.1 µg/L – Higher concentrations during/after monsoon season	Sackaria & Elango 2019
The Netherlands Part also Belgium/Flanders	Routine monitoring data: 2010 – 2014: monitoring campaign: May and June 2016	Routine monitoring data: 29,766 records from all 10 ten water companies (GW) Monitoring campaign: 90 groundwater locations used as drinking water source in The Netherlands and Flanders (GW)	408 pesticides (routine monitoring data) + 24 pesticides (monitoring campaign): 4 herbicides, 12 fungicides, 6 insecticides, 1 acaricide and 1 plant growth regulator	– 15 out of 24 recently authorized pesticides were detected, including neonicotinoids – Pesticides (and/or metabolites) were detected in the majority of drinking water sources. – In one third of the abstraction areas pesticide and/or metabolites concentrations exceeded water quality standards according to the WFD – Monitoring should focus on priority pesticides and be kept up to date with recently authorized pesticides	Sjerps *et al.* 2019
Italy	2010 - 2015	11 alluvial aquifers of central Italy (GW)	42 pesticides (16 insecticides; 18 herbicides; 8 fungicides)	– Increasing trends in pesticide concentrations over the study period were encountered for several pesticides – Some pesticides banned in 2009 showed a persistent occurrence – Mean concentrations were > 0.1 µg/L for 3 pesticides	Di Lorenzo *et al.* 2018

Region (coverage)	**Monitoring (or publication) period**	**Extent of review** (GW = groundwater; DW = drinking water)	**Pesticides**	**Outcomes**	**Reference**
Ireland	Monthly sampling between March 2010 to March 2012	Nationwide: 7 sites (GW)	10 pesticides and 6 of their metabolites	– 2 metabolites were encountered in higher concentrations than their active ingredients – Highest detections were encountered in sites with well drained soils underlain by gravel and limestone aquifers and within gravel lenses in lower permeability subsoil – Metabolites accounted for the majority of exceedances of 0.1 µg/L	McManus *et al.* 2017
England and Wales	Since 2003 with a small number of samples collected from 1992 onwards.	Nationwide: 2650 sites as part of the England and Wales Environment Agency's National Monitoring Programme (GW)	15 pesticides (including metabolites)	– Atrazine, its metabolite atrazine-desethyl and simazine in the top 5 out of > 800 organic compounds – Maximum concentrations detected for diazinon (> 150 µg/L) and chlorotoluron (> 30 µg/L)) – Although a large monitoring dataset, it did not provide sufficient data for any compound for one site to determine a trend. – However, there appeared to be a declining overall trend in both frequency and concentrations of atrazine detections from 2004 to 2009	Manamsa *et al.* 2016
France	2011: campaigns in the spring (485 sites) and fall (475 sites)	Nationwide: 494 groundwater sites (GW)	103 pesticides (48 herbicides, 29 fungicides and 26 insecticides)	– 43 pesticides were detected in at least 1 sample, with 14 and 3 (atrazine and 2 of its metabolites) of those in at least 10 and 96 samples, respectively – Detected pesticides were dominated by herbicides (68% of sites), followed by fungicides (7.5%) and insecticides (1.4%) – The ratios between the occurrence of parent molecules and metabolites varied depending on the pesticide – 32 pesticides (including metabolites) exceeded the legal standard at least once. More than half of these compounds have already been banned	Lopez *et al.* 2015
South Africa	February, April, July and October 2012 (4 seasons)	Water treatment plants in 7 major South African cities (DW)	Qualitative screening of 700 compounds (including several pesticides); quantitively screening of 2 pesticides (atrazine and terbuthylazine)	– Most frequently detected pesticides in the qualitative screening were atrazine (63%), metolachlor (75%) and terbuthylazine (100%) – The seasonal distribution of atrazine fitted with its agricultural use as herbicide for summer crops – Detected levels of atrazine and terbuthylazine were at least an order of magnitude less that the guideline values set by EPA and WHO, respectively – However, detected levels were > 0.1 µg/L in many occasions at several sampling sites	Odendaal *et al.* 2015

Region (coverage)	**Monitoring (or publication) period**	**Extent of review** (GW = groundwater; DW = drinking water)	**Pesticides**	**Outcomes**	**Reference**
Hungary	1990–2015	Nationwide: seven monitoring projects in over twenty sampling campaigns (GW)	49 pesticides (including metabolites)	– Contamination rates of 32–61% were denoted in the monitoring projects – Standards were rarely exceeded; only in the cases of point contaminations, where higher concentrations were determined for atrazine (concentrations up to 7540 ng/L)	Székács *et al.* 2015
India	NR (studies published between 2005 and 2014)	National review (10 studies) (GW)	Persistent organic pesticides (DDTs, HCHs, endosulfan)	– Concentrations up to 0.94 µg/L (DDTs), 1.1 µg/L (HCHs) and 8.9 µg/L (endosulfan) have been detected in groundwater – High concentrations of HCHs and endosulfan isomers and DDT metabolites have been detected across India	Yadav *et al.* 2015
Spain	4 consecutive years: 2007 (June–November), 2008 (June–November), 2009 (May–October) and 2010 (April–September).	265 samples from 112 wells and piezometers coming from 29 different aquifers located in 18 ground waterbodies from Catalonia (GW)	22 pesticides (18 herbicides and 4 insecticides)	– Pesticide concentrations in groundwater tend to decrease with time. – Several aquifers had pesticide concentrations that exceeded EU quality standards. – Pesticides that exceeded 0.1 µg/L more than once were atrazine, terbuthylazine and the triazine metabolites DEA and DIA	Köck-Schulmeyer *et al.* 2014
Ireland	2007 - 2008	Nationwide: 845 samples from 158 monitoring points (GW)	13 pesticides (9 herbicides, 4 insecticides)	– MCPA and mecoprop were the most frequently detected pesticides in groundwater – Mecoprop and diuron concentrations > 0.1 µg/L were noted. In addition, all detected 2,4-D and bentazone concentrations were > 0.1 µg/L	McManus *et al.* 2014
United States	1993–2011	Nationwide: 2542 samples from 1271 wells in 58 nationally distributed well networks (GW)	83 pesticides	– Pesticides were frequently detected (53% of all samples) – Concentrations seldom exceeded human-health benchmarks (1.8% of all samples) – For agricultural networks, concentrations of atrazine, metolachlor, and prometon decreased in time. – For urban networks, deethylatrazine concentrations increased and prometon concentrations decreased. —For major aquifers, concentrations of deethylatrazine and simazine increased.	Toccalino *et al.* 2014

Region (coverage)	**Monitoring (or publication) period**	**Extent of review** (GW = groundwater; DW = drinking water)	**Pesticides**	**Outcomes**	**Reference**
Lebanon	DW: December 2010 - December 2011 GW: 5 times between November 2010 and November 2011	Nationwide (GW + DW)	67 pesticides from various chemical classes	– Most frequently detected pesticides in both groundwater and drinking water samples were diazinon, DDE and DDD, hexachlorobenzene, aldrin, dieldrin and tetradifon – All were quantified at low levels (< 20 ng/L)	Kouzayha *et al.* 2013
New Zealand	September - December 2010	National review: 162 wells in 14 regions (GW)	22 organochlorines; 4 organophosphates; 23 organonitrogen herbicides; 17 acid herbicides	– Pesticides were detected in 38 wells (23%), with two or more pesticides detected in 15 wells (9%) – A total of 22 different pesticides were detected with herbicides (n = 17) as the most common pesticide group, followed by insecticides (3) and fungicides (2) – Concentrations of 9 pesticides exceeded 0.1 µg/L in 21 cases (7% of wells with pesticides detected). – Comparisons with earlier surveys (since 1990) indicate that a similar percentage of wells had pesticide detections at comparable levels and with the same types of pesticides	Close & Skinner 2012
Pakistan	1990 - 2010	Nationwide: 6 studies (GW)	Organochlorines	– HCH isomers around 0.1 µg/L (max. 16.7 µg/L) – Concentration ranges: heptachlor 0.02 – 0.17 µg/L (1 study); endrin 0.1–0.2 µg/L (1 study); dieldrin 0.06 µg/L (1 study); DDT traces – 400 µg/L (4 studies); DDT isomers traces – 0.82 µg/L (3 studies) – High DDT levels attributed to dumping, illegal use and (inappropriate) storage	Eqani *et al.* 2012

Region (coverage)	**Monitoring (or publication) period**	**Extent of review** (GW = groundwater; DW = drinking water)	**Pesticides**	**Outcomes**	**Reference**
Spain	Studies up to 2012	National review (GW)	80 pesticides (herbicides, insecticides and fungicides)	– The most studied pesticides have been triazines followed by phenyl urea herbicides), anilides and organophosphorus insecticides – Out of the 80 studied pesticides, 61 were reported in < 4 studies and 30 were not detected in any groundwater sample – Groundwater is considerably less contaminated than other water bodies – (max.) concentrations of many pesticides > 0.1 µg/L, including alachlor (9.95 µg/L) and metolachlor (5.37 µg/L), malathion (3.5 µg/L), atrazine (3.45 µg/L), chlorfenvinphos (2.5 µg/L), dimethoate (2.3 µg/L), DEA (1.98 µg/L), chlortoluron (1.7 µg/L), simazine (1.69 µg/L); parathion-methyl (1.5 µg/L), TBA (1.27 µg/L) and linuron (1.01 µg/L)	Jurado *et al.* 2012
Pakistan	Studies from 1988 to 2006	National review: 8 studies (GW)	Several (mostly insecticides)	– High variation in detection frequencies (between 0% and 100%) depending on the pesticide and area – Cotton growing areas showed higher pesticide detections and concentrations (often greater than 0.1 µg/L) – Higher pesticide groundwater concentrations probably related with the existence of ponding irrigation systems	Tariq *et al.* 2007

Annex 4.3-5 Recent reviews and large scale studies on the effects of pesticide use on vertebrate wildlife

Included are reviews and studies published between 2010 and 2020

1. Recent reviews or large scale studies on the effects of pesticide use on wild bird (and some mammal) populations.

Country	Study type	Study coverage	Organisms	Effects observed	Reference
Global					
Global	Systematic review	Scientific evidence published since 2013 about the impact of neonicotinoids on nontarget organisms	Birds	– There is considerable variation in the lethality of neonicotinoids to birds, both among bird species and individual pesticides, ranging from non-toxic to highly toxic – It is difficult to judge whether clearly demonstrated lethal and sublethal effects of seeds treated with certain neonicotinoids are manifested in wild bird populations in the field	Wood & Goulson 2017
Global	Systematic review	Direct and indirect effects of neonicotinoids and fipronil on vertebrate wildlife. Mainly laboratory-based studies available.	Mainly wild birds	– Exposure to seeds treated with certain neonicotinoids at dosages expected in the field was found to pose a risk to granivorous birds	Gibbons *et al.* 2015
		Two field studies on bird populations identified: 1 of imidacloprid and 1 of fipronil.	Insectivorous birds	– No reductions in bird abundances were observed in both studies	
		One field study of effect of fipronil use in locust control on mammals	Lesser hedgehog tenrec	– Decline in tenrec populations, likely due to depletion of termite prey	
Global	Systematic review	27 farmland bird and 22 farmland mammal species. 57 scientific publications.	Farmland birds & mammals	– Evidence for indirect effects of pesticides at the population level of 4 European farmland bird species – Studies on long-term populations effects of pesticide use in agriculture on mammals are lacking, but some studies show adverse short-term effects – Pesticides were considered to be among the major causes for population declines of farmland birds	Jahn *et al.* 2014
Global	Narrative review	65 articles addressing relationship between rice cultivation and impacts on aquatic birds, published from 1980s – 2010	Aquatic birds in rice habitats	– Direct mortality of many avian species documented as a result of rice pesticide applications – Pesticides cited were organochlorines, organophosphates and carbamates – Food resources of birds affected by pesticide applications in rice, but no resulting population effects cited	Parsons *et al.* 2010

Country	Study type	Study coverage	Organisms	Effects observed	Reference
Regional					
North America	Narrative review	Factors assessed: declines in aerial insects, contamination (incl. pesticides), breeding habitat loss, climate change, nonbreeding ground effects	Aerial insectivore birds	– While no single threat appears to be the cause of aerial insectivore declines, existing evidence suggests that several factors could be contributing to the declines – As general hypothesis, the authors propose that changes in availability of high-quality prey, due to complex interactions of multiple effects (e.g. broad application of pesticides, agricultural intensification and other land use changes, changes in climate), have variably impacted combinations of fledging success, post-fledging survival, and nonbreeding season body condition of aerial insectivores, all of which contribute to variations in population trends	Spiller & Dettmers 2019
North America	Systematic review	122 studies, covering 77 breeding bird species. Parameters assessed (among others); pesticide lethal and sublethal toxicity	Farmland birds	– Pesticides were the most predominant factor negatively affecting farmland bird populations (42% of all studies) – Aerial insectivores and shrubland species were most commonly affected by pesticides (64% of studies for aerial species, 70% for shrubland) – Pesticides affected food availability of birds (100% of studies), their survival (43% of studies) and reproduction (26% of studies)	Stanton *et al.* 2018
Estonia, France, Germany, Ireland Netherlands, Poland, Spain, Sweden	Large-scale monitoring	270 cereals farms. 13 components of agricultural intensification assessed, among which: Number of applications and total amount of a.i. applied of insecticides, herbicides, fungicides.	Breeding birds	– Application of fungicides (and correlated insecticides) was significantly and negatively associated with the total abundance of all breeding birds surveyed, as well as four individual breeding bird species. – Pesticides did not have any associations with the abundance of birds during winter, suggesting that effects of insecticides or fungicides occurred through impact on food supply during the nestling period.	Emmerson *et al.* 2016
National					
USA	Surveys & models	Population abundance of 29 species of grassland birds, 637 species of non-grassland birds, 36 species of insectivorous birds and 631 species of non-insectivorous birds. Annual mass of individual neonicotinoids applied at county-level	Birds	– The use of neonicotinoids resulted in annual decreases of 4% and 3% in population abundance of grassland birds and insectivorous birds, respectively, over the period 2008 – 2014 – When the longer-term effects of neonicotinoids on bird populations were considered, through their reduction of the number of birds left to reproduce, the average annual effect on grassland birds was 12% and on insectivorous birds 5% – The average annual reduction in non-grassland and non-insectivorous bird populations due to neonicotinoid use was about 2%	Li *et al.* 2020

Country	Study type	Study coverage	Organisms	Effects observed	Reference
USA	Narrative review	Direct and indirect effects of pesticides, and other drivers, on bird populations in the USA.	Birds	– The hypothesis that pesticides, particularly neonicotinoids, are indirectly affecting bird population status via reductions in food resources should be considered with caution and within context of other more likely causes – While it would be disingenuous to suggest that insecticides do not pose any potential risk to non-target arthropods proximal to insecticide use, and thus indirectly to avian species, it would also be misleading to suggest that these compounds are the primary driver for such indirect effects	Brain & Anderson 2019
USA	Surveys 1978 – 2012	Bobwhite abundance related to county-level total neonicotinoid use, in Texas. Other variables assessed: climate and land-use data.	Northern bobwhite (grassland bird)	– Of the six predictor variables tested in this study, the strongest negative association was between bobwhite abundance and neonicotinoid use – Neonicotinoid use was significantly negatively associated with bobwhite abundance in the time periods following neonicotinoid introduction (1994-2003) or after their widespread use (2004-2012) – Treated seeds probably present the biggest hazard to bobwhites and other granivorous species	Ertl *et al.* 2018
USA (California)	Surveys 1914 – 2013 1990 – 2013 for pesticide data	Abundance index based on 3 sets of bird survey data related to county-level total pesticides applied. Other variables assessed: land-use, harvested crop land, predator abundance, climate data.	Ring-necked pheasant	– Major changes in agricultural practices over the last three decades were associated with declines in pheasant numbers and likely reflected widespread loss of habitat – Recent negative impacts on pheasant numbers were also associated with relatively high levels of pesticide application	Coates *et al.* 2017
USA	Surveys 1980 – 2003	Number of breeding bird species; Fraction of farmland treated with herbicide or insecticide; Lethal pesticide risk	Grassland birds	– Best predictors of bird species declines were the (acute) lethal risk from insecticide use modelled from pesticide impact studies, followed by the loss of cropped pasture – A re-analysis of data by Hill *et al.* suggested that population trends of grassland birds in the U.S. were primarily associated with habitat availability, rather than insecticide use or insecticide acute toxicity	Mineau & Whiteside 2013 Hill *et al.* 2014
The Netherlands	Surveys 2003 – 2010	Imidacloprid concentrations in surface water. 15 passerine species that are common in farmlands and depend on invertebrates during the breeding season.	Farmland birds	– Higher concentrations of imidacloprid in surface water were consistently associated with lower or negative population growth rates of passerine insectivorous bird populations	Hallman *et al.* 2014

Country	Study type	Study coverage	Organisms	Effects observed	Reference
France	Large-scale monitoring	66 cereal fields located in three French departments. Bird abundance	Farmland and non-farmland birds	– Herbicide use intensity was correlated with higher total abundance of birds. – However, the abundance of herbivore specialists decreased with herbicide use intensity. This suggests that herbicides may contribute to the 'biotic homogenization' by replacing specialists with generalists.	Chiron *et al.* 2014
Canada	National estimates	Direct avian mortality from agricultural pesticides in Canada. Reproductive and indirect effects of pesticides were not included.	Breeding birds	– Pesticides were 6th source of bird mortality in Canada (out of 28 sources identified), after feral and domestic cats and collisions with power lines, houses and vehicles, resulting in 0.9 – 4.4. million casualties per year. This was 1 – 1.5 orders of magnitude lower than the first 5 sources. – Impact of this mortality on bird populations was not known, but may have significantly contributed to the decline of grassland/farmland bird species in North America (Mineau 2013).	Calvert *et al.* 2013 Mineau 2013
France	Monitoring study 2003 – 2005	Breeding bird abundance in 15 apple orchards under conventional, IPM and organic pest control, in south eastern France. Orchards were standardised for local and landscape features that might influence bird communities.	Breeding birds	– Birds were significantly more abundant in organic orchards than in IPM and conventional orchards. Bird abundance in IPM orchards was significantly higher than in conventional orchards in 2 out of 3 years. – The pest control strategy affected insectivores more than granivores.	Bouvier *et al.* 2011

2. Recent reviews of the risks and/or effects of pesticides to amphibians at environmentally relevant concentrations.

Country	Study type	Study coverage	Organisms	Effects observed	Reference
Global					
Global	Systematic review & meta-analysis	Behavioural changes induced by chemicals, including pesticides 116, mainly laboratory-based, studies	Amphibians	– Most studies focussed on insecticides – Insecticides generally invoked the strongest behavioural responses: they significantly increased rates of abnormal swimming, reduced activity and reduced the capacity to exhibit escape responses – *Note:* it was not specified whether effect concentrations were environmentally relevant	Sievers *et al.* 2019
Global *(but focus on Europe)*	Narrative review	Evidence of pesticide effects on amphibians under current conditions use	Amphibians	– The review concludes that exposure of amphibians does occur, and that this exposure may lead to decline of populations and harm individuals, which would be of high concern. Therefore, a specific environmental risk assessment scheme is needed for these groups	EFSA 2018

Country	Study type	Study coverage	Organisms	Effects observed	Reference
Global	Narrative review	Effects of endocrine disrupting chemicals (EDCs), including pesticides, in laboratory experiments and field monitoring	Amphibians, primarily frogs & toads	– Effects on reproduction (laboratory): Exposure to the herbicide atrazine in leopard frogs induce feminising effects on the male gonads (including 'intersex' – oocytes within testes) at concentrations measured in some aquatic environments – There is now a substantial and growing body of evidence from field studies indicating that agriculture, herbicides and/or pesticides are associated with increased intersex in anurans. Some of the strongest associations between intersex induction and EDCs come from studies that have analysed the effects of the agricultural environment or mixtures of agrochemicals, rather than those focused on single compounds – There are significant knowledge gaps on the effects of EDCs in amphibians and data are lacking for studies on wild populations. This prevents drawing definitive conclusions on the associations between endocrine disruption and wild amphibian populations	Orton & Tyler 2015
Global	Systematic review & meta-analysis	Effects of pesticides on survival and growth 66 survival studies & 45 growth studies, primarily laboratory-based	Amphibians	– At ecologically relevant concentrations: – Organophosphates, carbamates, neonicotinoids, phosphonoglycines (with POEA surfactant), triazines, adversely affected amphibian survival – Organophosphates and phosphonoglycines also adversely affected amphibian growth	Baker *et al.* 2013
Global	Systematic review & meta-analysis	Effects of pesticides on survival, mass, time to hatching, time to metamorphosis and frequency of abnormalities. 16 laboratory & mesocosm studies	Amphibians	– At ecologically relevant concentrations: o Pesticide exposure resulted in significant reduction of amphibian survival. o Effects of pesticides on body mass, developmental time and frequency of abnormalities were not statistically significant, partly due to high heterogeneity among studies.	Egea-Serrano *et al.* 2012
National					
Argentina	Field monitoring	Pesticides: cypermethrin, chlorpyrifos, endosulfan, glyphosate and 2,4-D. Enclosures in 91 ponds adjacent to agricultural field were monitored	4 amphibian species common in ponds in agricultural landscapes	– Applications to adjacent fields with the 3 insecticides reduced survival at 48h to <1 – 10%. No significant effects on acute survival observed for the most application with only herbicides – All pesticides, including herbicides, impaired the mobility of the amphibians (tadpoles)	Agostini *et al.* 2020

Country	Study type	Study coverage	Organisms	Effects observed	Reference
USA	Field monitoring	Samples of water, sediment and frog tissue from 21 sites in 7 states. Samples analyzed for >90 pesticides and metabolites, and for the amphibian chytrid fungus *Batrachochytrium dendrobatidis* (Bd)	Frogs	– Thirty-nine pesticides and pesticide degradates were detected in one or more of the samples collected during the study – Pesticide occurrence in water or sediment was not a strong predictor of occurrence in tissue, but pesticide concentrations in tissue were correlated positively to agricultural and urban land, and negatively to forested land – Frogs that tested positive for Bd were associated with sites that had higher total fungicide concentrations in water and sediment, but lower insecticide concentrations in sediments relative to frogs that were Bd negative – There were no other differences in pesticide occurrence between individual frogs that were positive or negative for Bd	Battaglin *et al.* 2016
United Kingdom	Field monitoring (1992 – 2000)	Ranavirus-consistent common frog mortality events in the UK in relation to the use of garden chemicals	Common frog (*Rana temporaria*)	– Ranavirosis occurrence (i.e. presence/absence) was not significantly associated with use of garden chemicals – Ranavirosis prevalence(i.e. proportion of the estimated total frog population killed in the mortality event) was positively associated with the use of herbicides and of slug pellets, in addition to several other abiotic and biotic variables	North *et al.* 2015

3. Recent reviews of the risks and/or effects of pesticides to reptiles at environmentally relevant concentrations.

Country	Study type	Study coverage	Organisms	Effects observed	Reference
Global					
Global	Narrative review of toxicity studies	16 articles of laboratory experiments of the toxicity of pesticides to lizards	Lizards	– Most studies assessed locomotor performance, histopathology, oxidative stress, neurotoxicology, and genetic damage – It was not clear from the review whether observed effects would occur at environmentally relevant concentrations	Freitas *et al.* 2020
Global (*but focus on Europe*)	Narrative review	Evidence of pesticide effects on reptiles under current conditions use	Reptiles	– Few laboratory and field studies of pesticide effects on reptiles are available. Some studies indicate that adverse effects on reptiles occur of pesticides at environmentally relevant pesticide concentrations, but other studies do not find such effects – However, the review concludes that exposure of reptiles does occur, and that there is evidence that active substances and authorised plant protection products in Europe do have toxic impacts on reptiles	EFSA 2018

Country	Study type	Study coverage	Organisms	Effects observed	Reference
Regional and national					
Europe	Assessment of risk of exposure, in Europe	Exposure risk is estimated based on occurrence of reptiles in agricultural areas where pesticides are used, reptile physiology and their life-history	Reptiles	– About 1/3 of all European species of reptiles exhibit a high risk of exposure to pesticides. This is especially the case in Southern Europe, with a particular focus on vineyards – Of the 21 species and 3 subspecies of reptiles with special conservation status in the EU, nearly half are at above-average risk of pesticide use	Mingo *et al.* 2016 Wagner *et al.* 2015
Germany	Field monitoring study	Fitness and enzymatic activity in lizards sampled from sites along a gradient of agricultural intensity (vineyards with mainly fungicide applications), compared to a natural, unexposed, site	Common wall lizard (*Podacris muralis*)	– Exposure to pesticides induced oxidative stress in wall lizards, which can have severe implications for individuals – Neurotoxic effects were not observed – An evident decrease in fitness of individuals was observed with increasing land use intensity (and presumed pesticide exposure)	Mingo *et al.* 2017
New Zealand	Laboratory studies	Oral acute toxicity of rodenticides (brodifacoum, coumatetralyl, pindone, diphacinone, cholecalciferol) and herbicides (glyphosate, clopyralid, triclopyr, metsulfuron-methyl, haloxyfop-methyl) Dietary risk assessment	Western fence lizard (*Sceloporus occidentalis*)	– Only pindone and triclopyr were toxic to fence lizards below 1750 µg/g (LD_{50} = 550 µg/g) – Acute dietary risk to reptiles to the tested rodenticides and or herbicides was low – Research into sub-lethal effects is also required	Weir *et al.* 2016
Portugal	Field monitoring study	4 sites under intensive agriculture compared to 2 sites under organic agriculture. Mixture of herbicide treatments in intensive agriculture sites.	Lacertid lizards	– Herbicide concentrations in soil (sum of 7 a.i.'s: mesotrione, glyphosate, bentazone, dicamba, dimethenamid-p, alachlor and terbuthylazine) ranged from 10-100 ug/kg soil in intensively cultivated sites; and from 0 -10 ug/kg soil in organic sites – No significant differences were observed between intensive and organic sites for population size and other demographic parameters – Lizards from intensively cultivated sites had a significantly lower body condition index and more blood parasites than animals from organic sites – No significant effects were observed for standard metabolic rate, locomotor performance, glutathione oxidative pathways and related enzyme activity, lipid peroxidation and liver and testis histology – Further results on thyroid and testes histology strongly suggest that exposure of lizards to the mixture of herbicides had thyroid disrupting effects that ultimately affected the male reproductive system	Amaral *et al.* 2012a, b Bicho *et al.* 2013

Annex 4.3-6 Recent large-scale studies and partial reviews of the effects of pesticide use on biodiversity

Included are studies, published between 2009 and 2020, that link pesticide use to biodiversity parameters such as number of species, taxonomic richness, etc.

Country	Study type	Study coverage	Elements of biodiversity studied	Effects observed	Reference
Global					
Global	Systematic review	73 peer-reviewed studies of entomofauna declines from various parts of the world, published since 1984 Insect declines are described in terms of population declines, biomass reductions as well as biodiversity losses	Insects	– Almost half of insect species are rapidly declining and a third are being threatened with extinction – The main drivers of species declines appear to be in order of importance: i) habitat loss and conversion to intensive agriculture and urbanisation (50% of reports); ii) pollution, mainly that by synthetic pesticides and fertilisers (26% of reports)	Sánchez-Bayo & Wyckhuys 2019
Global	Meta-analysis	20 studies, 49 observations comparing organic with conventional farming	Soil organisms	– Differences in soil biodiversity between organic and conventional farming practices were not detected	De Graaff *et al.* 2019
Global	Meta-analysis	94 studies, 184 observations comparing organic with conventional farming	arthropods, birds, microbes & plants	– Species richness on organic farms was on average 34% higher than on conventional farms – Plants benefited the most from organic farming; but arthropods, birds & microbes also showed a substantial positive effect – Current studies are heavily biased towards northern and western Europe and North America, while other regions with large areas of organic farming remain poorly investigated	Tuck *et al.*2015
Global	Meta-analysis	39 studies on 23 crops in 14 countries on 6 continents	Wild bees	– Organic farm management (lacking or having highly reduced use of pesticides and fertilizers) resulted in a 50% higher wild bee richness when compared to conventional farming	Kennedy *et al.* 2013
Regional and national					
Japan	Moderate-scale monitoring	21 irrigation ponds	Wide range of freshwater organisms	– A fungicide and an insecticide, alone or in combination with other stressors (invasive alien fish, concrete bank protection) had a significantly negative correlation the taxonomic richness of freshwater animals sampled in the study ponds	Ito *et al.* (2020)

Country	Study type	Study coverage	Elements of biodiversity studied	Effects observed	Reference
USA	Moderate-scale monitoring	4 sites in California, monitored between 1972 and 2012 Annual use of 5 neonicotinoid insecticides, as well as of 4 of the most widely used non-neonicotinoid insecticide classes	67 species of butterflies	– A negative association was detected butterfly diversity (Shannon index) and increasing neonicotinoid applications, while controlling for land use and other factors	Forister *et al.* 2016
USA	Moderate-scale monitoring	4 no-input, 6 organic and 15 conventional apple orchards Pesticide use was based on bee toxicity and volumes applied ("pesticide toxicity scores")	Wild bees	– Pesticide use did not adversely affect total bee species richness – Pesticide use was negatively associated with lower richness of sweat bees (*Lasioglossum* spp.); but with higher richness of mining bees (*Andrena* spp.)	Mallinger *et al* 2015
USA	moderate-scale monitoring	19 apple orchards with various pesticide use intensities	Wild bees	– Species richness of solitary wild bees decreased significantly with increasing pesticide use intensity; but richness of social wild bees did not – Fungicide sprays before and insecticide use after bloom had the strongest relationships with declines in the wild bee fauna. period. Unlike honeybees that reside in orchards during bloom only, wild bees have a greater exposure risk to pesticides as they actively forage in and around orchards weeks before and after apple bloom	Park *et al.* 2015
France	Large-scale monitoring	66 cereal fields located in three French departments	Farmland and non-farmland birds	– Herbicide dose was associated with negative effects on the Community Specialization Index. Overall, the proportion of habitat specialists, particularly of herbivorous species, decreased, and the proportion of generalists increased as herbicide doses increased. However, herbicide dose was positively correlated with bird richness and total abundance – No influence of insecticide or fungicide doses could be detected	Chiron *et al.* 2014
Germany, France, Australia (southern Victoria)	Large-scale monitoring	72 sampling sites Parameters assessed (among others): pesticide residues, macroinvertebrate abundance and taxa	Stream invertebrates	– Pesticide residues were associated with effects on both the species and family richness in both regions (statistically significant) – Decrease in taxonomic richness between the uncontaminated and highly contaminated categories ranged from 42% for the European species-level data to 27% for the Australian family-level data – Effects in Europe were detected at concentrations below environmental standards	Beketov *et al.* 2013

Country	Study type	Study coverage	Elements of biodiversity studied	Effects observed	Reference
Estonia, France, Germany, Ireland Netherlands, Poland, Spain, Sweden	Large-scale monitoring	270 cereals farms. 13 components of agricultural intensification assessed, among which: Number of applications and total amount of a.i. applied of insecticides, herbicides, fungicides	Wild plants, carabid beetles, breeding birds	– Number of wild plant species declined as the frequency of herbicide and insecticide application and the amounts of active ingredients of fungicides increased – Number of carabid species was negatively affected by the amounts of active ingredients of insecticide applied – Bird species diversity declined with increasing frequency of fungicide application (note: frequencies of fungicide and insecticide applications were closely correlated)	Geiger *et al.* 2010 Emmerson *et al.* 2016
Italy	Large-scale monitoring	Vine fields in northern Italy with or without applications of the insecticide fenitrothion	Wild bees, bumblebees, butterflies	– At field scale: Wild bee species richness declined after 2 or 3 insecticide applications, but not bumblebee or butterfly richness. – At the landscape level: richness of wild bees, bumblebees and butterflies was lower in areas with higher pesticide loads	Brittain *et al.* 2010
Canada	Retrospective review	Comparison of areas where imperilled species occur versus areas where they have been lost Pesticides variables were area treated with herbicides or with insecticides, or total area treated with pesticides (herbicide + insecticides + fungicides)	62 species of mammals, birds, amphibians, reptiles	– High losses of imperilled species were observed in regions with high proportions of agricultural land cover. However, losses of imperilled species are significantly more strongly related to the proportion of the region treated with agricultural pesticides. Species losses were more strongly related to herbicide use than to use of other pesticides	Gibbs *et al.* 2009

References

Agostini MG, Roesler I, Bonetto C, Ronco AE & Bilenca D (2020). Pesticides in the real world: The consequences of GMO-based intensive agriculture on native amphibians. *Biological Conservation* 241: 108355. https://doi.org/10.1016/j.biocon.2019.108355

Albuquerque AF, Ribeiro JS, Kummrow F, Nogueira AJA, Montagner CC & Umbuzeiro GA (2016) Pesticides in Brazilian freshwaters: A critical review. *Environmental Science: Processes and Impacts* 18(7): 779–787. https://doi.org/10.1039/c6em00268d

Ali U, Syed JH, Malik RN, Katsoyiannis A, Li J, Zhang G & Jones KC (2014) Organochlorine pesticides (OCPs) in South Asian region: A review. *Science of the Total Environment* 476–477: 705–717. https://doi.org/10.1016/j.scitotenv.2013.12.107

Alquwaizany AS, Alfadul SM, Khan MA & Alabdulaaly AI (2019). Occurrence of organic compounds in groundwater of Saudi Arabia. *Environmental Monitoring and Assessment*, 191(10). https://doi.org/10.1007/s10661-019-7723-6

Amaral MJ, Carretero MA, Bicho RC, Soares AMVM, Mann RM (2012a) The use of a lacertid lizard as a model for reptile ecotoxicology studies: Part 1 – Field demographics and morphology. *Chemosphere* 87: 757–764. doi:10.1016/j.chemosphere.2011.12.075

Amaral MJ, Bicho RC, Carretero MA, Sanchez-Hernandez JC, Faustino AMR, Soares AMVM, Mann RM (2012b) The use of a lacertid lizard as a model for reptile ecotoxicology studies: Part 2 – Biomarkers of exposure and toxicity among pesticide exposed lizards. *Chemosphere* 87: 765–774. doi:10.1016/j.chemosphere.2012.01.048

Ansara-Ross TM, Wepener V, van den Brink PJ & Ross MJ (2012) Pesticides in South African fresh waters. *African Journal of Aquatic Science* 37(1): 1–16. https://doi.org/10.2989/16085914.2012.666336

Baker NJ, Bancroft BA & Garcia TS (2013) A meta-analysis of the effects of pesticides and fertilizers on survival and growth of amphibians. *Science of the Total Environment* 449: 150–156 http://dx.doi.org/10.1016/j.scitotenv.2013.01.056

Battaglin WA, Smalling KL, Anderson C, Calhoun D, Chestnut T & Muths E (2016) Potential interactions among disease, pesticides, water quality and adjacent land cover in amphibian habitats in the United States. *Science of the Total Environment* 566–567: 320–332 http://dx.doi.org/10.1016/j.scitotenv.2016.05.062

Beketov MA, Kefford BJ, Schäfer RB & Liess M (2013) Pesticides reduce regional biodiversity of stream invertebrates. *PNAS* 110(27): 11039–11043. doi:10.1073/pnas.1305618110

Bhandari G, Atreya K, Scheepers PTJ & Geissen V (2020) Concentration and distribution of pesticide residues in soil: Non-dietary human health risk assessment. *Chemosphere* 126594. https://doi.org/10.1016/j.chemosphere.2020.126594

Bicho RC, Amaral MJ, Faustino AMR, Power DM, Rêma A, Carretero MA, Soares AMVM, Mann RM (2013) Thyroid disruption in the lizard *Podarcis bocagei* exposed to a mixture of herbicides: a field study. *Ecotoxicology* 22: 156–165. DOI 10.1007/s10646-012-1012-2

Bouvier J-C, Ricc B, Agerberg J & Lavigne C (2011) Apple orchard pest control strategies affect bird communities in southeastern France. *Environmental Toxicology and Chemistry* 30(1): 212–219. DOI: 10.1002/etc.377

Brain RA & Anderson JC (2019) The agro-enabled urban revolution, pesticides, politics, and popular culture: a case study of land use, birds, and insecticides in the USA. *Environmental Science and Pollution Research* 26: 21717-21735. https://doi.org/10.1007/s11356-019-05305-9

Brittain CA, Vighi M, Bommarco R, Settele J & Potts SG (2010). Impacts of a pesticide on pollinator species richness at different spatial scales. *Basic and Applied Ecology* 11(2): 106–115. https://doi.org/10.1016/j.baae.2009.11.007

Bruce-Vanderpuije P, Megson D, Reiner EJ, Bradley L, Adu-Kumi S & Gardella JA (2019) The state of POPs in Ghana – A review on persistent organic pollutants: Environmental and human exposure. *Environmental Pollution* 245: 331–342. https://doi.org/10.1016/j.envpol.2018.10.107

Calvert AM, Bishop CA, Elliot RD, Krebs EA, Kydd TM, Machtans CS & Robertson GJ (2013) A synthesis of human-related avian mortality in Canada. *Avian Conservation and Ecology* 8(2): 11. http://dx.doi.org/10.5751/ACE-00581-080211

Camenzuli L, Scheringer M & Hungerbühler K (2016) Local organochlorine pesticide concentrations in soil put into a global perspective. *Environmental Pollution* 217 : 11–18. https://doi.org/10.1016/j.envpol.2015.08.028

Cattan P, Charlier JB, Clostre F, Letourmy P, Arnaud L, Gresser J & Jannoyer M (2019) A conceptual model of organochlorine fate from a combined analysis of spatial and mid-to long-term trends of surface and ground water contamination in tropical areas (FWI). *Hydrology and Earth System Sciences*, 23(2), 691–709. https://doi.org/10.5194/hess-23-691-2019

Chen Y, Zang L, Liu M, Zhang C, Shen G, Du W, Sun Z, Fei J, Yang L, Wang Y, Wang X & Zhao M (2019) Ecological risk assessment of the increasing use of the neonicotinoid insecticides along the east coast of China. *Environment International* 127: 550–557. https://doi.org/10.1016/j.envint.2019.04.010

Chiron F, Chargé R, Julliard R, Jiguet F & Muratet A (2014) Pesticide doses, landscape structure and their relative effects on farmland birds. *Agriculture, Ecosystems and Environment* 185: 153– 160 http://dx.doi.org/10.1016/j.agee.2013.12.013

Close ME & Skinner A (2012) Sixth national survey of pesticides in groundwater in New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 46(4): 443–457. https://doi.org/10.1080/00288330.2012.707131

Coates PS, Brussee BE, Howe KB, Fleskes JP, Dwight IA, Connelly DP, Meshriy MG, Gardner SC (2017) Long-term and widespread changes in agricultural practices influence ring-necked pheasant abundance in California. *Ecology and Evolution* 7: 2546–2559. DOI: 10.1002/ece3.2675

Coscollà C &Yusà V (2016) Pesticides and agricultural air quality. Chapter 17 ***In:*** *The Quality of Air. M de la Guardia & S Armenta (eds).* Comprehensive Analytical Chemistry, Volume 73, Elsevier, Amsterdam.

Dahshan H, Megahed AM, Abd-Elall AMM, Abd-El-Kader MAG, Nabawy E & Elbana MH (2016) Monitoring of pesticides water pollution – The Egyptian River Nile. *Journal of Environmental Health Science and Engineering* 14(1): 1–9. https://doi.org/10.1186/s40201-016-0259-6

De Graaff MA, Hornslein N, Throop HL, Kardol P & van Diepen LTA (2019) Effects of agricultural intensification on soil biodiversity and implications for ecosystem functioning: A meta-analysis. Chapter 1 (pp 1-44) *In:* Advances in Agronomy, Vol. 155 . Elsevier Inc. https://doi.org/10.1016/bs.agron.2019.01.001

De Souza RM, Seibert D, Quesada HB, de Jesus Bassetti F, Fagundes-Klen MR & Bergamasco R (2020) Occurrence, impacts and general aspects of pesticides in surface water: A review. *Process Safety and Environmental Protection* 135: 22–37. https://doi.org/10.1016/j.psep.2019.12.035

Derbalah A, Chidya R, Jadoon W & Sakugawa H (2019) Temporal trends in organophosphorus pesticides use and concentrations in river water in Japan, and risk assessment. *Journal of Environmental Sciences (China)* 79: 135–152. https://doi.org/10.1016/j.jes.2018.11.019

Di Lorenzo T, Cifoni M, Fiasca B, Di Cioccio A & Galassi DMP (2018) Ecological risk assessment of pesticide mixtures in the alluvial aquifers of central Italy: Toward more realistic scenarios for risk mitigation. *Science of the Total Environment* 644: 161–172. https://doi.org/10.1016/j.scitotenv.2018.06.345

Dirbaba NB, Li S, Wu H, Yan X & Wang J (2018) Organochlorine pesticides, polybrominated diphenyl ethers and polychlorinated biphenyls in surficial sediments of the Awash River Basin Ethiopia. *PLoS ONE* 13(10). https://doi.org/10.1371/journal.pone.0205026

EFSA (2018) Scientific Opinion on the state of the science on pesticide risk assessment for amphibians and reptiles. EFSA PPR Panel (EFSA Panel on Plant Protection Products and their Residues): Ockleford C, Adriaanse P, Berny P, Brock T, Duquesne S, Grilli S, Hernandez-Jerez AF, Bennekou SH, Klein M, Kuhl T, Laskowski R, Machera K, Pelkonen O, Pieper S, Stemmer M, Sundh I, Teodorovic I, Tiktak A, Topping CJ, Wolterink G, Aldrich A, Berg C, Ortiz-Santaliestra M, Weir S, Streissl F and Smith RH. *EFSA Journal* 16(2): 5125. https://doi.org/10.2903/j.efsa.2018.5125

Egea-Serrano A, Relyea RA, Tejedo M & Torralva M (2012) Understanding of the impact of chemicals on amphibians: a meta-analytic review. *Ecology and Evolution* 2(7): 1382–1397. doi: 10.1002/ece3.249

Elibariki R & Maguta MM (2017) Status of pesticides pollution in Tanzania – A review. *Chemosphere* 178: 154–164. https://doi.org/10.1016/j.chemosphere.2017.03.036

Emmerson M, Morales MB, Oñate JJ, Batáry P, Berendse F, Liira J, Aavik T, Guerrero I, Bommarco R, Eggers S, Pärt T, Tscharntke T, Weisser W, Clement L & Bengtsson J (2016) How agricultural intensification affects biodiversity and ecosystem services. Chapter 2 In: *Advances in Ecological Research* 55: 43-97 http://dx.doi.org/10.1016/bs.aecr.2016.08.005

Eqani AS, Malik RN, Alamdar A & Faheem H (2012) Status of organochlorine contaminants in the different environmental compartments of Pakistan: A review on occurrence and levels. *Bulletin of Environmental Contamination and Toxicology* 88(3): 303–310. https://doi.org/10.1007/s00128-011-0496-4

Ertl HMH, Mora MA, Brightsmith DJ, Navarro-Alberto JA (2018) Potential impact of neonicotinoid use on Northern bobwhite (*Colinus virginianus*) in Texas: A historical analysis. *PLoS ONE* 13(1): e0191100. https://doi.org/10.1371/journal.pone.0191100

Forister ML, Cousens B, Harrison JG, Anderson K, Thorne JH, Waetjen D. Nice CC, De Parsia M, Hladik ML, Meese R, Van Vliet H & Shapiro AM (2016) Increasing neonicotinoid use and the declining butterfly fauna of lowland California. *Biology Letters* 12(8). https://doi.org/10.1098/rsbl.2016.0475

Fosu-Mensah BY, Okoffo ED, Darko G & Gordon C (2016) Assessment of organochlorine pesticide residues in soils and drinking water sources from cocoa farms in Ghana. *SpringerPlus* 5 (1). https://doi.org/10.1186/s40064-016-2352-9

Freitas LM, Paranaíiba JFFS, Peréz APS, Machado MRF & Lima FC (2020) Toxicity of pesticides in lizards. *Human and Experimental Toxicology* 39(5): 596-604. [OI: 10.1177/0960327119899980

Geiger F, Bengtsson J, Berendse F, Weisser WW, Emmerson M, Morales MB, Ceryngier P, Liira J, Tscharntke T, Winqvist C, Eggers S, Bommarco R, Pärt T, Bretagnolle V, Plantegenest M, Clement LW, Dennis C, Palmer C, Oñate JJ, Guerrero I, Hawro V, Aavik T, Thies C, Flohre A, Hänke S, Fischer C, Goedhart PW & Inchausti P (2010) Persistent negative effects of pesticides on biodiversity and biological control potential on European farmland. *Basic and Applied Ecology* 11: 97–105. https://doi.org/10.1016/j.baae.2009.12.001

Gibbons D, Morrissey C & Mineau P (2015) A review of the direct and indirect effects of neonicotinoids and fipronil on vertebrate wildlife. *Environmental Science and Pollution Research International* 22(1): 103-118. DOI:10.1007/s11356-014-3180-5

Gibbs KE, MacKey RL & Currie DJ (2009) Human land use, agriculture, pesticides and losses of imperiled species. *Diversity and Distributions* 15(2): 242–253. https://doi.org/10.1111/j.1472-4642.2008.00543.x

Grung M, Lin Y, Zhang H, Steen AO, Huang J, Zhang G & Larssen T (2015) Pesticide levels and environmental risk in aquatic environments in China – A review. *Environment International* 81: 87-97. https://doi.org/10.1016/j.envint.2015.04.013

Guida Y de S, Meire RO, Torres JPM & Malm O (2018) Air contamination by legacy and current-use pesticides in Brazilian mountains: An overview of national regulations by monitoring pollutant presence in pristine areas. *Environmental Pollution* 242: 19-30. https://doi.org/10.1016/j.envpol.2018.06.061

Hallmann CA, Foppen RPB, van Turnhout CAM, de Kroon H & Jongejans E (2014) Declines in insectivorous birds are associated with high neonicotinoid concentrations. *Nature* 511: 341. doi:10.1038/nature13531

Hao Y, Li Y, Ha, X, Wang T, Yang R, Wang P, Xiao K, Li W, Lu H, Fu J, Wang Y, Shi J, Zhang Q & Jiang G (2019) Air monitoring of polychlorinated biphenyls, polybrominated diphenyl ethers and organochlorine pesticides in West Antarctica during 2011–2017: Concentrations, temporal trends and potential sources. *Environmental Pollution* 249: 381–389. https://doi.org/10.1016/j.envpol.2019.03.039

Hasan MK, Shahriar A & Jim KU (2019) Water pollution in Bangladesh and its impact on public health. *Heliyon* 5(8): e02145. https://doi.org/10.1016/j.heliyon.2019.e02145

Hill JM, Egan JF, Stauffer GE, Diefenbach DR (2014) Habitat availability is a more plausible explanation than insecticide acute toxicity for U.S. grassland bird species declines. *PLoS ONE* 9(5): e98064. doi:10.1371/journal.pone.0098064

Hladik ML & Kolpin DW (2016) First national-scale reconnaissance of neonicotinoid insecticides in streams across the USA. *Environmental Chemistry* 13 (1): 12–20. https://doi.org/10.1071/EN15061

Humann-Guilleminot S, Binkowski ŁJ, Jenni L, Hilke G, Glauser G & Helfenstein F (2019) A nation-wide survey of neonicotinoid insecticides in agricultural land with implications for agri-environment schemes. *Journal of Applied Ecology* 00: 1–13. https://doi.org/10.1111/1365-2664.13392

Hvězdová M, Kosubová P, Košíková M, Scherr KE, Šimek Z, Brodský L, Šudoma M, Škulcová L, Sáňka M, Svobodová M, Krkošková L, Vašíčková J, Neuwirthová N, Bielská L & Hofman J (2018) Currently and recently used pesticides in Central European arable soils. *Science of the Total Environment* 613-614: 361-370. https://doi.org/10.1016/j.scitotenv.2017.09.049

Isogai N, Hogarh JN, Seike N, Kobara Y, Oyediran F, Wirmvem MJ, Ayonghe SN, Fobil J & Masunaga S (2018) Atmospheric monitoring of organochlorine pesticides across some West African countries. *Environmental Science and Pollution Research* 25(32): 31828-31835. https://doi.org/10.1007/s11356-016-7284-y

Ito HC, Shiraishi H, Nakagawa M & Takamura N (2020) Combined impact of pesticides and other environmental stressors on animal diversity in irrigation ponds. *PLoS One* 15. doi:10.1371/journal.pone.0229052

Jahn T, Hötker H, Oppermann R, Bleil R, Vele L (2014) Protection of biodiversity of free living birds and mammals in respect of the effects of pesticides. Report No. (UBA-FB) 001830. Federal Environment Agency, Dessau-Roßlau, Germany. https://www.umweltbundesamt.de/en/publikationen/protection-of-biodiversity-of-free-living-birds

Jurado A, Vàzquez-Suñé E, Carrera J, López de Alda M, Pujades E & Barceló D (2012) Emerging organic contaminants in groundwater in Spain: A review of sources, recent occurrence and fate in a European context. *Science of the Total Environment* 440: 82–94. https://doi.org/10.1016/j.scitotenv.2012.08.029

Jurado A, Walther M & Díaz-Cruz MS (2019) Occurrence, fate and environmental risk assessment of the organic microcontaminants included in the Watch Lists set by EU Decisions 2015/495 and 2018/840 in the groundwater of Spain. *Science of the Total Environment* 663: 285–296. https://doi.org/10.1016/j.scitotenv.2019.01.270

K'oreje KO, Okoth M, Van Langenhove H & Demeestere K (2020) Occurrence and treatment of contaminants of emerging concern in the African aquatic environment: Literature review and a look ahead. *Journal of Environmental Management* 254. https://doi.org/10.1016/j.jenvman.2019.109752

Karki AJ, Cappelli P, Dirks C, Pekar H, Hellenäs KE, Rosén J & Westerberg E (2020) New efficient methodology for screening of selected organic micropollutants in raw- and drinking water from 90 Swedish water treatment plants. *Science of the Total Environment* 724. https://doi.org/10.1016/j.scitotenv.2020.138069

Kennedy CM, Lonsdorf E, Neel MC, Williams NM, Ricketts TH, Winfree R, Bommarco R, Brittain C, Burley AL, Cariveau D, Carvalheiro LG, Chacoff NP, Cunningham SA, Danforth BN, Dudenhöffer JH, Elle E, Gaines HR, Garibaldi LA, Gratton C, Holzschuh A, Isaacs R, Javorek SK, Jha S, Klein AM, Krewenka K, Mandelik Y, Mayfield MM, Morandin L, Neame LA, Otieno M, Park M, Potts SG, Rundlöf M, Saez A, Steffan-Dewenter I, Taki H, Felipe Viana B, Westphal C, Wilson JK, Greenleaf SS & Kremen C (2013) A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. *Ecology Letters* 16(5): 584–599. https://doi.org/10.1111/ele.12082

Kirchner M, Jakobi G, Körner W, Levy W, Moche W, Niedermoser B, Schaub M, Ries L, Weiss P, Antritter F, Fischer N, Henkelmann B & Schramm KW (2016) Ambient air levels of organochlorine pesticides at three high alpine monitoring stations: Trends and dependencies on geographical origin. *Aerosol and Air Quality Research* 16(3): 738–751. https://doi.org/10.4209/aaqr.2015.04.0213

Knauer K (2016) Pesticides in surface waters: a comparison with regulatory acceptable concentrations (RACs) determined in the authorization process and consideration for regulation. *Environmental Sciences Europe* 28 (1): 0–6. https://doi.org/10.1186/s12302-016-0083-8

Köck-Schulmeyer M, Ginebreda A, Postigo C, Garrido T, Fraile J, López de Alda M & Barceló D (2014) Four-year advanced monitoring program of polar pesticides in groundwater of Catalonia (NE-Spain). *Science of the Total Environment* 470–471: 1087–1098. https://doi.org/10.1016/j.scitotenv.2013.10.079

Komárek M, Čadková E, Chrastný V, Bordas F & Bollinger JC (2010) Contamination of vineyard soils with fungicides: A review of environmental and toxicological aspects. *Environment International* 36(1): 138–151. https://doi.org/10.1016/j.envint.2009.10.005

Kouzayha A, Al Ashi A, Al Akoum R, Al Iskandarani M, Budzinski H & Jaber F (2013) Occurrence of pesticide residues in Lebanon's water resources. *Bulletin of Environmental Contamination and Toxicology* 91(5): 503–509. https://doi.org/10.1007/s00128-013-1071-y

Leusch FDL, Neale PA, Arnal C, Aneck-Hahn NH, Balaguer P, Bruchet A, Escher BI, Esperanza M, Grimaldi M, Leroy G, Scheurer M, Schlichting R, Schriks M & Hebert A (2018) Analysis of endocrine activity in drinking water, surface water and treated wastewater from six countries. *Water Research* 139: 10–18. https://doi.org/10.1016/j.watres.2018.03.056

Levillain J, Cattan P, Colin F, Voltz M & Cabidoche Y-M (2012) Analysis of environmental and farming factors of soil contamination by a persistent organic pollutant, chlordecone, in a banana production area of French West Indies. *Agriculture, Ecosystems & Environment* 159: 123-132. https://doi.org/10.1016/j.agee.2012.07.005

Li J, Li F & Liu Q (2015) Sources, concentrations and risk factors of organochlorine pesticides in soil, water and sediment in the Yellow River estuary. *Marine Pollution Bulletin* 100(1): 516–522. https://doi.org/10.1016/j.marpolbul.2015.09.003

Li Y, Miao R & Khanna M (2020) Neonicotinoids and decline in bird biodiversity in the United States. *Nature Sustainability*. https://www.nature.com/articles/s41893-020-0582-x

Lopez B, Ollivier P, Togola A, Baran N & Ghestem JP (2015) Screening of French groundwater for regulated and emerging contaminants. *Science of the Total Environment* 518–519: 562–573. https://doi.org/10.1016/j.scitotenv.2015.01.110

Łozowicka B, Kaczyński P, Wolejko E, Piekutin J, Sagitov A, Toleubayev K, Isenova G & Abzeitova E (2016) Evaluation of organochlorine pesticide residues in soil and plants from east Europe and central Asia. *Desalination and Water Treatment* 57(3): 1310–1321. https://doi.org/10.1080/19443994.2014.996008

Malaj E, von der Ohe PC, Grote M, Kühne R, Mondy CP, Usseglio-Polatera P, Brack W & Schäfer RB (2014) Organic chemicals jeopardize the health of freshwater ecosystems on the continental scale. *Proceedings of the National Academy of Sciences* 111 (26): 9549–9554. https://doi.org/10.1073/pnas.1321082111

Mallinger RE, Werts P & Gratton C (2015) Pesticide use within a pollinator-dependent crop has negative effects on the abundance and species richness of sweat bees, *Lasioglossum* spp., and on bumble bee colony growth. *Journal of Insect Conservation* 19(5): 999–1010. https://doi.org/10.1007/s10841-015-9816-z

Malyan SK, Singh R, Rawat M, Kumar M, Pugazhendhi A, Kumar A, Kumar V & Kumar SS (2019) An overview of carcinogenic pollutants in groundwater of India. *Biocatalysis and Agricultural Biotechnology* 21. https://doi.org/10.1016/j.bcab.2019.101288

Manamsa K, Crane E, Stuart M, Talbot J, Lapworth D & Hart A (2016) A national-scale assessment of micro-organic contaminants in groundwater of England and Wales. *Science of the Total Environment* 568: 712–726. https://doi.org/10.1016/j.scitotenv.2016.03.017

McManus SL, Richards KG, Grant J, Mannix A & Coxon CE (2014) Pesticide occurrence in groundwater and the physical characteristics in association with these detections in Ireland. *Environmental Monitoring and Assessment* 186(11): 7819-7836. https://doi.org/10.1007/s10661-014-3970-8

McManus SL, Coxon CE, Mellander PE, Danaher M & Richards KG (2017) Hydrogeological characteristics influencing the occurrence of pesticides and pesticide metabolites in groundwater across the Republic of Ireland. *Science of the Total Environment* 601–602: 594–602. https://doi.org/10.1016/j.scitotenv.2017.05.082

Meffe R & de Bustamante I (2014). Emerging organic contaminants in surface water and groundwater: A first overview of the situation in Italy. *Science of the Total Environment* 481 (1): 280–295. https://doi.org/10.1016/j.scitotenv.2014.02.053

Menzies R, Soares Quinete N, Gardinali P & Seba D (2013) Baseline occurrence of organochlorine pesticides and other xenobiotics in the marine environment: Caribbean and Pacific collections. *Marine Pollution Bulletin* 70 (1–2): 289–295. https://doi.org/10.1016/j.marpolbul.2013.03.003

Mineau P (2013) Avian mortality from pesticides used in agriculture in Canada. *Unpublished manuscript*, provided as Annex to Calvert *et al.* 2013.

Mineau P & Whiteside M (2013) Pesticide acute toxicity Is a better correlate of U.S. grassland bird declines than agricultural intensification. *PLoS ONE* 8(2): e57457. doi:10.1371/journal.pone.0057457

Mingo V, Lötters S & Wagner N (2017) The impact of land use intensity and associated pesticide applications on fitness and enzymatic activity in reptiles—A field study. *Science of the Total Environment* 590–591: 114–124 http://dx.doi.org/10.1016/j.scitotenv.2017.02.178

Mingo V, Lötters S & Wagner N (2016) Risk of pesticide exposure for reptile species in the European Union. *Environmental Pollution* 215: 164e169 http://dx.doi.org/10.1016/j.envpol.2016.05.011

Mishra K, Sharma RC & Kumar S (2012) Contamination levels and spatial distribution of organochlorine pesticides in soils from India. *Ecotoxicology and Environmental Safety* 76(1): 215–225. https://doi.org/10.1016/j.ecoenv.2011.09.014

Mochungong P & Zhu J (2015) DDTs, PCBs and PBDEs contamination in Africa, Latin America and South-southeast Asia – a review. *AIMS Environmental Science* 2(2): 374–399. https://doi.org/10.3934/environsci.2015.2.374

Mohaupt V, Völker J, Altenburger R, Birk S, Kirst I, Kühnel D, Küster E, Semerádová S, Šubelj G & Whalley C (2020) Pesticides in European rivers, lakes and groundwaters – Data assessment. ETC/ICM Technical Report 1/2020. https://www.eionet.europa.eu/etcs/etc-icm/products/etc-icm-report-1-2020-pesticides-in-european-rivers-lakes-and-groundwaters-data-assessment

Morrissey CA, Mineau P, Devries JH, Sanchez-Bayo F, Liess M, Cavallaro MC & Liber K (2015) Neonicotinoid contamination of global surface waters and associated risk to aquatic invertebrates: A review. *Environment International* 74: 291–303. https://doi.org/10.1016/j.envint.2014.10.024

Nesser GAA, Abdelbagi AO, Hammad AMA, Tagelseed M & Laing MD (2016) Levels of pesticides residues in the White Nile water in the Sudan. *Environmental Monitoring and Assessment* 188 (6). https://doi.org/10.1007/s10661-016-5367-3

North AC, Hodgson DJ, Price SJ & Griffiths AGF (2015) Anthropogenic and ecological drivers of amphibian disease (Ranavirosis). *PLoS ONE* 10(6): e0127037. doi:10.1371/journal.pone.0127037

Odendaal C, Seaman MT, Kemp G, Patterton HE & Patterton HG (2015) An LC-MS/MS based survey of contaminants of emerging concern in drinking water in South Africa. *South African Journal of Science* 111(9–10). https://doi.org/10.17159/sajs.2015/20140401

Orton F & Tyler CR (2015) Do hormone-modulating chemicals impact on reproduction and development of wild amphibians? *Biological Reviews* 90: 1100–1117. doi: 10.1111/brv.12147

Pan L, Sun J, Li Z, Zhan Y, Xu S & Zhu L (2018) Organophosphate pesticide in agricultural soils from the Yangtze River Delta of China: concentration, distribution, and risk assessment. *Environmental Science and Pollution Research* 25(1): 4–11. https://doi.org/10.1007/s11356-016-7664-3

Park MG, Blitzer EJ, Gibbs J, Losey JE & Danforth BN (2015) Negative effects of pesticides on wild bee communities can be buffered by landscape context. *Proceedings of the Royal Society B* 282: 20150299. http://dx.doi.org/10.1098/rspb.2015.0299

Parsons KC, Mineau P & Renfrew RB (2010) Effects of pesticide use in rice fields on birds. *Waterbirds* 33 (Special Publication 1): 193-218. https://doi.org/10.1675/063.033.s115

PBL (2019) Geïntegreerde gewasbescherming nader beschouwd – Tussenevaluatie van de nota Gezonde Groei, Duurzame Oogst (Integrated pest management closer considered – Interim evaluation of the policy for healthy growth and sustainable yield). Netherlands Environmental Assessment Agency (PBL), The Hague, The Netherlands. (in Dutch). https://www.rivm.nl/publicaties/tussenevaluatie-van-nota-gezonde-groei-duurzame-oogst-deelproject-milieu

Pelosi C, Bertrand C, Daniele G, Coeurdassier M, Benoit P, Nélieu S, Lafay F, Bretagnolle V, Gaba S, Vulliet E & Fritsch C (2021) Residues of currently used pesticides in soils and earthworms: A silent threat? *Agriculture, Ecosystems and Environment* 305. https://doi.org/10.1016/j.agee.2020.107167

Pietrzak D, Kania J, Malina G, Kmiecik E & Wątor K (2019) Pesticides from the EU First and Second Watch Lists in the Water Environment. *Clean - Soil, Air, Water* 47(7). https://doi.org/10.1002/clen.201800376

Pirsaheb M, Hossini H, Asadi F & Janjani H (2017) A systematic review on organochlorine and organophosphorus pesticides content in water resources. *Toxin Reviews* 36(3): 210–221. https://doi.org/10.1080/15569543.2016.1269810

Postigo C & Barceló D (2015) Synthetic organic compounds and their transformation products in groundwater: Occurrence, fate and mitigation. *Science of the Total Environment* 503–504: 32–47. https://doi.org/10.1016/j.scitotenv.2014.06.019

Rauert C, Harner T, Schuster JK, Eng A, Fillmann G, Castillo LE, Fentanes O, Ibarra MV, Miglioranza KSB, Rivadeneira IM, Pozo K & Aristizábal Zuluaga BH (2018) Air monitoring of new and legacy POPs in the Group of Latin America and Caribbean (GRULAC) region. *Environmental Pollution* 243: 1252–1262. https://doi.org/10.1016/j.envpol.2018.09.048

Sackaria M & Elango L (2020). Organic micropollutants in groundwater of India—A review. *Water Environment Research* 92(4): 504–523. https://doi.org/10.1002/wer.1243

Sánchez-Bayo F and Wyckhuys KAG (2019) Worldwide decline of the entomofauna: A review of its drivers. *Biological Conservation* 232: 8–27. https://doi.org/10.1016/j.biocon.2019.01.020

Sánchez-Bayo F, Goka K & Hayasaka D (2016) Contamination of the Aquatic Environment with Neonicotinoids and its Implication for Ecosystems. *Frontiers in Environmental Science* 4. https://doi.org/10.3389/fenvs.2016.00071

Schreiner VC, Link M, Kunz S, Szöcs E, Scharmüller A, Vogler B, Beck B, Battes KP, Cimpean M, Singer HP, Hollender J & Schäfer RB (2021) Paradise lost? Pesticide pollution in a European region with considerable amount of traditional agriculture. *Water Research* 188. doi:10.1016/j.watres.2020.116528

Shunthirasingham V, Oyiliagu CE, Cao Z, Gouin T, Wania F, Lee S-C, Pozo K, Harner T & Muir DCG (2010) Spatial and temporal pattern of pesticides in the global atmosphere. *Journal of Environmental Monitoring* **12:** 1650–1657. https://doi.org/10.1039/C0EM00134A

Sievers M, Hale R, Parris KM, Melvin SD, Lanctôt CM & Swearer SE (2019) Contaminant-induced behavioural changes in amphibians: A meta-analysis. *Science of the Total Environment* 693: 133570. https://doi.org/10.1016/j.scitotenv.2019.07.376

Silva V, Mol HGJ, Zomer P, Tienstra M, Ritsema CJ & Geissen V (2019) Pesticide residues in European agricultural soils – A hidden reality unfolded. *Science of the Total Environment* 653: 1532–1545. https://doi.org/10.1016/j.scitotenv.2018.10.441

Sjerps RMA, Kooij PJF, van Loon A & Van Wezel AP (2019) Occurrence of pesticides in Dutch drinking water sources. *Chemosphere* 235: 510–518. https://doi.org/10.1016/j.chemosphere.2019.06.207

Spiller KJ & Dettmers R (2019) Evidence for multiple drivers of aerial insectivore declines in North America. *The Condor: Ornithological Applications* 121(2): 1-13. https://doi.org/10.1093/condor/duz010

Stanton RL, Morrissey CA & Clark (2018) Analysis of trends and agricultural drivers of farmland bird declines in North America: A review. *Agriculture, Ecosystems and Environment* 254: 244–254. https://doi.org/10.1016/j.agee.2017.11.028

Stehle S, Bub S & Schulz R (2018) Compilation and analysis of global surface water concentrations for individual insecticide compounds. *Science of the Total Environment* 639: 516–525. https://doi.org/10.1016/j.scitotenv.2018.05.158

Stehle S and Schulz R (2015) Agricultural insecticides threaten surface waters at the global scale. *Proceedings of the National Academy of Sciences* 112(18): 5750–5755. https://doi.org/10.1073/pnas.1500232112

Stone WW, Gilliom RJ & Ryberg KR (2014). Pesticides in U.S. streams and rivers: Occurrence and trends during 1992-2011. *Environmental Science and Technology* 48 (19): 11025–11030. https://doi.org/10.1021/es5025367

Sun J, Pan L, Tsang DCW, Zhan Y, Zhu L & Li X (2018) Organic contamination and remediation in the agricultural soils of China: A critical review. *Science of the Total Environment* 615: 724–740. https://doi.org/10.1016/j.scitotenv.2017.09.271

Sun Y, Cao M, Wan Y, Wang H, Liu J, Pan F, He W, Huang H & He Z (2020) Spatial variation of 2,4-D and MCPA in tap water and groundwater from China and their fate in source, treated, and tap water from Wuhan, Central China. *Science of the Total Environment* 727: 138691. https://doi.org/10.1016/j.scitotenv.2020.138691

Székács A, Mörtl M & Darvas B (2015) Monitoring pesticide residues in surface and ground water in Hungary: Surveys in 1990-2015. *Journal of Chemistry* 2015: Article ID 717948. https://doi.org/10.1155/2015/717948

Taiwo AM (2019) A review of environmental and health effects of organochlorine pesticide residues in Africa. *Chemosphere* 220: 1126–1140. https://doi.org/10.1016/j.chemosphere.2019.01.001

Tang W, Wang D, Wang J, Wu Z, Li L, Huang M, Xu S & Yan D (2018) Pyrethroid pesticide residues in the global environment: An overview. *Chemosphere* 191 (308): 990–1007. https://doi.org/10.1016/j.chemosphere.2017.10.115

Tariq MI, Afzal S, Hussain I & Sultana N (2007) Pesticides exposure in Pakistan: A review. *Environment International* 33(8): 1107–1122. https://doi.org/10.1016/j.envint.2007.07.012

Toccalino PL, Gilliom RJ, Lindsey BD & Rupert MG (2014) Pesticides in groundwater of the United States: decadal-scale changes, 1993-2011. *Ground Water* 52: 112–125. https://doi.org/10.1111/gwat.12176

Tuck SL, Winqvist C, Mota F, Ahnström J, Turnbull LA & Bengtsson J (2014) Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. *Journal of Applied Ecology* 51: 746–755. doi: 10.1111/1365-2664.12219

UNEP (2017) Second global monitoring report. Global monitoring plan for persistent organic pollutants. Stockholm Convention on Persistent Organic Pollutants. Conference of Parties, 8th meeting, Geneva, 24 April 5 May 2017. Document UNEP/POPS/COP.8/INF/38. United Nations Environment Programme (UNEP), Geneva. http://www.pops.int/TheConvention/ConferenceoftheParties/Meetings/COP8/tabid/5309/Default.aspx

Vorkamp K & Rigét FF (2014) A review of new and current-use contaminants in the Arctic environment: Evidence of long-range transport and indications of bioaccumulation. *Chemosphere* 111: 379–395. https://doi.org/10.1016/j.chemosphere.2014.04.019

Vulliet E, Tournier M, Vauchez A, Wiest L, Baudot R, Lafay F, Kiss A & Cren-Olivé C (2014) Survey regarding the occurrence of selected organic micropollutants in the groundwaters of overseas departments. *Environmental Science and Pollution Research* 21(12): 7512–7521. https://doi.org/10.1007/s11356-014-2619-z

Wagner N, Mingo V, Schulte U, Lötters S (2015) Risk evaluation of pesticide use to protected European reptile species. *Biological Conservation* 191: 667–673. http://dx.doi.org/10.1016/j.biocon.2015.08.002

Wang A, Hu X, Wan Y, Mahai G, Jiang Y, Huo W, Zhao X, Liang G, He Z, Xia W & Xu S (2020) A nationwide study of the occurrence and distribution of atrazine and its degradates in tap water and groundwater in China: Assessment of human exposure potential. *Chemosphere* 252. https://doi.org/10.1016/j.chemosphere.2020.126533

Wang X, Wang C, Zhu T, Gong P, Fu J & Cong Z (2019) Persistent organic pollutants in the polar regions and the Tibetan Plateau: A review of current knowledge and future prospects. *Environmental Pollution* 248: 191–208. https://doi.org/10.1016/j.envpol.2019.01.093

Weir SM, Yu S, Knox A, Talent LG, Monks JM & Salice CJ (2016) Acute toxicity and risk to lizards of rodenticides and herbicides commonly used in New Zealand. *New Zealand Journal of Ecology* 40(3): 342-350. DOI: 10.20417/nzjecol.40.43

Wolfram J, Stehle S, Bub S, Petschick LL & Schulz R (2018) Meta-Analysis of Insecticides in United States Surface Waters: Status and Future Implications. *Environmental Science and Technology* 52 (24): 14452–14460. https://doi.org/10.1021/acs.est.8b04651

Wood TJ & Goulson D (2017) The environmental risks of neonicotinoid pesticides: a review of the evidence post 2013. *Environmental Science and Pollution Research* 24(21): 17285–17325. https://doi.org/10.1007/s11356-017-9240-x

Yadav IC, Devi NL, Syed JH, Cheng Z, Li J, Zhang G & Jones KC (2015) Current status of persistent organic pesticides residues in air, water, and soil, and their possible effect on neighboring countries: A comprehensive review of India. *Science of the Total Environment* 511: 123–137. https://doi.org/10.1016/j.scitotenv.2014.12.041

Zhong G, Xie Z, Cai M, Möller A, Sturm R, Tang J, Zhang G, He J & Ebinghaus R (2012) Distribution and air-sea exchange of current-use pesticides (CUPs) from East Asia to the high Arctic Ocean. *Environmental Science and Technology* 46(1): 259–267. https://doi.org/10.1021/es202655k

Zubrod JP, Bundschuh M, Arts G, Brühl CA, Imfeld G, Knäbel A, Payraudeau S, Rasmussen JJ, Rohr J, Scharmüller A, Smalling K, Stehle S, Schulz R & Schäfer RB (2019). Fungicides: An overlooked pesticide class? *Environmental Science and Technology* 53: 3347–3365. https://doi.org/10.1021/acs.est.8b04392

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them: Envisioning a chemical-safe world

Annex to Chapter 4.4 – Adverse human health effects of pesticides

Annex 4.4-1 Associations between pesticide exposure and diseases or other health outcomes which were reviewed by Ntzani *et al.* 2020, but for which no meta-analysis was performed.

See Box 4.4-3 of the main report. No meta-analysis was performed, generally, if insufficient relevant studies were available or if exposures or exposure assessments were highly heterogeneous.

Health outcome	Pesticide or pesticide class	Ntzani *et al.* 2020
Effects on adults		
Non-Hodgkin lymphoma	Lindane	No meta-analysis (MA) performed
Hodgkin lymphoma	Various	No MA performed
Leukemia	Various	No MA performed
Multiple myeloma	Pesticides[1]	No MA performed
Prostate cancer	Chlordecone	No MA performed
Kidney cancer	Various	No MA performed
Stomach cancer	Pesticides	No MA performed
Liver cancer	Various	No MA performed
Brain cancer	Various	No MA performed
Colorectal cancer	Various	No MA performed
Thyroid cancer	Various	No MA performed
Pancreatic cancer	Various	No MA performed
Skin cancer	Various	No MA performed
Cancer of lips, oral cavity or pharynx	Various	No MA performed
Esophageal cancer	Various	No MA performed
Larynx cancer	Various	No MA performed
Ovarian cancer	Various	No MA performed
Parkinson's disease	Lindane	No MA performed
	Dieldrin	No MA performed
Male fertility	Dibromochloropropane (DBCP)	Non-assessed
Amyotrophic Lateral Sclerosis (ALS)	Pesticides	No MA performed
Thyroid diseases	Various	No MA performed
Kidney diseases	Various	No MA performed
Immune/ autoimmune diseases	Various	No MA performed
Asthma (childhood)	Various	No MA performed
Alzheimer's disease and dementia	Various	No MA performed
Autism-spectrum disorders	Various	No MA performed
Effects on child development		
Congenital malformations	Pesticides (exposure during pregnancy)	No MA performed
Weight gain	p,p'-DDE (exposure during pregnancy)	Non-assessed
Neurodevelopment	Organophosphates (exposure during pregnancy)	Non-assessed
Brain tumours	Pesticides (exposure during pregnancy)	No MA performed

[1] Pesticides means pesticides in general, without distinction of specific classes or active ingredients.

Reference

Ntzani EE, Markozannes G, Tagas C, Rizos E & Karalexi M (2020). Epidemiological evidence review report: Exposure to pesticides and fertilizers and human health – Part A. Department of Hygiene and Epidemiology, University of Ioannina School of Medicine, Ioannina, Greece. Supporting document to Environmental and health impacts of pesticides and fertilizers and ways of minimizing them. https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them

Envisioning A Chemical-Safe World

Chapter 5 of 12

The environmental, human health and economic impacts of pesticides

Contents

1 Global drivers, actors and policies affecting pesticides and fertilizer use

2 Status and trends of pesticide use

3 The regulatory and policy environment for pesticide management

4 Environmental and health effects of pesticide use

About ii

5 The environmental, human health and economic impacts of pesticides 1

5.1 Overview 1

5.2 Burden of disease 3

5.3 Monetary costs 4

5.3.1 The benefits of pesticide use 4
5.3.2 The costs of pesticide use 5
5.3.3 Cost-benefit assessments 9

5.4 Ecosystem services 11

5.4.1 Introduction 11
5.4.2 The impact of pesticides on ecosystem services or nature's contributions to people 13

References 17

6 Current pesticide risk reduction and risk management

7 Status and trends of fertilizer use

8 The regulatory and policy environment of fertilizer management and use

9 Environmental and health effects of fertilizer use

10 The impact of fertilizer use

11 Current fertilizer risk reduction and risk management

12 Transformative actions to minimize the adverse impacts of pesticide and fertilizers

List of Boxes

Box 5.4-1 IPBES reporting categories for nature's contributions to people (NCP) which are most susceptible to being impacted by the use of pesticides. 12

List of Figures

Figure 5.1-1 The environmental and health impact of pesticide use consists of any durable changes in the condition of people or their environment brought about by the (adverse) effect(s) of a pesticide. 1

Figure 5.4-1 Percentage dependence on animal-mediated pollination of leading global crops that are directly consumed by humans and traded on the global market. 13

Figure 5.4-2 Spider chart of pest management's impact on soil functions and ecosystem services. 14

List of Tables

Table 5.2-1 Human disease burden associated with certain pesticides in the EU and the United States in 2010. 4

Table 5.3-1 Direct and indirect costs of pesticide use, as they may be borne by the pesticide user or society as a whole. 6

Table 5.3-2 Estimates of overall indirect costs of pesticide use. 8

Table 5.3-3 Estimates of the economic cost of exposure to certain endocrine disrupting pesticides in the EU and the United States. Costs consisted of hospitalization, physicians' services, nursing home care, medical appliances and related items. 8

Table 5.2-4 Estimates of the net return of pesticide use, at farm level, without taking into account hidden costs and externalities. 10

Table 5.2-5 Estimates of benefit-cost ratios of pesticide use, including hidden costs and externalities. 10

About

In December 2017, Resolution 4 of the 3rd Session of the United Nations Environment Assembly (UNEA 3) requested "the Executive Director to present a report on the environmental and health impacts of pesticides and fertilizers and ways of minimizing them, given the lack of data in that regard, in collaboration with the World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO) and other relevant organizations by the fifth session of the United Nations Environment Assembly". In response to this request, UNEP published a *Synthesis Report on the Environmental and Health Impacts of Pesticides and Fertilizers and Ways to Minimize Them*[1] in February 2022 (United Nations Environment Programme [UNEP] 2022).

The overall goal of the synthesis report is to provide the information base to enable other advocacy actions to be taken by stakeholders to minimize the adverse impacts of pesticides and fertilizers. Specific objectives of the synthesis report are to:

- Update understanding of current pesticide and fertilizer use practices;
- Present major environmental and health effects of pesticides and fertilizers, during their life cycle, and identify key knowledge gaps;
- Review current management practices, legislation and policies aimed at reducing risks in the context of the global chemicals, environmental and health agenda;
- Identify opportunities to minimize environmental and health impacts, including proven and innovative approaches.

This chapter on "The environmental, human health and economic impacts of pesticides" is the 5th in a series of 12 chapters that make up a comprehensive compilation of scientific information. The chapters were developed to both inform and further elaborate on the information provided in the synthesis report. Please note that the disclaimers and copyright from the synthesis report apply

1 The Synthesis report is available at https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

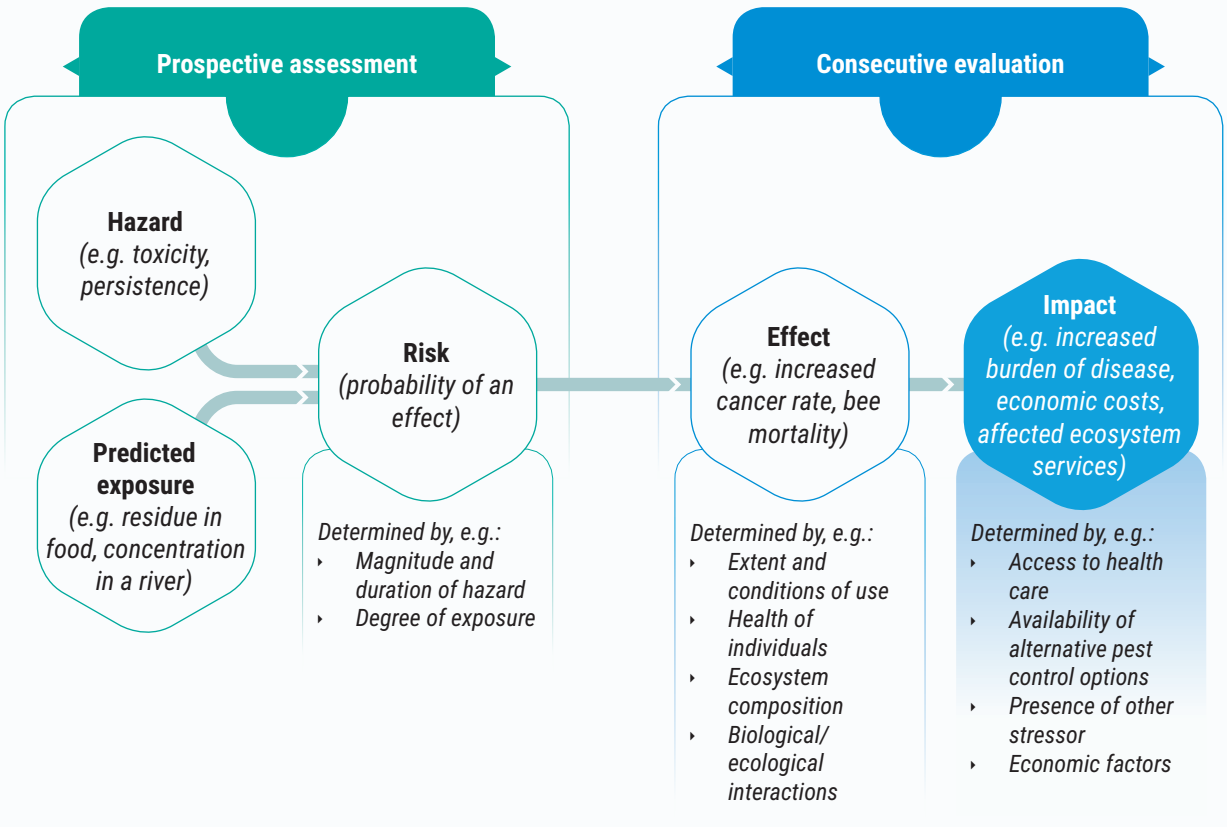
The environmental, human health and economic impacts of pesticides

5

5.1 Overview

In Chapter 4, steps in pesticide evaluation (Figure 5.1-1) were described, from the estimation of hazard and exposure to risk assessment, to impact evaluation. The effects of pesticides on the environment, health and agricultural sustainability were reviewed in that chapter. In this chapter available data about the ultimate impacts of pesticides on the environment and health are evaluated from three different perspectives:

Figure 5.1-1 The environmental and health impact of pesticide use consists of any durable changes in the condition of people or their environment brought about by the (adverse) effect(s) of a pesticide.



the burden of disease; monetary costs; and effects on ecosystem services. The impact of pesticides is defined as a durable change in the condition of the environment or people brought about by their (adverse) effect(s) (Chapter 4, Box 4.2-1).

Pesticides are sold and used because of the benefits they are expected to provide. The potential benefits of pest control (of which pesticides are an instrument) include reduced crop losses, reduced prevalence of human vector-borne diseases, longer shelf life of agricultural commodities, greater livestock yields, reduced soil disturbance, and better protection of wooden structures.

The overall adverse human health impact of pesticides can be quantified as burden of disease. So far, no international estimates are available of the burden of disease caused by pesticides, with the exception of self-poisoning (Prüss-Ustün *et al*. 2016a; Prüss-Ustün *et al*. 2016b; WHO 2016; Landrigan *et al*. 2018; WHO 2019). [Chapter 5.2]

No recent independent global or regional reviews of the economic benefits of pesticide use are available (Dobson 2007; Wiese and Steinman 2020). While the pesticide industry certainly collects information about the costs and benefits of their products internally, such data do not seem to be publicly available for systematic review and independent analysis. [Chapter 5.3.1]

Pesticide use has different types of costs: direct costs, which are all the monetary and non-monetary expenses borne by farmers and other pesticide users; indirect or hidden costs (e.g., occupational health effects, development of pest resistance, or reduction in crop pollination); and external costs or externalities, which are the costs of pesticide use borne by society as a whole (e.g., pesticide regulation, treatment of polluted water, or clean-up of stocks of obsolete pesticides) (Ajayi *et al*. 2002; Bourguet and Guillemaud 2016).

Despite great uncertainties associated with estimates of the indirect environmental and health costs of pesticide use, these costs are likely to be high. The most recent review of annual indirect environmental and health costs, published in 2016, found that they ranged from USD 5.5 million in Niger in 1996 to almost USD 12 billion in the United States in 2005. However, these were considered to be underestimates and to be based on outdated information (Bourguet and Guillemaud 2016). More recent estimates of the health costs of endocrine disrupting pesticides amount to tens of billions US dollars, in both Europe (Trasande *et al*. 2015; Trasande *et al*. 2016) and the United States (Attina *et al*. 2016). [Chapter 5.3.2]

Very few comprehensive assessments are available comparing the overall costs of pesticide use with their estimated benefits (Bourguet and Guillemaud 2016). If indirect costs are not taken into account, benefit-cost ratios at farm level tend to average between 3 and 6 (i.e., USD 1 in expenditure on pesticides and their application yields USD 3-6 in benefits to the farmer) (Fernandez-Cornejo, Jans and Smith 1998; Zilberman *et al*. 1991; Popp, Pető and Nagy 2013; Fernandez-Cornejo *et al*. 2014; Bourguet and Guillemaud 2016). If hidden private costs and externalities are included, benefit-cost ratios are much reduced and in some cases are below 1 (Bourguet and Guillemaud 2016). Most indirect costs are borne by society as a whole. They are generally not taken into account in decision-making about pest control at either the private or government level. [Chapter 5.3.3]

Pesticide use may affect ecosystem services, in particular pollination, soil function, pest regulation, food production and maintenance of future options. There is clear evidence that pesticides adversely affect the natural regulation of pests and other detrimental organisms (MA 2005). High levels of pesticide use also impact pollination (Dainese *et al*. 2019), although it is less clear whether sublethal exposure of pollinators leads to a reduction in pollination services (Kovács-Hostyánszki *et al*. 2016). The circumstances under which pesticide use affects soil functions are currently unclear and require further research (Dornbush

and von Haden 2017). Food and feed production can be positively or negatively influenced by pesticides, depending on the circumstances of their use.

Relatively few studies have been conducted on pesticides' impact on biodiversity and the associated capacity of (agro-)ecosystems to adapt to change. Where such studies have been carried out, biodiversity was generally shown to be adversely affected by pesticide use (Potts, Imperatriz-Fonseca and Ngo 2016; Stavi, Bel and Zaady 2016; Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services [IPBES] 2019). [Chapter 5.4]

Large gaps still exist in our understanding of pesticides' ultimate environmental, health and economic impact under current conditions of use. Thus, informed decision-making about the best and most sustainable forms of pest and vector management, and the role of pesticides therein, is handicapped by lack of the comprehensive knowledge needed to develop sound national and regional policies.

5.2 Burden of disease

The overall health impacts of pesticides can be quantified as the burden of disease attributable to pesticides. The burden of disease is generally expressed as deaths or as disability-adjusted life years (DALYs). One DALY can be thought of as one lost year of "healthy life". The sum of these DALYs across the population, or the burden of disease, is considered a measurement of the gap between current health status and an ideal health situation in which the entire population lives to an advanced age free of disease and disability. DALYs are calculated as the sum of the "years of life lost" due to premature mortality in the population and the "years lost due to disability" for people living with the health condition or its consequences (World Health Organization [WHO] n.d.).

The use of pesticides may also lower the burden of disease, for instance when used in the control of vectors of human disease such as malaria, or to reduce the production of mycotoxins on certain crops.

WHO has published global assessments of the burden of disease from environmental risks and environmental determinants of health, as well as the public health impact of chemicals (Prüss-Ustün *et al*. 2016a; Prüss-Ustün *et al*. 2016b; WHO 2016; WHO 2019). Estimates of unintentional poisoning with chemicals are provided, but so far have not been disaggregated for pesticides. The only pesticide-specific estimate in the WHO burden of disease publications is for intentional self-poisoning by pesticides (Chapter 4.4.4). The *Lancet* Commission on Pollution and Health (Landrigan *et al*. 2018) noted that the contribution of pesticides to the global burden of disease is not quantified despite their widespread use and potential effects on human health.

Fantke, Friedrich and Jolliet (2012) modelled the consumer health impact of 133 pesticides applied in 2003 in 24 European countries. They estimated that dietary exposure to these pesticides resulted in an overall burden of disease of 1,959 DALYs per year, for all 24 countries combined, corresponding to 2.3 minutes per person per year. They concluded that dietary pesticide exposure contributed little to the overall burden of disease in Europe. Just 13 pesticide active ingredients contributed to 90 per cent of the total number of DALYs; in early 2020 only three of these were still approved in the EU (European Union [EU] 2020).

Bellanger *et al*. (2015) and Attina *et al*. (2016) estimated the number of cases of disease resulting from exposure to endocrine disrupting chemicals in the EU and the United States in 2010 (Table 5.2-1). Prenatal exposure to organophosphate pesticides was associated with almost 60,000 cases of IQ loss and intellectual disability in Europe, but many fewer in the United States. Tens of thousands of cases of adult diabetes and fibroids in women were estimated

Table 5.2-1 Human disease burden associated with certain pesticides in the EU and the United States in 2010. Bellanger *et al.* (2015); Attina *et al.* (2016).

Pesticide or pesticide metabolite	Adverse health effect	Annual number cases	
		EU	United States
Organophosphate pesticides	IQ loss and intellectual disability	59,300	7,500
DDE	Childhood obesity	1,555	857
DDE	Adult diabetes	28,200	24,900
DDE	Fibroids	56,700	37,000

to have been caused by exposure to DDE on both continents. These estimates have come under criticism as being insufficiently based on toxicological evidence (Bond and Dittrich 2017), but so far alternative assessments have not been published.

In conclusion, the impact of exposure to pesticides on the burden of disease has barely been quantified so far, although certain pesticides are known to cause chronic disease while others are strongly associated with it (Chapter 4.4.3).

5.3 Monetary costs

5.3.1 The benefits of pesticide use

Pesticides are considered a critical component for the growth of agricultural productivity and food supply (National Research Council 2000; Pingali 2012; Popp, Pető and Nagy 2013; Organisation for Economic Co-operation and Development and Food and Agriculture Organization of the United Nations 2019). They are sold and used because of the benefits they are expected to provide. The potential benefits of pesticide use include reduced crop losses, prevention of the introduction of harmful pests, lower costs of agricultural production, contribution to food security, reduced prevalence of human vector-borne diseases, longer shelf life of agricultural commodities, greater livestock yields, reduced soil disturbance, and better protection of wooden structures.

Cooper and Dobson (2007) list almost 60 primary and secondary benefits of pesticides. These benefits are economic, social and environmental. However, it has been argued that this analysis confuses benefits that derive from control of a pest with those deriving from the use of a pesticide. As there are often several possible ways to control damage caused by pests, it is misleading to attribute all the benefits of pest control to pesticides (Edwards-Jones 2007). In many situations, pesticides offer advantages over other control methods, associated with characteristics such as ease of use, speed and consistency of control, and reduction of pest and disease risks. However, that does not mean pesticides necessarily offer greater net benefits than other control methods. It is only by comparing the costs and benefits of each control method in a given situation that the relative merits of different pest management options can be assessed (Edwards-Jones 2007; Waterfield and Zilberman 2012).

The economic benefits of pesticide use can be expressed in monetary terms. Such assessments may be made for individual pesticide-pest-crop situations and, more broadly, at the level of cropping systems or regions/countries. Deloitte (2018) estimated that Australian dollars (AUD) 20.6 billion of Australian agricultural output in 2015-16 was attributable to the use of crop protection products, or 73 per cent of the total value of crop production that year. Mark Goodwin Consulting Ltd. (n.d.), which assessed the value of increased yield and quality of field crops, nut/

fruit crops and vegetables in the United States between 2008 and 2010, estimated that using crop protection products added USD 81.8 billion in crop value or about 36 per cent of the total value of these crops.

Both of these studies calculated the share of agricultural output value attributable to pesticides on the basis of crop losses that would occur if pesticides were not used (based on Gianessi 2009 for insecticides). However, such assessments may overestimate losses due to pests, diseases and weeds since farmers will use alternative pest management options to compensate at least in part for the absence of pesticides (Chapters 2.7.2 and 2.7.5). On the other hand, there can also be an opportunity cost for alternatives such as labour that could have been spent on other income generating activities, children kept out of school (in some countries) to perform labour, or more contamination with mycotoxins.

The United States Environmental Protection Agency evaluates the benefits of new active ingredients or significant new uses of a pesticide as part of the registration process. It assesses potential advantages that can lead to greater flexibility of use for growers, better outcomes or lower costs (United States Environmental Protection Agency [US EPA] 2018)

No independent global or regional reviews appear to have been made of studies on the economic benefits of pesticide use at either individual pesticide level or a larger geographical scale. Cooper and Dobson (2007) noted that the economic benefits of pesticide use were not well documented in the scientific literature. Recently, Wiese and Steinman (2020) concluded that even in the case of glyphosate (currently the most widely used pesticide globally) no scientific basis exists for published economic calculations of its yield benefits in non-GMO arable farming in the European Union (EU). Only limited advances appear to have been made since Sexton, Lei and Zilberman (2007) reviewed the economics of pesticides and pest control and reported that "only now we are beginning to understand the productivity of pesticides, how productivity changes over time, and how environmental factors affect productivity".

While the pesticide industry certainly collects information about the costs and benefits of their products internally, such data do not appear to be systematically reviewed and independently analysed. Furthermore, such studies tend not to take into account other pest management options or the costs of externalities.

5.3.2 The costs of pesticide use

The use of pesticides may have different types of costs (Ajayi *et al.* 2002; Bourguet and Guillemaud 2016):

- *Direct costs.* These are all the monetary and non-monetary expenses incurred by a farmer or other pesticide user which are noticeably related to applying a pesticide (e.g., cost of the pesticide product and spray equipment, labour costs for application, pesticide registration costs insofar as they are included in the price of the pesticide product).

- *Indirect or hidden costs.* Pesticide use may cause indirect effects, such as occupational health effects, development of pest resistance leading to increased pesticide use, and reduction of crop pollination due to honeybee kills. The costs of such indirect or hidden effects are (mainly) borne by the pesticide user, either in the short or long term.

- *External costs or externalities.* These are costs of pesticide use borne by society as a whole (e.g., part of or all costs of regulatory control, increased costs of water treatment because of pesticide pollution, costs due to environmental effects of pesticides outside treated areas, health effects on residents or bystanders, clean-up costs for obsolete pesticide stocks).

A more extensive list of these types of costs is provided in Table 5.3-1.

Indirect costs and externalities of pesticide use are often not taken into account when farmers or other pesticide users decide to apply a pesticide. Such costs also tend not to be considered when national governments develop pest and pest management policies, although this is changing in some parts of the world.

Table 5.3-1 **Direct and indirect costs of pesticide use, as they may be borne by the pesticide user or society as a whole.** Based on Ajayi *et al.* (2002); Bourguet and Guillemaud (2016).

Category of costs	Type of costs	Direct private costs	Indirect costs	
			"hidden" private	"externalities"
		Borne by pesticide user	Borne by pesticide user	Borne by society as a whole
Pesticide application	Product	●		●[1]
	Transport	●		
	Labour	●		
	Equipment	●		●[1]
	Storage	●		
	Personal protective equipment	●		
Environmental costs	Pesticide resistance, leading to greater difficulty and/or expenses to manage pests		●	●[2]
	Decrease of pollination, resulting in crop yield/quality reduction and/or rental of bee colonies		●	●[2]
	Decrease in natural enemies, leading to pest resurgence or new pests		●	●[2]
	Decreased soil fertility leading to reduced yields		●	
	Adverse effects on livestock and domestic animals		●	●
	Impact on other economic activities (e.g. fisheries, beekeeping, silk production, organic food production) resulting from pesticide exposure			●
	Loss of biodiversity; effects on non-target organisms			●
	Environmental pollution: treatment or clean-up costs (e.g. drinking water treatment)			●
	Clean-up of pesticide spills, obsolete pesticide stocks and empty pesticide containers	●		●
Health costs	Medical costs after occupational poisoning		●	●[3]
	Loss of productivity after occupational poisoning of applicators and other workers.		●	●
	Medical costs after bystander/residential poisoning			●
	Loss of productivity after bystander/residential poisoning			●
	Health costs due to dietary exposure (pesticide residues in food or water)			●
Regulatory costs	Public research, communication, expertise on pesticides			●
	Pesticide regulation, registration, control, enforcement			●
	Mandatory pesticide handling practices (e.g. storage, disposal)		●	

1 In the case of subsidies or tax exemptions.
2 Pesticide resistance, pest resurgence and pollinator declines tend to develop area-wide and are not limited to the farmer's field.
3 In the case of government support to the health sector.

Examples of authoritative studies on the environmental, health and economic development costs of pesticide use include those conducted in the United States by Pimentel and colleagues at Cornell University (Pimentel and Burgess 2014) and work done in Africa, Asia, Europe and South America under the Pesticide Policy Project by Hermann Waibel and colleagues at the University of Hannover, Germany (Leibniz University Hannover n.d.), among others.

The UNEP report *Costs of Inaction on the Sound Management of Chemicals reviewed* studies that quantified indirect costs of the production and use of chemicals (United Nations Environment Programme [UNEP] 2013). It revealed that there were limited monetized and quantified data ready for use in policy decision-making. Nevertheless, a relatively a large fraction of available studies that are reasonably disaggregated with respect to chemicals tend to address pesticides.

Based on several studies from Sub-Saharan Africa (SSA), UNEP (2013) estimated the costs of lost work, medical treatment and hospitalizations due to pesticide poisoning among smallholder farm workers in 37 SSA countries at USD 4.4 billion in 2005 (range USD 1.4 billion- USD 8.1 billion). This would mean the health costs associated with pesticide use in Africa amounted to almost double the market value of crop protection products sold in the region (see Chapter 2, Figure 2.4-9). The UNEP study was limited to acute health and lost work costs resulting directly from working with pesticides on small landholdings. No attempt was made to estimate other costs. However, these other costs are likely to be substantial (e.g., the costs of bystander effects, suicide and self-poisoning, chronic health effects, lost livelihoods and lives, environmental impact and effects on farm animals).

Bourguet and Guillemaud (2016) conducted an extensive review of studies on the indirect environmental and health costs of pesticides. This review included 61 articles published between 1980 and 2014. Most studies that monetized adverse pesticide impacts concerned acute human health effects. Studies on the environmental costs of pesticides were much more limited. Only 10 independent data sets were available which combined estimates of environmental, health and regulatory costs at the national level (Table 5.3-2). Overall, indirect costs of pesticide use ranged from about USD 5.5 million in Niger in 1996 to almost USD 12 billion in the United States in 2005.

Bourguet and Guillemaud (2016) noted that the indirect costs of pesticide use were underestimated due to the fact that several costs had never been evaluated. In addition, they pointed out that these estimates were almost all outdated since the current impact of pesticide use was probably very different from that during the 1980s and 1990s. This is because of the difficulties of estimating the economic costs of unintentional impacts of pesticide use, particularly for goods without market values. Most estimates of external costs to date must therefore be considered as minimum costs. UNEP (2013) also stressed that monetized environmental and health effects are often underestimated, only cover one or a few types of costs, and hardly ever address costs with respect to human welfare.

Trasande *et al*. (2015, 2016) assessed the human health costs that could reasonably be attributed to exposure to endocrine disrupting chemicals in the EU. They focused on nine chemicals or groups of chemicals for which they considered there was sufficient evidence of adverse human health effects. Organophosphates and dichlorodiphenyldichloroethylene (DDE), a metabolite of dichlorodiphenyltrichloroethane (DDT), were the pesticides included in their evaluation. Attina *et al*. (2016) conducted a study that applied the same methodology in the United States.

In the EU the health costs of exposure to DDE ranged from several tens to hundreds of millions of euros (Table 5.3-3); prenatal exposure to organophosphates was costed at 47 billion to 195 billion euros, the highest cost of all nine groups of chemicals evaluated. In the United States health costs associated with organophosphate pesticide exposure were about USD 34 billion, about a quarter of those in the EU. This was due to lower exposure of the population to organophosphates, reportedly resulting from stricter regulations of this group of pesticides.

Table 5.3-2 Estimates of overall indirect costs of pesticide use. Bourguet and Guillemaud (2016).

Country	Year	Estimated annual costs in USD (2013) million			
		Health effects	Regulatory actions	Environmental effects	Total
Thailand	1995	1.26	558	5.58	565
Germany	1996	18	168	9.3	195
Niger	1996	4.40	0.15	0.89	5.44
United Kingdom	1996	2.3	319	63	384
Mali	1999	3.71	1.58	38.1	43.4
Pakistan	2002	78.1	9.71	815	903
United States*	1986	245-487	3,203	204-4,029	3,652-7,719
United States*	2002	1,309	4,989	1,470-1,508	7,768-7,806
United States*	2005	1,493	4,229	5,974	11,696
Thailand	2010	2.99	357	16.9	377

* Estimates for the United States are from independent authors and data sets.

The costs of adult diabetes as a result of DDE exposure, on the other hand, were almost double in the United States compared with the EU.

Based on these estimates, it was concluded that there is a substantial probability of very high disease costs across the human lifespan associated with exposure to certain pesticides, both in the EU and the United States. Given the limited number of pesticides that have been evaluated in this manner, however, the total health costs of pesticide exposure are likely to be considerably underestimated (Grandjean and Bellanger 2017). The reanalysis by Bourguet and Guillemaud (2016) of North American data published earlier suggested that each per cent of cancers attributable to pesticides was associated with a cost of about USD 20 billion annually,

Table 5.3-3 Estimates of the economic cost of exposure to certain endocrine disrupting pesticides in the EU (Trasande *et al.* 2015; Trasande *et al.* 2016) **and the United States** (Attina *et al.* 2016). **Costs consisted of hospitalization, physicians' services, nursing home care, medical appliances and related items.**

Pesticide or pesticide metabolite	Adverse health effect	Costs		
		Low estimate	Base estimate	High estimate
European Union		EUR million		
Organophosphates	IQ loss and intellectual disability	46,800	146,000	195,000
DDE	Childhood obesity	24.6	24.6	86.4
DDE	Adult diabetes	835	835	16,700
DDE	Fibroids	163	163	163
United States		USD million		
Organophosphates	IQ loss and intellectual disability	14,300	33,700	59,500
DDE	Childhood obesity	n.a.	29.6	57.3
DDE	Adult diabetes	n.a.	1,800	13,500
DDE	Fibroids	n.a.	259	595

n.a. = not available

confirming the very high costs of chronic illness to society.

Estimates of environmental costs of pesticide use are much more limited. Bourguet and Guillemaud (2016) found only five studies based on data recorded after 2000 and only one article published since 2006. Most estimates of environmental costs referred to losses of natural enemies and bees. However, this does not mean that environmental costs are less important than human health costs from a monetary point of view. Of the 10 data sets presenting the overall indirect costs of pesticides listed in Table 5.3-1, eight estimated (considerably) higher environmental than human health costs.

In the future, owing to various initiatives concerning the economic valuation of ecosystem services and biodiversity, better information will likely become available with which to estimate the environmental costs of pesticide use. For instance, the Economics of Ecosystems and Biodiversity (TEEB) Initiative is attempting to "make nature's values visible" and mainstream the values of biodiversity and ecosystem services into decision-making. TEEB follows a structured approach to valuation that is intended to help decision-makers recognize the benefits provided by ecosystems and biodiversity and demonstrate their values in economic terms.

As part of TEEB, the AgriFood programme seeks to provide a comprehensive economic evaluation of the "eco-agri-food systems" complex, and assess to what extent the economic environment in which farmers operate is distorted by significant externalities, both negative and positive (TEEB 2015). To date, no economic studies on the costs of pesticides with regard to biodiversity and ecosystems services have been conducted under TEEB. It is expected that data on this topic will become available in the near future.

Similarly, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) includes valuation of biodiversity and ecosystem services in its assessments. For instance, the IPBES assessment on pollinators, pollination and food production includes a detailed chapter on economic valuation of pollinator gains and losses (Gallai *et al.* 2016) which provides a summary of the economic values of pollination services from local to global scales.

5.3.3 Cost-benefit assessments

Agricultural pest management decisions are generally made at the farm level, although they may be constrained by government regulation and resource availability. The farmer's objective is assumed to be profit maximization, where profit is the revenue from selling output on the market after subtracting production costs (Waterfield and Zilberman 2012).

Determining a truly optimal pest control strategy requires full knowledge of locations-specific pest pressures, how pest pressure relates to crop damage, and the range of impacts of available pest control technologies including the use of specific pesticides. Farmers' decisions are further complicated by the dynamic changes in the cropping system, both over the course of the season and across seasons: current pest control decisions may have consequences far into the future (e.g., build-up of resistance, pest resurgence). Farmers also have different attitudes towards accepting risks respecting both the development of pests and diseases and investments needed for pest control. Deciding on an economically optimal pest control strategy is therefore very difficult for individual farmers (Waterfield and Zilberman 2012).

Ghimire and Woodward (2013) conducted a macro-economic analysis of under- and overuse of pesticides in 75 countries between 1990 and 2000. They found that, after correcting for agricultural and climatic variables, underuse of pesticides was predominant in low income countries while high income countries tended to overuse pesticides. More recent assessments appear to indicate that pesticide overuse occurs frequently in specific cropping systems in low and middle income countries, e.g., China (Zhang *et al.* 2015; Wu *et al.* 2018) and Thailand (Grovermann, Schreinemachers and Berger 2013; Schreinemachers *et al.* 2020) as well as high income countries, e.g., France (Lechenet *et al*. 2017), the Netherlands (Skevas, Stefanou and Oude Lansink 2014) and the

Table 5.2-4 Estimates of the net return of pesticide use, at farm level, without taking into account hidden costs and externalities.

Period	Region	Benefit-cost ratio	Source
1980-1991	Canada, France, United States	0.1-8.0	Fernandez-Cornejo, Jans and Smith (1998)
Before 1991	United States	3.0-6.5	Zilberman *et al*. (1991)
2002-2009	United States	6.5	Popp, Pető and Nagy (2013)
1980s-1990s	United States, Europe	1-8	Fernandez-Cornejo *et al*. (2014)
"most widely cited"	mainly United States	4	Bourguet and Guillemaud (2016)

Note: The benefit-cost ratio is the unit value increase in agricultural output for each unit value of aggregate pesticide expenditures (i.e. direct costs). A benefit-cost ratio >1 suggests that pesticide use is economically justified.

United States (US EPA 2014). No up-to-date comprehensive review is available. As a result, it has been argued that pesticide use reductions rarely cause yield losses (Vasileiadis 2017). Frisvold (2019), however, cautions against a possible bias associated with certain study methodologies and calls for the use of careful econometric assessments as the basis for reducing pesticide use.

Various evaluations exist of the net return, or net benefit, of pesticide use at farm level without taking into account hidden costs and externalities (Table 5.3-4). Benefit-cost ratios tend to average between 3 and 6, suggesting that USD 1 in pesticide expenditures yield USD 3-6 in benefits to the farmer. However, net benefits were highly variable, depending on crop, year and aggregation level (Fernandez-Cornejo, Jans and Smith 1998). Even the most recent review by Bourguet and Guillemaud (2016) mostly cites data that go back to the 1980s and 1990s. This is relevant since Fernandez-Cornejo, Jans and Smith (1998) and Fernandez-Cornejo *et al*. (2014) found that the net benefits of pesticide use fell between the 1950s and 1990s, while more recent independent economic data quantifying pesticide net benefits at farmer level do not seem to be available (see Chapter 5.3.1). Furthermore, most data appear to have been generated in North America.

It should be noted that the benefit-cost ratios reported here are comparisons of the average value of production (across all pesticides) against the average cost of (all) pesticides. In that respect, they are not necessarily a good representation of the private value of the use of a pesticide or of decision-making by the individual user. However, particular farmers' pesticide use decisions will generally not consider either the hidden costs to themselves or external costs imposed on society (Sexton, Lei and Zilberman 2007). These are many of the environmental, health and regulatory costs associated with pesticide use listed in Table 5.3-1. Calculating the net benefits to society

Table 5.2-5 Estimates of benefit-cost ratios of pesticide use, including hidden costs and externalities.
Based on Bourguet and Guillemaud (2016).

Year	Country	Annual costs or benefits, in USD million				Benefit-cost ratio
		Direct costs	Indirect costs (hidden and externalities)	Total costs	Benefits	
1996	Germany	1,309	196	1505	2199	1.46
2002	Pakistan	533	186	719	610	0.85
2005	United States	12,153	8,985	21,138	48,611	2.30
2013	United States	6,914	31,404	38,318	26,983	0.70

of pesticide use requires consideration of all costs and benefits.

Very few studies are available that compare the overall costs of pesticide use with their estimated benefits (Bourguet and Guillemaud 2016) (Table 5.3-5). When hidden costs and externalities are included in the calculations, benefit-cost ratios are much reduced compared with calculations based only on private costs. In the case from Pakistan and the 2013 re-estimation for the United States, the benefit-cost ratio was even below 1 (Bourguet and Guillemaud 2016).

Recently Lee *et al.* (2020) reviewed the cost-effectiveness of banning Highly Hazardous Pesticides (HHPs) to prevent suicides due to pesticide self-ingestion across 14 countries. They estimated that banning HHPs across these countries could result in up to 28,000 fewer suicide deaths per year at an annual cost of USD 0.007 per capita. National bans were found to be cost-effective in countries where a high proportion of suicides are attributable to pesticide self-poisoning.

It is important to bear in mind that rational decision-making on the use of pesticides requires a comparison of the costs and benefits of alternative farming systems in which different pest control methods are used. In this way farmers, and society, can decide which pest control method offers the most appropriate levels of costs and benefits (Edwards-Jones 2007; Bourguet and Guillemaud 2016). So far, comprehensive assessments of both the benefits and costs of pest control and pesticide use in specific farming systems have been rare.

5.4 Ecosystem services

5.4.1 Introduction

Ecosystem services have been defined as the benefits people obtain from ecosystems (Millennium Ecosystem Assessment [MA] 2005). The ways in which humans benefit from ecosystems and the consequences of ecosystem change for their well-being were reviewed globally for the first time by the Millennium Ecosystem Assessment, carried out between 2001 and 2005 (MA 2005). The presumption is that ecosystem service approaches provide a better basis for environmental decision-making than other approaches because they make explicit the connection between human well-being and ecosystem structures and processes (van Wensem *et al.* 2017)

Different typologies of ecosystem services have been proposed. The most common include the following (MA 2005; Food and Agriculture Organization of the United Nations and Secretariat of the Convention on Biological Diversity 2016):

- *Regulating services* are defined as the benefits obtained from the regulation of ecosystem processes such as climate regulation, natural hazard regulation, water purification and waste management, pollination and pest control.
- *Supporting services* are those that support the delivery of other services, such as soil formation and supplying habitat for species, which enable ecosystems to continue to supply provisioning and regulating services.
- *Provisioning services* refers to the goods and physical products obtained from ecosystems, such as food, freshwater, wood, fibre, genetic resources and medicines.
- *Cultural services* include non-material benefits that people obtain from ecosystems, such as spiritual enrichment, intellectual development, recreation and aesthetic values.

The MA mainly refers to pesticides as a threat to ecosystem services in general terms. It only mentions one service that, with high certainty,

has been degraded by the use of pesticides: natural pest regulation. In many agricultural areas, pest control provided by natural enemies has been replaced by the use of pesticides. Such pesticide use has itself degraded the capacity of agroecosystems to provide pest control (MA 2005).

The MA was succeeded by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (Chapter 3.2.5). One of the more recent key elements of the IPBES conceptual framework is the notion of nature's contributions to people (NCP), which builds on the ecosystem service concept popularized by the MA. While ecosystem services were based primarily on ecology and economics, NCP more broadly attempt to incorporate social, cultural and indigenous elements and experiences. Moreover, many ecosystem services fit into more than one of the four original categories (Díaz *et al.* 2018). The ecosystem services concept is nevertheless still applied by many institutions today.

NCP are all the contributions, both positive and negative, of living nature (i.e., diversity of organisms, ecosystems, and their associated ecological and evolutionary processes) to the quality of life for people. Beneficial contributions from nature include food provision, water purification, flood control and artistic inspiration. Detrimental contributions include disease transmission and predation that damages people or their assets, among others. Many NCP may be perceived as benefits or detriments depending on the cultural, temporal or spatial context (Intergovernmental Science-Policy Platform on

Box 5.4-1 IPBES reporting categories for nature's contributions to people (NCP) which are most susceptible to being impacted by the use of pesticides Díaz *et al.* (2018); Nienstedt *et al.* (2012).

Reporting category	How pesticides can affect nature's contributions to people (NCP)	Likelihood of adverse impact
2. Pollination and dispersal of seeds and other propagules	Pesticides can directly or indirectly reduce pollinator populations, leading to reduced pollination, which in turn can result in lower crop quality and yield as well as loss of diversity in natural vegetation.	Moderate to high (Chapter 4.3.3)
8. Formation, protection and decontamination of soils and sediments	Pesticides can affect soil organisms and plants which contribute to the formation of soils, the supply of organic matter and nutrient cycling. This in turn can affect soil fertility, leading to changes in the production of agricultural commodities, in forests, and in the diversity and production of natural vegetation.	Low to moderate (Chapter 4.3.2 and 4.3.4)
10. Regulation of detrimental organisms and biological processes	The main objective of using a pesticide is to reduce detrimental organisms, but pesticides can also adversely affect the natural capacity to regulate pests and diseases. Moreover, pesticide use can lead to the development of pest and disease resistance.	Moderate to high (Chapter 4.3.3 and 4.3.3)
12. Food and feed	Increased food production is the principal objective of agricultural pesticide use. However, effects on, for example, resistance, soils, pollinators and the natural enemies of pests can reduce food and feed production. Pesticide use in fish farming can increase production, but pollution may also reduce the sustainability of aquaculture as well as adversely affect aquatic ecosystems.	Low to high (Chapter 4.3.3 and 4.3.5)
18. Maintenance of options	Pesticides may affect biodiversity, which can in turn reduce the capacity of (agro-)ecosystems to keep options open to support good quality of life in the future, e.g. to adapt to change, new pests and diseases and the development of antibiotic resistance, or to produce new medicines or materials.	Low to high (Chapter 4.3.4, 4.3.5 and 4.3.6)

Note: Many other NCP or ecosystem services can also potentially be affected, but pesticides are likely to be less important compared to other stressors.

Biodiversity and Ecosystem Services [IPBES] 2019).

The 18 reporting categories which have been recommended for IPBES assessments partly overlap with the ecosystem services of the MA. Five of these categories are most likely to be directly affected by the use of pesticides: pollination, soil formation, regulation of detrimental organisms, production of food and feed, and maintenance of options (Box 5.4-1).

Ecosystem services are increasingly being considered in the risk assessment and management, as well as impact monitoring, of chemicals in general and pesticides in particular (US EPA 2016; European Food Safety Authority Scientific Committee 2016; Devos *et al.* 2019). Nevertheless, significant challenges still need to be addressed with regard to the use of ecosystem services-based risk assessments in regulatory decision-making (Munns *et al.* 2017).

5.4.2 The impact of pesticides on ecosystem services or nature's contributions to people

Pesticides are mentioned several times in *The Global Assessment Report on Biodiversity and Ecosystem Services* (IPBES 2019) as drivers of change affecting ecosystem services. However, specific cases are not provided. Evidence that pesticides are affecting ecosystem services from other sources is summarized below.

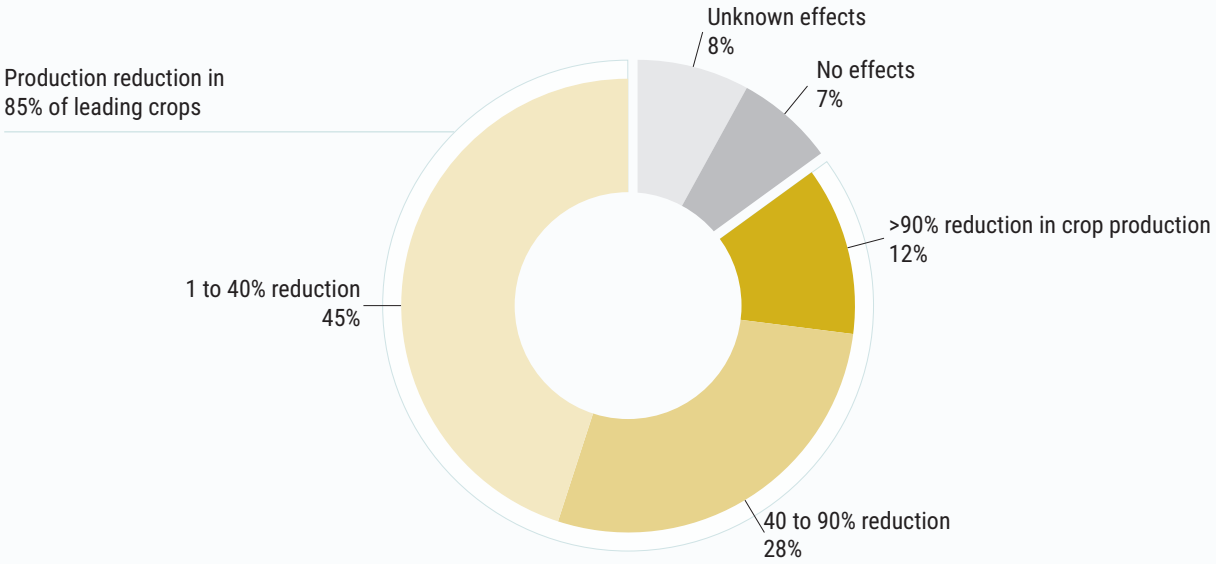
Pollination services

Biotic pollination is an ecosystem function that is fundamental to plant reproduction, agricultural production and the maintenance of terrestrial biodiversity. As part of IPBES, a specific assessment report on pollinators, pollination and food production was elaborated (Potts, Imperatriz-Fonseca and Ngo 2016). It outlines many ways in which pesticides can affect pollinators, as well as evidence of cases where this has occurred in real pesticide use situations (Chapter 4.3.3).

The link between the presence of pollinators and pollination services is generally very clear. The absence or reduction of pollination will result in a reduction in fruit set, crop yield and quality for many crops (Figure 5.4-1).

The most recent estimate of the global annual market value of additional crop production

Figure 5.4-1 Percentage dependence on animal-mediated pollination of leading global crops that are directly consumed by humans and traded on the global market. Potts, Imperatriz-Fonseca and Ngo (2016).



Note: Only crops that produce fruits or seeds for direct human use as food (107 crops) are included.

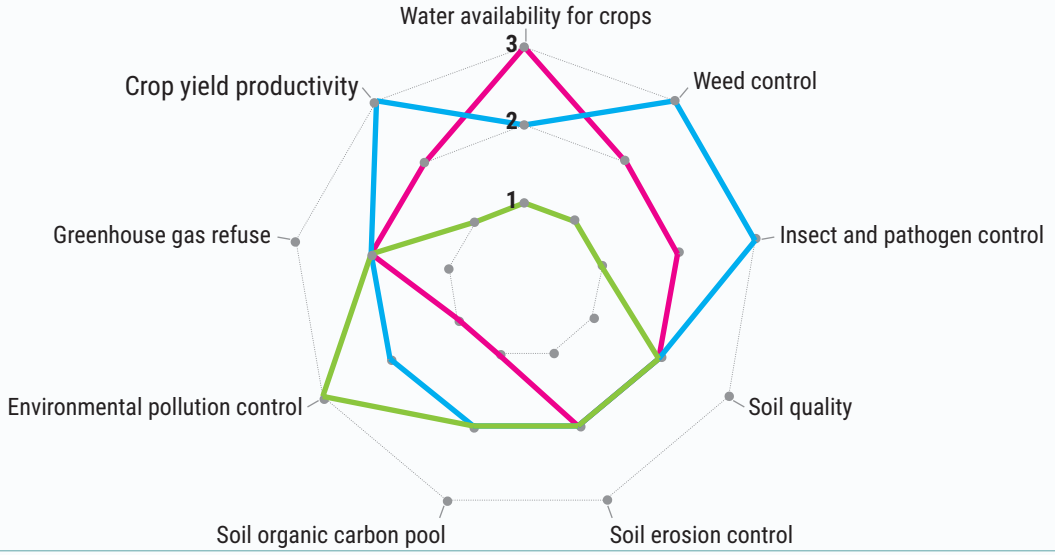
directly linked with pollination services ranges from USD 235-577 billion (in 2015 US dollars). In addition, in the absence of animal pollination, changes in global crop supplies could increase prices for consumers and reduce profits for producers, resulting in a potential annual net loss of economic benefits of USD 160-191 billion globally to crop consumers and producers, and a further USD 207-497 billion to producers and consumers in other, non-crop markets (e.g., non-crop agriculture, forestry and food processing). Nevertheless, current economic indicators fail to capture the full range of benefits provided by pollinators and the full costs of supporting managed pollinators (Gallai *et al.* 2016).

There is little argument that high levels of pesticide use in a pollinator dependent crop will result in a reduction in pollination (Potts, Imperatriz-Fonseca and Ngo 2016; Potts *et al.* 2016). This is the main reason why instructions such as to avoid spraying while crops are flowering may be put on a pesticide label. An example often cited is from Sichuan, China, where apples and pears are hand-pollinated because continuous insecticide applications were one of the main causes of the virtual extinction of bees in the area (Partap and Ya 2012).

Another issue is whether sublethal effects of pesticide exposure affect the provision of pollination. In the IPBES review, Kovács-Hostyánszki *et al.* (2016) suggest that there are indications that sublethal effects of pesticide exposure can impair the ability of bees to provide pollination, which could have wider implications for sustained production of pollinator-dependent crops and the reproduction of many wild plants. However, there is currently no evidence of such impacts on pollination under field conditions.

In conclusion, there is considerable evidence of acute adverse effects of pesticides on pollinators (Chapter 4.3.3). There is also a direct and strong link between pollinator abundance and diversity on the one hand and pollination as an ecosystem service on the other (Dainese *et al.* 2019). Whether

Figure 5.4-2 Spider chart of pest management's impact on soil functions and ecosystem services. Stavi, Bel and Zaady (2016).



Impact scale:	1	Low	2	Medium	3	High
Pest management system		Chemical		Integrated		Organic

Note: Impacts are separately presented for the three levels of intensity of pest management: chemical, integrated and organic. The major soil functions and ecosystem services are graded for each of the pest management intensities according to the scale of the following: 1 for low score for positive agro-environmental ranking, 2 for moderate score, and 3 for high score.

sublethal exposure of pollinators to pesticides leads to a reduction in pollination services has not been clearly shown so far.

Soil functions

Stavi, Bel and Zaady (2016) reviewed the impact of different cropping systems on soil functions and ecosystem services. The studies they evaluated were categorized, among others, according to three pest management approaches: chemical pest management, integrated pest management, and organic pest management. From an agro-environmental point of view, chemical pest management scored well on water availability for crops, but had a medium or low positive impact on all other soil functions and ecosystem services (Figure 5.4-2). Pest management in organic cropping systems had a highly positive impact on environmental pollution control, but a low impact on water availability for crops as well as on weed, insect and pathogen control. Integrated pest management scored best on weed, pest and disease control, as well as on crop yield productivity; it had the best average ranking overall for all soil functions and services.

However, our understanding of how pesticides affect soil functions and its implications for ecosystem services provided by soils is still very limited (Dornbush and von Haden 2017).

Natural regulation of pests and diseases

As indicated above, the Millennium Assessment considered natural pest regulation to be an ecosystem service that had undeniably been adversely affected by pesticide use (MA 2005). Pesticides kill or otherwise adversely affect the natural enemies of pests, which in turn may reduce biological control of pests and diseases.

Recent studies have confirmed that this ecosystem service continues to be compromised by the use of pesticides under current use conditions (Chapter 4.3.3).

No reviews of the economic impact of a reduction in biocontrol potential, or the costs of pest resurgence and secondary pest development resulting from pesticide use, were available. Nevertheless, Losey and Vaughan (2006) estimated the value of natural control attributable to insects of native agricultural pests in the United States at USD 4.5 billion. Roubos, Rodriguez-Saona and Isaacs (2014) noted that this type of information is not available for most cropping systems, but the limited data compiled more recently show that this earlier estimate may be much too low. Wyckhuys *et al*. (2020), who assessed the economic value of biological control across 23 countries in Asia and the Pacific, estimated that classical biological control, through introductions of natural enemies of pests, ensured annual accruing farm benefits of USD 14.6 to 19.5 billion per year in critical food, feed and fibre crops in these countries.

Production of food and feed

The principal objective of agricultural pesticide use is the maintenance or increase of food production. In many cases, pesticides have contributed to a reduction of crop losses and agricultural production growth (Chapters 2.7.1 and 2.7.3). However, pesticide use can also have adverse effects on the extent and sustainability of agricultural production, for example through the development of pest resistance and effects on soil processes, pollinators and natural pest control (Chapters 4.3.3 and 4.3.4). Pesticide use can also adversely affect other forms of food production such as fisheries and aquaculture (Chapter 4.3.5), while pesticide residues may influence the quality of food and feed (Chapter 4.4.6).

The use of pesticides will therefore have both a positive and negative impact on the production of food and feed as an ecosystem service.

As indicated above, relatively few comprehensive cost-benefit assessments of pesticide use are in the public domain. When only the direct costs of pesticides to the farmer are considered, the production benefits of using pesticides often – but not always – outweigh their costs. On the other hand, if indirect cost (externalities) are included, costs greatly increase and may surpass benefits (Chapter 5.3.3).

Maintenance of options

Pesticides may affect biodiversity, which in turn can reduce the capacity of (agro-)ecosystems to adapt to change, new pests and diseases, and the development of antibiotic resistance, or to produce new medicines or materials.

In major biodiversity status reports pesticides are often listed as a key driver of biodiversity loss in agricultural and natural ecosystems, but the number of studies able to directly link pesticide use with adverse effects on biodiversity parameters (such as species richness) are relatively rare. Moreover, the large majority of studies have been conducted in North America and Europe with an almost complete absence of data from other parts of the world (Chapter 4.3.6). Nevertheless, whenever large-scale studies or reviews have been available, most of them have shown adverse effects of pesticide use on biodiversity (Chapter 4.3.6).

References

Ajayi, O.C., Camera, M., Fleischer, G., Haidara, F., Sow, M., Traore, A. and Van Der Valk, H. (2002). *Socioeconomic Assessment of Pesticide Use in Mali*. Pesticide Policy Project, Special Issue Publication Series No 6. University of Hannover, Germany. https://www.ifgb.uni-hannover.de/fileadmin/ifgb/4_Materials_Research/PPP_Special_Paper_Series/ppp_s06.pdf.

Attina, T.M., Hauser, R., Sathyanarayana, S., Hunt, P.A., Bourguignon, J.P., Myers, J.P. *et al*. (2016). Exposure to endocrine-disrupting chemicals in the USA: A population-based disease burden and cost analysis. *The Lancet Diabetes and Endocrinology* 4(12), 996-1003. https://doi.org/10.1016/S2213-8587(16)30275-3.

Bellanger, M., Demeneix, B., Grandjean, P., Zoeller, R.T. and Trasande, L. (2015). Neurobehavioral deficits, diseases, and associated costs of exposure to endocrine-disrupting chemicals in the European Union. *Journal of Clinical Endocrinology and Metabolism* 100(4), 1256-1266. https://doi.org/10.1210/jc.2014-4323.

Bond, G.G. and Dietrich, D.R. (2017). Human cost burden of exposure to endocrine disrupting chemicals. A critical review. *Archives of Toxicology* 91, 2745-2762. https://doi.org/10.1007/s00204-017-1985-y.

Bourguet, D. and Guillemaud, T. (2016). The hidden and external costs of pesticide use. In *Sustainable Agriculture Reviews*. *Volume 19*. Springer International Publishing Switzerland. Chapter 12, 35-120. https://doi.org/10.1007/978-3-319-26777-7_2.

Cooper, J. and Dobson, H. (2007). The benefits of pesticides to mankind and the environment. *Crop Protection* 26(9), 1337-1348. https://doi.org/10.1016/j.cropro.2007.03.022.

Dainese, M., Martin, E.A., Aizen, M.A., Albrecht, M., Bartomeus, I., Bommarco, R. *et al*. (2019). A global synthesis reveals biodiversity-mediated benefits for crop production. *Science Advances* 5(10), eaax0121. https://doi.org/10.1126/sciadv.aax012.

Deloitte (2018). *Economic Activity Attributable to Crop Protection Products.* Report for CropLife Australia. Canberra: Deloitte Access Economics. https://www2.deloitte.com/content/dam/Deloitte/au/Documents/Economics/deloitte-au-economics-activity-attributable-crop-protection-010218.pdf.

Devos, Y., Munns Jr., W.R., Forbes, V.E., Maltby, L., Stenseke, M., Brussaard, L. *et al.* (2019). Applying ecosystem services for pre-market environmental risk assessments of regulated stressors. *EFSA Journal* 17(S1), e170705. European Food Safety Authority. https://doi.org/10.2903/j.efsa.2019.e170705.

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R.T., Molnár, Z. *et al*. (2018). Assessing nature's contributions to people – Recognizing culture, and diverse sources of knowledge, can improve assessments. *Science* 359 (6373), 270-272. https://doi.org/10.1126/science.aap8826.

Dornbush, M.E. and von Haden, A.C. (2017). Intensified agroecosystems and their effects on soil biodiversity and soil functions. In *Soil Health and Intensification of Agroecosystems*. Al-Kaisi, M.M. and Lowery, B. (eds.). Academic Press. Chapter 8, 173-193. https://doi.org/10.1016/B978-0-12-805317-1.00008-7.

Edwards-Jones, G. (2007). Do benefits accrue to "pest control" or "pesticides?': A comment on Cooper and Dobson. *Crop Protection* 27(6), 965-967. https://doi.org/10.1016/j.cropro.2007.11.018.

European Food Safety Authority Scientific Committee (2016). Guidance to develop specific protection goals options for environmental risk assessment at EFSA, in relation to biodiversity and ecosystem services. *EFSA Journal* 14(6), 4499. European Food Safety Authority. https://doi.org/10.2903/j.efsa.2016.4499.

European Union (2020). EU pesticides database. European Commission, Brussels, Belgium. https://ec.europa.eu/food/plant/pesticides/eu-pesticides-database/active-substances/?event=search.as. Accessed 2 January 2021.

Fantke, P., Friedrich, R. and Jolliet, O. (2012). Health impact and damage cost assessment of pesticides in Europe. *Environment International* 49, 9-17. https://doi.org/10.1016/j.envint.2012.08.001.

Fernandez-Cornejo, J., Jans, S. and Smith, M. (1998). Issues in the economics of pesticide use in agriculture: A review of the empirical evidence. *Review of Agricultural Economics* 20(2), 462-488. https://doi.org/10.2307/1350002.

Fernandez-Cornejo, J., Nehring, R., Osteen, C., Wechsler, S., Martin, A. and Vialou, A. (2014). *Pesticide Use in U.S. Agriculture: 21 Selected Crops, 1960-2008. Economic Information Bulletin 124*. Washington, D.C.: United States Department of Agriculture, Economic Research Service. https://doi.org/10.2139/ssrn.2502986.

Food and Agriculture Organization of the United Nations and Secretariat of the Convention on Biological Diversity (2016). *Mainstreaming Ecosystem Services and Biodiversity into Agricultural Production and Management in East Africa. Technical Guidance Document.* Rome. http://www.fao.org/3/a-i5603e.pdf.

Frisvold, G.B. (2019) How low can you go? Estimating impacts of reduced pesticide use. *Pest Management Science* 75, 1223-1233. https://doi.org/10.1002/ps.5249.

Gallai, N., Garibaldi, L.A., Li, X., Breeze, T.D., Espírito Santo, M.M., Rodriguez Fernandez, J. *et al*. (2016). Economic valuation of pollinator gains and losses. In *The Assessment Report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on Pollinators, Pollination and Food production.* Potts, S.G., Imperatriz-Fonseca, V.L. and Ngo, H.T. (eds.). Bonn: Secretariat of the Intergovernmental Science- Policy Platform on Biodiversity and Ecosystem Services. Chapter 4, 205-273. https://www.ipbes.net/assessment-reports/pollinators.

Ghimire, N. and Woodward, R.T. (2013). Under- and over-use of pesticides: An international analysis. *Ecological Economics* 89, 73-81. http://dx.doi.org/10.1016/j.ecolecon.2013.02.003.

Gianessi, L.P. (2009). *The Value of Insecticides in U.S. Crop Production. Executive Summary*. Washington, D.C.: CropLife Foundation. https://croplifefoundation.files.wordpress.com/2012/07/insecticide-benefits-execsum.pdf.

Grandjean, P. and Bellanger, M. (2017). Calculation of the disease burden associated with environmental chemical exposures: Application of toxicological information in health economic estimation. *Environmental Health* 16, 123. https://doi.org/10.1186/s12940-017-0340-3.

Grovermann, C., Schreinemachers, P. and Berger, T. (2013). Quantifying pesticide overuse from farmer and societal points of view: An application to Thailand. *Crop Protection* 53, 161e168. http://dx.doi.org/10.1016/j.cropro.2013.07.013.

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (2019). *The Global Assessment Report on Biodiversity and Ecosystem Services: Summary for Policymakers.* Díaz, S., Settele, S., Brondízio, E.S., Ngo, H.T., Guèze, M., Agard, J. *et al.* (eds.). Bonn: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://ipbes.net/sites/default/files/2020-02/ipbes_global_assessment_report_summary_for_policymakers_en.pdf.

Kovács-Hostyánszki, A., Li, J., Pettis, J., Settele, J., Aneni, T., Espíndola, A. *et al.* (2016). Drivers of change of pollinators, pollination networks and pollination. In *The Assessment Report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on Pollinators, Pollination and Food production.* Potts, S.G., Imperatriz-Fonseca, V.L. and Ngo, H.T. (eds.). Bonn: Secretariat of the Intergovernmental Science- Policy Platform on Biodiversity and Ecosystem Services. Chapter 2, 27-149. https://www.ipbes.net/assessment-reports/pollinators.

Landrigan, P.J., Fuller, R., Acosta, N.J.R., Adeyi, O., Arnold, R., Basu, N.N. *et al*. (2018). The *Lancet* Commission on Pollution and Health. *The Lancet* 391(10119), P462-P512. https://www.thelancet.com/commissions/pollution-and-health.

Lechenet, M., Dessaint, F., Py, G., Makowski, D. and Munier-Jolain, N. (2017). Reducing pesticide use while preserving crop productivity and profitability on arable farms. *Nature Plants* 3, 17008. https://doi.org/10.1038/nplants.2017.8.

Lee, Y.Y., Chisholm, D., Eddleston, M., Gunnell, D., Fleischmann, A., Konradsen, F. *et al.* (2020). The cost-effectiveness of banning highly hazardous pesticides to prevent suicides due to pesticide self-ingestion across 14 countries: a economic modelling study. *The Lancet Global Health* 9, (3), e291-e300. https://doi.org/10.1016/S2214-109X(20)30493-9.

Leibniz University Hannover (n.d.). The Pesticide Policy Project (PPP), Institute of Development and Agricultural Economics. https://www.ifgb.uni-hannover.de/en/research/completed-research-projects/pesticide-policy-project/. Accessed 7 January 2021

Losey. J.E. and Vaughan, M. (2006). The economic value of ecological services provided by insects. *BioScience* 56(4), 311. https://doi.org/10.1641/0006-3568(2006)56[311:tevoes]2.0.co;2.

Mark Goodwin Consulting Ltd. (n.d.). *The Contribution of Crop Protection Products to the United States Economy.* Washington, D.C.: CropLife America. https://static1.squarespace.com/static/59b55b2b37c581fbf88309c2/t/5a2a8074f9619a97da953a70/1512734840313/The+Contribution+of+Crop+Protection+Products+to+the+US+Economy.pdf.

Millennium Ecosystem Assessment (2005). *Ecosystems and Human Well-being: Synthesis.* Washington, D.C.: Island Pres. https://www.millenniumassessment.org/en/Reports.html.

Munns Jr., W.R., Poulsen, V., Gala, W.R., Marshall, S.J., Rea, A.W., Sorensen, M.T. and von Stackelberg, K.J. (2017). Ecosystem services in risk assessment and management. *Integrated Environmental Assessment and Management* 13(1), 62-73. https://doi.org/10.1002/ieam.1835.

National Research Council (2000). *The Future Role of Pesticides in US Agriculture.* Washington, D.C.: National Academies Press. https://doi.org/10.17226/9598.

Nienstedt, K.M., Brock, T.C.M., van Wensem J., Montforts, M., Hart, A., Aagaard, A. *et al.* (2012). Development of a framework based on an ecosystem services approach for deriving specific protection goals for environmental risk assessment of pesticides. *Science of The Total Environment* 415, 31-38. https://doi.org/10.1016/j.scitotenv.2011.05.057.

Organisation for Economic Co-operation and Development and Food and Agriculture Organization of the United Nations (2019). *OECD-FAO Agricultural Outlook 2019-2028.* Paris and Rome. https://doi.org/10.1787/agr_outlook-2019-en.

Partap, U. and Ya, T. (2012). The human pollinators of fruit crops in Maoxian County, Sichuan, China. *Mountain Research and Development* 32(2), 176-186. https://doi.org/10.1659/MRD-JOURNAL-D-11-00108.1.

Pimentel, D. and Burgess, M. (2014). Environmental and economic costs of the application of pesticides primarily in the United States. In *Integrated Pest Management – Pesticide Problems. Volume 3*. Pimentel, D. and Peshin, R. (eds.). Dordrecht: Springer. Chapter 2, 47-72. https://www.springer.com/gp/book/9789400777958.

Pingali, P.L. (2012). Green Revolution: Impacts, limits, and the path ahead. *Proceedings of the National Academies of Sciences of the United States* 109(31), 12302-12308. https://www.pnas.org/cgi/doi/10.1073/pnas.0912953109.

Popp, J., Pető, K. and Nagy, J. (2013). Pesticide productivity and food security. A review. *Agronomy for Sustainable Development* 33, 243-255. https://doi.org/10.1007/s13593-012-0105-x.

Potts, S.G., Imperatriz-Fonseca, V., Ngo, H.T., Aizen, M.A., Biesmeijer, J.C., Breeze, T.D. *et al*. (2016). Safeguarding pollinators and their values to human well-being. *Nature* 540, 220-229. https://doi.org/10.1038/nature20588.

Potts, S.G., Imperatriz-Fonseca, V.L. and Ngo, H.T. (eds.) (2016). *Assessment Report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on Pollinators, Pollination and Food Production.* Bonn: Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://ipbes.net/assessment-reports/pollinators.

Prüss-Ustün, A., Wolf, J., Corvalán, C., Bos, R. and Neira, M. (2016a). *Preventing Disease through Healthy Environments: A Global Assessment of the Burden of Disease from Environmental Risks.* Geneva: World Health Organization. https://www.who.int/publications/i/item/9789241565196.

Prüss-Ustün, A., Wolf, J., Corvalán, C., Neville, T., Bos, R. and Neira, M. (2016b). Diseases due to unhealthy environments: An updated estimate of the global burden of disease attributable to environmental determinants of health. *Journal of Public Health* 39(3), 464-475. https://doi.org/10.1093/pubmed/fdw085.

Roubos, C.R., Rodriguez-Saona, C. and Isaacs, R. (2014). Mitigating the effects of insecticides on arthropod biological control at field and landscape scales. *Biological Control* 75, 28-38. https://doi.org/10.1016/j.biocontrol.2014.01.006.

Schreinemachers, P., Grovermann, C., Praneetvatakul, S., Heng, P., Nguyen, T.T.L., Buntong, B. *et al.* (2020). How much is too much? Quantifying pesticide overuse in vegetable production in Southeast Asia. *Journal of Cleaner Production* 244, 118738. https://doi.org/10.1016/j.jclepro.2019.118738.

Sexton, S.E., Lei, Z. and Zilberman, D. (2007) The economics of pesticides and pest control. *International Review of Environmental and Resource Economics* 1(3), 271-326. https://www.nowpublishers.com/article/Details/IRERE-0007.

Skevas, T., Stefanou, S.E. and Oude Lansink, A. (2014), Pesticide use, environmental spillovers and efficiency: A DEA risk-adjusted efficiency approach applied to Dutch arable farming. *European Journal of Operational Research* 237, 658-664. http://dx.doi.org/10.1016/j.ejor.2014.01.046.

Stavi, I., Bel, G. and Zaady, E. (2016). Soil functions and ecosystem services in conventional, conservation, and integrated agricultural systems. A review. *Agronomy for Sustainable Development* 36, 32. https://doi.org/10.1007/s13593-016-0368-8.

The Economics of Ecosystems and Biodiversity (2015). *TEEB for Agriculture & Food: An Interim Report*. http://teebweb.org/our-work/agrifood/reports/preliminary-reports-exploratory-studies/.

Trasande, L., Zoeller, R.T., Hass, U., Kortenkamp, A., Grandjean, P., Myers, J.P. *et al.* (2015). Estimating burden and disease costs of exposure to endocrine-disrupting chemicals in the European Union. *Journal of Clinical Endocrinology and Metabolism* 100(4), 1245-1255. https://doi.org/10.1210/jc.2014-4324.

Trasande, L., Zoeller, R.T., Hass, U., Kortenkamp, A., Grandjean, P., Myers, J.P. *et al.* (2016). Burden of disease and costs of exposure to endocrine disrupting chemicals in the European Union: An updated analysis. *Andrology* 4(4), 565-572. https://doi.org/10.1111/andr.12178.

United Nations Environment Programme (2013). *Costs of Inaction on the Sound Management of Chemicals.* Geneva. https://www.unenvironment.org/resources/report/costs-inaction-initiative.

United States Environmental Protection Agency (2014). *Benefits of Neonicotinoid Seed Treatments to Soybean Production*. Washington D.C: US EPA Biological and Economic Analysis Division. https://www.epa.gov/pollinator-protection/benefits-neonicotinoid-seed-treatments-soybean-production.

United States Environmental Protection Agency (2016). *Generic Ecological Assessment Endpoints (GEAEs) for Ecological Risk Assessment: Second Edition with Generic Ecosystem Services Endpoints Added*. https://www.epa.gov/sites/production/files/2016-08/documents/geae_2nd_edition.pdf.

United States Environmental Protection Agency (2018). *Suggestions for Applicants on Information Considered when Evaluating Benefits of New Conventional Active Ingredients/Significant New Use Site Requests*. https://www.epa.gov/pesticide-registration/suggestions-applicants-information-considered-when-evaluating-benefits-new.

van Wensem, J., Calow, P., Dollacker, A., Maltby, L., Olander, L., Tuvendal, M. and Van Houtven, G. (2017). Identifying and assessing the application of ecosystem services approaches in environmental policies and decision making. *Integrated Environmental Assessment and Management* 13(1), 41-51. https://doi.org/10.1002/ieam.1836.

Vasileiadis, V.P. (2017). Less pesticide rarely causes loss. *Nature Plants* 3, 17016. https://doi.org/10.1038/nplants.2017.16.

Waterfield, G. and Zilberman, D. (2012). Pest management in food systems: An economic perspective. *Annual Review of Environment and Resources* 37, 223-45. https://doi.org/10.1146/annurev-environ-040911-105628.

Wiese, A. and Steinmann, H.-H. (2020). Yield effects associated with glyphosate use in non-GMO arable farming: A review. *Crop Protection* 134, 105148. https://doi.org/10.1016/j.cropro.2020.105148.

World Health Organization (2016). *The Public Health Impact of Chemicals: Knowns and Unknowns*. Geneva. https://apps.who.int/iris/handle/10665/206553.

World Health Organization (2019). *The Public Health Impact of Chemicals: Knowns and Unknowns: Data Addendum for 2016.* Geneva. https://apps.who.int/iris/handle/10665/279001.

World Health Organization (n.d.). Health statistics and information systems – Metrics: Disability-Adjusted Life Year (DALY). https://www.who.int/data/gho/data/themes/mortality-and-global-health-estimates. Accessed 13 December 2020.

Wu, Y., Xi, X., Tang, X., Luo, D., Gu, B., Lam S.K. *et al*. (2018). Policy distortions, farm size, and the overuse of agricultural chemicals in China. *Proceedings of the National Academies of Sciences of the United States* 115(27), 7010-7015. https://doi.org/10.1073/pnas.1806645115.

Wyckhuys, K.A.G., Lu, Y., Zhou, W., Cock, M.J.W., Naranjo, S.E., Fereti, A. *et al*. (2020). Ecological pest control fortifies agricultural growth in Asia- Pacific economies. *Nature Ecology and Evolution* 4(11), 1522-1530. https://doi.org/10.1038/s41559-020-01294-y.

Zhang, C., Hu, R., Shi, G., Jin, Y., Robson, M.G. and Huang, X. (2015) Overuse or underuse? An observation of pesticide use in China. *Science of The Total Environment* 538, 1-6. http://dx.doi.org/10.1016/j.scitotenv.2015.08.031.

Zilberman, D., Schmitz, A., Casterline, G., Lichtenberg, E. and Siebert, J.B. (1991). The economics of pesticide use and regulation. *Science* 253 (5019), 518-522. https://doi.org/10.1126/science.253.5019.518.

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them

Envisioning A Chemical-Safe World

Chapter 6 of 12

Current pesticide risk reduction and risk management

Contents

1 Global drivers, actors and policies affecting pesticides and fertilizer use

2 Status and trends of pesticide use

3 The regulatory and policy environment for pesticide management

4 Environmental and health effects of pesticide use

5. The environmental, human health and economic impacts of pesticides

About ii

6 Current pesticide risk reduction and risk management 1

6.1 Managing and reducing the environmental and health risks of pesticides 1

6.2 Assessing current risk reduction 2

6.2.1 Regulatory measures 2
6.2.2 Market-based measures 6
6.2.3 Training and awareness building 8
6.2.4 Engineering controls and technology 9
6.2.5 The effectiveness of current risk reduction 11

6.3 Strengthening pesticide management: options for actions 14

6.3.1 Strengthening governance of the production, trade and use of pesticides 14
6.3.2 Strengthening monitoring of pesticide use and effects 15
6.3.3 Strengthening pesticide evaluation 16
6.3.4 Innovate pest management through targeted research and development (R&D) 17
6.3.5 Inform and educate for change 18
6.3.6 The need for policy coherence 18

References 20

7 Status and trends of fertilizer use

8 The regulatory and policy environment of fertilizer management and use

9 Environmental and health effects of fertilizer use

10 The impact of fertilizer use

11 Current fertilizer risk reduction and risk management

12 Transformative actions to minimize the adverse impacts of pesticide and fertilizers

List of Figures

Figure 6.2-1 The hierarchy of controls as applied for occupational risks by the United States National Institute for Occupational Safety and Health (NIOSH). 11

List of Tables

Table 6.2-1 Regulatory and policy measures used to reduce the environmental and health risks of pesticides: strengths, limitations, opportunities and challenges. 2

Table 6.2-2 Economic measures used to reduce the environmental and health risks of pesticides: strengths, limitations, opportunities and challenges. 7

Table 6.2-3 Training and awareness building used to reduce the environmental and health risks of pesticides: strengths, limitations, opportunities and challenges. 9

Table 6.2-4 Engineering controls and technologies that can be used to reduce the environmental and health risks of pesticides: strengths, limitations, opportunities and challenges. 10

Table 6.2-5 Examples of pesticide risk mitigation measures for environmental and human health risks at different levels in the hierarchy of controls. 12

Table 6.2-6 Effectiveness of pesticide risk reduction measures during the last two decades and overall trend, with a focus on low and lower-middle income countries. 13

About

In December 2017, Resolution 4 of the 3rd Session of the United Nations Environment Assembly (UNEA 3) requested "the Executive Director to present a report on the environmental and health impacts of pesticides and fertilizers and ways of minimizing them, given the lack of data in that regard, in collaboration with the World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO) and other relevant organizations by the fifth session of the United Nations Environment Assembly". In response to this request, UNEP published a *Synthesis Report on the Environmental and Health Impacts of Pesticides and Fertilizers and Ways to Minimize Them*[1] in February 2022 (United Nations Environment Programme [UNEP] 2022).

The overall goal of the synthesis report is to provide the information base to enable other advocacy actions to be taken by stakeholders to minimize the adverse impacts of pesticides and fertilizers. Specific objectives of the synthesis report are to:

- Update understanding of current pesticide and fertilizer use practices;
- Present major environmental and health effects of pesticides and fertilizers, during their life cycle, and identify key knowledge gaps;
- Review current management practices, legislation and policies aimed at reducing risks in the context of the global chemicals, environmental and health agenda;
- Identify opportunities to minimize environmental and health impacts, including proven and innovative approaches.

This chapter on "Current pesticide risk reduction and risk management" is the 6th in a series of 12 chapters that make up a comprehensive compilation of scientific information. The chapters were developed to both inform and further elaborate on the information provided in the synthesis report. Please note that the disclaimers and copyright from the synthesis report apply

1 The Synthesis report is available at https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

Current pesticide risk reduction and risk management

6.1 Managing and reducing the environmental and health risks of pesticides

The sound management of chemicals, including hazardous waste, aims to prevent and, where that is not feasible, reduce or minimize the potential for exposure of the environment and people to hazardous chemicals. Sound management includes the prevention, reduction, remediation, minimization and elimination of risks during the life cycle of chemicals (production, storage, transport, use and disposal) and of risks posed by the chemicals in products and articles (United Nations Environment Programme [UNEP] 2013; UNEP 2019).

In this chapter current measures to reduce pesticide risks are reviewed. The strengths and limitations of various types of measures are assessed, and key opportunities and constraints with regard to future effective implementation are identified. Much of the chapter is based on reviews and analyses in the previous chapters.

In Chapter 6.2 current risk reduction is described under regulatory measures; market-based measures; training and awareness building; and engineering controls and technology. Its effectiveness of current risk reduction is appraised in a summary format. It should be emphasized that this appraisal may diverge, in part depending on situations in regions, countries or locally. Moreover, although the appraisal is based on inputs by many experts and stakeholders, it remains open to differing views.

On the basis of this appraisal (together with inputs obtained through stakeholder consultations), various options to strengthen pesticide risk reduction are identified in Chapter 6.3. These options are intended to enhance existing risk reduction measures. They build on current experiences in different parts of the world and often do not require complex interventions. However, the ultimate impact that such measures have on the environmental and human health risks of pesticide use are likely to be limited.

More fundamental changes in the ways pests and pesticides are managed are proposed in Chapter 12. These transformative actions are also more comprehensive. They address broader linkages and root causes. To be successfully implemented, transformative actions will require more time and greater effort in all sectors. Nevertheless, the long-term impact of such actions on the adoption of sustainable pest management is likely to be greater.

6.2 Assessing current risk reduction

6.2.1 Regulatory measures

International conventions

Several international conventions address pesticides directly or indirectly. Their contributions to reducing the risks of pesticide use have been variable or not specifically assessed (Chapter 3.2.2). One of the main constraints of many conventions is that their success largely depends on national capacities to implement their provisions and on levels of commitment to do so. Another limitation is that relatively few current use pesticides are addressed under the conventions which directly pertain to pesticides.

On the other hand, these conventions are existing, operational international legal frameworks that address certain aspects of pesticide manufacturing, distribution and use. Most countries in the world are now a Party to them. Therefore, they present an opportunity to strengthen international and transboundary aspects of pesticide management.

Voluntary international instruments and mechanisms

Several voluntary international instruments and mechanisms specifically address pesticides. As in the case of legally binding conventions, their contributions to pesticide risk reduction have not been systematically evaluated (Chapter 3.2.3).

The advantage of voluntary instruments over conventions or treaties is that they tend to cover a broader range of standards and guidance relevant to the sound management of pesticides. Thus, they potentially have a more wide-ranging impact on national pesticide management. The disadvantage of voluntary instruments is that they lack compliance mechanisms, which can make their implementation less effective and more sensitive to national policy changes.

Voluntary instruments and mechanisms such as the Food and Agriculture Organization of the United Nations and World Health Organization (FAO/WHO) International Code of Conduct on

Table 6.2-1 Regulatory and policy measures used to reduce the environmental and health risks of pesticides: strengths, limitations, opportunities and challenges.

Measure	Strengths	Limitations	Opportunities	Challenges
International conventions Chapter 3.2	Address transboundary pesticide risks and set international standards Promote strengthening of national legislation Most countries are Parties to these conventions	The objectives and obligations of respective international conventions are by definition limited in scope Pesticide issues may exist that are outside the scope of current conventions	International frameworks are in place which can, in principle, be strengthened Global coordination, communication and information sharing are in place	Contribution to risk reduction largely depends on national priorities and capacity for implementation Addressing risk reduction measures outside the scope of respective conventions is limited and not obligatory
Regional instruments and mechanisms Chapter 3.3	Assembling countries that face similar situations facilitates mutual understanding and collaboration	Effectiveness of regional activities still highly dependent on national implementation	Regional collaboration mechanisms may be increasingly established	Mobilization of funding for regional structures and activities

Measure	Strengths	Limitations	Opportunities	Challenges
Highly Hazardous Pesticides (HHPs) risk reduction Chapter 3.2	Potentially high impact for limited investments	Addresses a limited number of pesticides; others may also pose high risks	Global Action Plan for HHPs risk reduction developed under the Strategic Approach to International Chemicals Management (SAICM)	Creating a sense of urgency on the part of policymakers Identifying cost-effective alternatives
National pesticide legislation Chapter 3.4	Provides a legal basis to implement sound pesticide management	Public health and other non-agricultural pesticides are often inadequately covered Implementation and enforcement weak in many countries	Most countries have some form of pesticide legislation in place Successes exist with national bans on HHPs for the purpose of suicide prevention	Effective implementation of legislation requires political awareness and support
Pest and pesticide management policy Chapter 3.4	Clarifies pest management and pesticide use goals and targets for all stakeholders	Very few countries have established dedicated, stand-alone national policies	Lessons can be learned from countries that have experience with such policies	Formulating policies that are applicable within broader national priorities and development plans Converting adopted policy into concrete actions
Control and enforcement Chapter 3.4	One of the main tools to ensure effectiveness of legislation	Even if legislation is in place, control and enforcement are often inadequate Coordination among responsible government entities is often inadequate	Lessons can be learned from countries with effective control and enforcement systems	Establishing or strengthening effective enforcement systems with only limited human and financial resources
Pesticide evaluation and registration Chapter 3.4	Ensures that pesticides authorized for use are effective and do not pose unacceptable risks, based on the available knowledge	Human and financial capacities for pesticide registration in low income countries are very limited Duplication of pesticide evaluations by different national regulatory authorities	Pesticide regulatory evaluations increasingly publicly available More regional collaboration for pesticide registration	Optimizing the use of limited resources to conduct efficacy and risk evaluations of pesticides, both nationally and internationally
Post-registration monitoring Chapter 4.3.7	Allows risk mitigation measures to be taken if adverse environmental, health, agronomic or economic effects are observed	Post-registration monitoring of pesticide use and impact is almost absent in low and middle income countries	Monitoring of pesticide sales or use is increasingly required Experiences exist with low-cost, community-driven monitoring	Integrating monitoring into the pesticide regulatory system Developing cost-effective monitoring systems, especially in low and middle income countries

Pesticide Management, the FAO/WHO Codex Alimentarius, the Globally Harmonized System of Classification and Labelling of Chemicals (GHS) of the United Nations, and WHO's pre-qualification scheme for vector control pesticides are broadly supported by governments and other stakeholders. Consequently, there are opportunities to strengthen their implementation further.

Regional instruments and mechanisms

Regional instruments and mechanisms can be an effective way to optimize limited national resources for pesticide regulation and management. Since countries in the same region tend to have similar issues that need to be tackled (as well as cultural, ecological and economic commonalities), mutual understanding and collaboration may be easier than in larger geographical country groupings.

Regional collaboration on pesticide registration is one way to optimize the use of limited resources. Such collaboration has been successfully implemented in West Africa and the European Union (EU). In other regions (e.g., East Africa, the Pacific, the Caribbean, the Andean region) countries collaborate on specific aspects of pesticide registration or have broader regional programmes. While fully regional pesticide registration systems exist only in West Africa and the EU, in most other parts of the world governments continue to prefer national evaluations and decision-making, thereby duplicating much work (Chapter 3.3).

It is not entirely clear what makes certain regional cooperation schemes very effective, while others have remained a good intention but with only limited results to be shown. However, it appears that initiatives which originated from the bottom up – that is, from countries themselves rather than top-down (e.g., from a regional economic organization) – have been most effective.

Various recent initiatives to strengthen or establish regional collaboration on pesticide management provide an opportunity to further develop this approach.

Highly Hazardous Pesticides initiatives

Initiatives to phase out or strictly regulate the use of Highly Hazardous Pesticides (HHPs) are potentially an effective way to reduce pesticide risks. They require relatively limited efforts which can have high returns. Considerable attention has been given during the last 15 years to reducing the risks posed by HHPs, but targeted activities by countries to identify these pesticides and mitigate their risks have so far been limited (Chapter 3.2.3).

A major obstacle to reducing the risks of HHPs appears to be lack of national capacity to identify cost-effective lower-risk alternatives. Moreover, for risk reduction measures to be effective they often need to be far-reaching (e.g., prohibition or severe restriction of the pesticide's use), which requires high-level national political support.

The Action Plan on HHPs, currently under discussion through the Strategic Approach to International Chemicals Management (SAICM), presents an important opportunity to create a sense of urgency and mobilize resources to reduce the risks posed by HHPs or regulate these chemicals much more strictly.

National legislation

Comprehensive pesticide legislation provides the legal basis for many risk reduction measures, in the case of both governments and the private sector. The fact that most countries have such legislation in place means the need to regulate pesticide distribution and use has been recognized. Therefore, a good opportunity exists to further strengthen pesticide legislation. While the number of countries with pesticide legislation has not increased much in the last decades, the quality and comprehensiveness of this legislation is generally considered to have improved. Gaps in legislation exist in many countries, however, related in particular to public health and other non-agricultural pesticides (Chapter 3.4.1).

Effective implementation of pesticide legislation faces a bottleneck in many countries, especially those with limited human and financial resources. This concerns not only establishing and

maintaining the infrastructure and instruments needed to implement legislation, but also ensuring enforcement. One of the main reasons for relative lack of implementation of legislation is that pesticide (or chemicals) management competes with many other important national priority issues, especially in low and middle income countries.

Pest and pesticide management policies

National pest and pesticide management policies set out the desired direction in which pest and pesticide management should develop in the future. They define clear goals and targets that will need to be achieved over a given period of time, including pesticide risk reduction. If broadly endorsed and supported, such policies are potentially useful tools to guide activities by all relevant stakeholders in the same direction and mobilize resources to do so.

So far, the adoption of stand-alone national pest and/or pesticide management policies has been rare and has been limited mainly to Europe and North America (Chapter 3.4.5). Given the limited experience with such policies, it is difficult to know whether they would effectively contribute to reducing the risks posed by pesticides in different economic and cultural situations.

Control and enforcement

Even if pesticide legislation is in place, effective control and enforcement are required to avoid legislation being merely a paper tiger. Inadequate enforcement is a problem in many countries, irrespective of their economic development, but this problem is especially critical in low and middle income countries (Chapter 3.4.3). Many stakeholders consider lack of effective control and enforcement to be one of the main impediments to sound pesticide management. For instance, trade in illegal, substandard or counterfeit pesticides is increasing in many countries.

Given that human and financial resources are deficient in many low and middle income countries, strengthening enforcement capacity is a major challenge. Due to government budget cuts and the decentralization of enforcement activities, pesticide inspection and control may actually have worsened rather than improved in recent years.

Pesticide evaluation and registration

Pesticides are evaluated before they are authorized for use to determine whether they are effective, and whether they pose unacceptable risks under local conditions of use. Efficacy and risk assessments of pesticides are sophisticated when compared to assessments of many other chemicals on the market. Pesticide evaluations, which are becoming increasing data rich and complex, require considerable specialized technical capacity.

Pesticide registration authorities, especially in low and middle income countries, are often understaffed and have inadequate technical expertise (Chapter 3.4.2). Very few are currently in a position to conduct comprehensive, locally relevant and scientifically sound efficacy and risk evaluations of pesticides submitted for registration. However, high-quality evaluations of pesticides are being elaborated by regulators with greater resources (e.g., in the EU, the United States and Canada) and by international entities (e.g., the FAO/WHO Joint Meeting on Pesticide Residues, JMPR). Such evaluations are increasingly used by other regulators for local assessments of pesticides, thereby minimizing duplication of work.

Pesticide risk assessment methods have mainly been developed for temperate regions and for industrialized agriculture. Although some methods can be extrapolated to (sub-)tropical and hot, arid (agro-)ecosystems, clear gaps exist in the understanding of pesticide risks in these regions. Furthermore, certain pesticide risks are still insufficiently addressed, such as combination toxicity, endocrine disruption and ecosystem effects (Chapter 4).

Post-registration monitoring

Post-registration monitoring of pesticides in food, drinking water and the environment (as well as for environmental and human health effects) is an essential tool to identify unexpected direct and indirect effects of pesticides.

Monitoring of possible human health effects is mainly limited to pesticide residues in food, although a few (mainly high income) regions/countries have established regular surveillance programmes (Chapter 4.4.6). There are poison centres in less than half the world's countries. Pesticide poisoning cases are not monitored in most low and lower-middle income regions (Chapter 4.4.2).

In a few countries pesticide concentrations in the environment are monitored on a regular basis, particularly in surface water and groundwater (Chapter 4.3.2). Monitoring of pesticide effects on non-target organisms, on the other hand, is rare even in high income countries (Chapters 4.3.4 and 4.3.5).

Consequently, in most countries post-registration monitoring of environmental and human health effects is not an integral part of the pesticide registration process. Potential risks are generally assessed before a pesticide is authorized, but few data are gathered about a pesticide's fate and effects after it has been authorized for use. Information on the effects of pesticide use under real-life conditions is primarily compiled through ad hoc research projects. While this may seem the most-cost-effective approach to identify risks of pesticide use, unanticipated direct and indirect effects of pesticides on the environment and human health continue to become known.

To further strengthen post-registration monitoring of pesticide use and effects, it should be possible to take advantage of existing well-developed methodologies, especially for pesticide residues in the environment and food.

6.2.2 Market-based measures

Subsidies and taxes

Subsidies and taxes imposed on pesticides are potentially strong instruments to influence the type of pest management measures adopted by producers. Direct subsidies on conventional pesticides are currently not very common in countries, whatever their income level. Neither are subsidies that promote the use of low-risk pesticides (e.g., bioprotectants), although use of these pesticides appears to be growing (Chapter 2.7.11). Indirect subsidies, through exemptions from value added tax (VAT) or other general taxes, are still rather common, although they are increasingly being abandoned in high income countries. There is little quantitative information about the effects of providing subsidies, or abandoning them, on the types and quantities of pesticides used.

Taxes on pesticides, with the aim of discouraging pesticide use or moving towards low-risk products, have been implemented in only a few countries. For pesticide taxes to be effective, the tax rates should reflect damage/risks to the environment and human health (a "banded tax system"), as this not only encourages more conservative use of pesticides but also provides an incentive to use less harmful products (Chapter 2.7.11). However, experience so far shows that high rates of taxation are needed to reduce pesticide use. Furthermore, including a mechanism to redistribute tax revenues (as a form of compensation for farmers, or for earmarking further environmental uses and/or rebating it to the affected population and/or the sector) has been found to increase public acceptance.

The few low and middle income countries that have established pesticide taxes have done so at relatively low rates when compared to high income countries. It is unlikely that these taxes will affect pesticide use. However, redistributing revenues from relatively low taxes towards training, advisory services, and research and development (R&D) on low-risk alternatives may increase the effectiveness of such fiscal measures.

Private standards

Private standards, or voluntary sustainability standards, for agricultural commodities often include the application of integrated pest managment (IPM) and the prohibition of highly toxic pesticides among their requirements (Chapter 2.7.12). It can be expected that implementing private standards that include pest and pesticide management requirements will reduce the overall quantity of pesticides applied and/or their risks. However, no systematic

Table 6.2-2 Economic measures used to reduce the environmental and health risks of pesticides: strengths, limitations, opportunities and challenges.

Measure	Strengths	Limitations	Opportunities	Challenges
Pesticide subsidies (both direct and indirect)	Can promote the use of low-risk pesticides	Can result in excessive general pesticide use (including high-risk products)	Countries are increasingly abandoning general direct and indirect subsidies for pesticides Some countries have introduced targeted subsidies on low-risk or biological pesticides	Ensure that subsidies promote the use of low-risk pesticides
Pesticide taxes	Differentiated taxes can shift the behaviours of producers and consumers towards use of low-risk pesticides Even relatively low taxes can generate revenues for sound pest and pesticide management	Demand for pesticides is relatively inelastic, and high taxes tend to be needed in order to reduce pesticide use Taxes have been found ineffective if used in isolation from other policy measures	Lessons can be learned from countries which have applied different forms of pesticide taxes	Address concerns about producer competitiveness on the global market Ensure that tax revenues are used to support producers
Private standards	Standards that require organic production or IPM may reduce the environmental and human health effects of pest management A direct link is created between sound pest and pesticide management and the opportunity to sell produce	The number and strictness of private standards may disfavour small-scale farmers with limited technical support	Consumers and retailers, especially in middle and high income countries, are increasingly purchasing food produced with no or few pesticides	Certain (elements of) private standards may not be based on sound science Capacity needs to be built so that smallholder farmers can meet the standards Extra investments required to be made by producers should be covered through a bonus on the price paid for commodities produced according to the standard
Internalization of indirect health and environmental costs	Creates a level playing field for pesticides with different environmental and human health risks	Costs and benefits are not the only drivers of farmer behaviours regarding pesticide use	Lessons learned from targeted and differentiated pesticide subsidies and taxes, while still limited, might be applied elsewhere	Difficulty of calculating the value of externalities Existing estimates are outdated

assessments of the impact of such standards on pesticide use or risks have so far been conducted.

Overall coverage of voluntary sustainability standards is still limited (about 1 per cent of global cropland). However, in the case of certain crops (e.g., cocoa, coffee, tea, bananas) they are applied on 10-30 per cent of total acreage. For some of these crops these standards can achieve considerable reductions of pesticide risks. In most low and middle income countries private standards are used mainly for export crops, and much less for production destined to be sold at local markets.

It will be challenging to expand the amount of cropland under voluntary standards while at the same time building capacity for smallholder farmers, making it possible for them to meet requirements and not be left out of export and local markets.

Internalization of externalities

The use of pesticides may cause many externalities, that is, indirect effects whose costs are not borne by the pesticide user but by society as a whole (Chapter 5.2). Even conservative estimates of such externalities amount to billions of dollars globally. Externalities also lead to an uneven playing field for pest management practices with different degrees of environmental and human health effects (e.g., those of high-risk conventional pesticides versus low-risk biopesticides) since indirect environmental and human health costs are passed on to society.

Governments typically correct for this type of market failure by imposing taxes, providing subsidies or strengthening regulations. However, as discussed above, neither subsidies on low-risk pest management options nor taxes on high-risk pesticides are much used globally. Externalities therefore continue to be passed on to society as a whole. Stricter regulations on pesticides are increasingly being imposed, but so far mainly in high and middle income economies.

Overall, economic measures are potentially powerful instruments to influence pesticide use and direct behaviours towards lower risk pest management approaches. However, the use of economic instruments for this purpose has been limited up to now.

6.2.3 Training and awareness building

Awareness about the risks of pesticides to the environment and human health, as well as ways such risks can be minimized, is a prerequisite (although it is not in itself sufficient) for changing policy and behaviours towards more sustainable pest and pesticide management. Awareness of and concerns about such risks appear to have increased considerably on the part of the public, pesticide users and policymakers in the last 10 to 15 years (Chapters 2.7.14 and 2.7.18). At the same time, such awareness has not led to fundamental changes in the quantities of pesticides used and the ways they are used. A challenge in the near future will be to ensure that awareness building effectively translates into changing national policies, as well as the behaviours of pesticide users and consumers.

There is currently more information available on sound pest and pesticide management than ever before, for example through the internet and smartphone apps. However, it has also become increasingly difficult for farmers, spray applicators, other pesticide users, input dealers and extension staff to identify relevant and reliable information among the large volume of data available. Moreover, independent public extension and advisory services have seen their budgets and staff cut. Thus pesticide users have fewer options for obtaining independent information, especially in low income countries where they often do not have the means to turn to private advisory services.

Training on good pesticide application practices has been shown to increase the knowledge and awareness of farmers and other pesticide users about the risks posed by pesticides. However, in many cases better knowledge and awareness has not led to long-term changes in behaviours which could reduce pesticide risks (Chapter 2.7.20). Training and awareness building, when carried out in isolation, is unlikely to be sufficiently effective in reducing these risks (Chapter 2.7.21).

Table 6.2-3 Training and awareness building used to reduce the environmental and health risks of pesticides: strengths, limitations, opportunities and challenges.

Measure	Strengths	Limitations	Opportunities	Challenges
Awareness building about pesticide risks for policymakers, pesticide users and the public	Prerequisite for changing policy and behaviours Awareness has increased in the last 15 years	Greater awareness has not (yet) led to fundamental changes in pest and pesticide management	Social media allow better targeting of information	Ensure that awareness effectively translates into changes in policy and behaviours
Training on judicious pesticide use	Increases knowledge and awareness about good practices and risks	Often does not improve the behaviours of pesticide users	Integration into broader policy measures to reduce risks posed by pesticides	Avoid judicious pesticide training being provided in isolation from that in IPM
Training in biointensive IPM	Leads to changes in farmers' behaviours which reduce risks of and reliance on pesticides	Relatively expensive, so that upscaling has been constrained	Integration into broader policy measures to reduce risks posed by pesticides Combine with modern technologies to provide information and maintain contact	Mobilize long-term funding Long-term political commitment

More intensive and participatory capacity building approaches, such as farmer field schools, have been shown to be effective in achieving long-lasting changes in behaviours and a transformation towards biointensive integrated pest management (Chapter 2.7.5). However, due to the relatively high cost and a lack of national commitment, mainstreaming this type of capacity building into extension and advisory services, and upscaling it to reach more farmers/pesticide users, has been slow and is subject to considerable constraints. This may contribute to unwarranted impacts including on the environment, MRLs and trade etc.

Training and awareness building are important elements of a strategy aimed at reducing reliance on pesticides and minimizing their environmental and human health risks. However, they need to be combined with other policy measures to be successful in the long run.

Given the gender differences in health risks posed by pesticides, but also in the different roles that men and women may have in agricultural production and pesticide use, care should be taken that information dissemination, training and awareness building are gender responsive. This to ensure that appropriate crop protection technologies and practices are adopted, farmers' exposure to pesticides is reduced, and environmental quality is improved.

6.2.4 Engineering controls and technology

Engineering controls

Engineering controls, such as closed and ventilated tractor cabins, closed circuit tank filling and rinsing systems, and drift-reducing nozzles are being used to reduce the environmental and human health risks posed by pesticides (Chapter 2.6).

Many of these technologies have been developed for tractor mounted or pulled spray equipment and have become mandatory in certain high income countries. However, most pesticide application in low and lower-middle income countries is carried out using manual or motorized backpack sprayers, for which such technologies are less or are not applicable. Advanced sprayer technology that reduces environmental and occupational exposure therefore has only limited relevance currently in low and lower-middle income countries.

Precision agriculture

A variety of technologies are being used for precision spraying, i.e., varying the amount of pesticide applied in a field according to the site-specific characteristics of the crop. These technologies used include variable rate application and intermittent (spot) spraying, linked to (remote) sensors that identify canopy density or other characteristics such as the presence of crop diseases (Chapter 2.6). The objective is to apply pesticides only where they are needed and in the right amount.

Other techniques for precision spraying include robotic spray platforms and drones, which generally reduce occupational exposure and may limit the quantity of pesticide applied. However, using drones may increase pesticide drift.

Currently, precision spraying technologies require considerable investments and are primarily used in the case of high-value specialty crops. As these technologies are being used increasingly, however, they may become more affordable, including in low and middle income countries, especially if actively promoted through economic incentives.

Personal protective equipment

The purpose of using personal protective equipment (PPE) is to reduce occupational exposure to pesticides. Since even good quality PPE does not provide complete protection against

Table 6.2-4 Engineering controls and technologies that can be used to reduce the environmental and health risks of pesticides: strengths, limitations, opportunities and challenges.

Measure	Strengths	Limitations	Opportunities	Challenges
Engineering control	Risk reduction is integrated into the application equipment Direct reduction of environmental and human exposure	Increases costs of equipment	Greater use of these technologies may reduce costs and increase availability	Develop and market effective engineering controls appropriate for application methods used in low and middle income countries
Precision spraying	Optimizes use of pesticides May directly reduce environmental and human exposure	Significantly increases costs of equipment and application May be dependent on external technical inputs (e.g. remote sensing data)	Greater use of these technologies may reduce costs and increase availability	Provide affordable access to these technologies by smallholder farmers
Personal protective equipment (PPE)	Direct reduction of environmental and human exposure Relatively cheap and easy to use	Not worn because uncomfortable, expensive, unavailable May lead to unsafe practices if seen as offering complete protection	Voluntary standards and certification systems require use of appropriate PPE	Creating a culture in which PPE use is standard Developing affordable, effective and comfortable PPE, particularly for use in low and middle income countries

pesticide exposure, it is generally considered a last line of defence after other risk reduction measures have been applied. Furthermore, the effectiveness of PPE under actual working conditions may be overestimated (Chapter 4.4.5).

In many parts of the world farmers and farm workers do not have access to appropriate PPE, either because it is not available on the local market or because it is too expensive. Even if PPE is available, working conditions (e.g., high temperature and humidity) may be such that pesticide users will not use certain items because they are uncomfortable or even dangerous (e.g., they may cause heat stress).

Relying on the use of PPE as the single or main way to reduce occupational risks, especially those of more hazardous pesticides, is therefore a questionable approach. Regulatory decisions to authorize a pesticide under the assumption (or even if there is a requirement) that PPE will be worn may be severely flawed, especially in the case of subsistence farmers in hot climatic conditions.

However, appropriate PPE will likely contribute to reducing pesticide exposure (if used as a last line of defence) in combination with other risk reduction measures. This is especially true in situations where a culture of voluntary PPE use already exists, where compliance can be ensured, and where pesticide users obtain clear (economic) benefits from using PPE (e.g., on farms producing under voluntary sustainability standards, or where PPE use is directly linked to professional certification of the farmer or pesticide applicator).

6.2.5 The effectiveness of current risk reduction

One way to look at the effectiveness of pesticide risk reduction measures is through the hierarchy of controls. This is a generic, but well-established method of ranking risk reduction measures according to their reliability and efficacy. The principles embodied in the hierarchy of controls (Figure 6.2-1) date back to the 1940s and have been applied since that time as a standard for risk reduction in occupational health and safety, for example in the United States (Office of Technology and Assessment 1985; National Institute for Occupational Safety and Health 2015) and the EU (European Agency for Safety and Health at Work 2021) as well as by the International Labour Organization (ILO) (Alli 2008). The risk reduction methods at the top of the triangle are potentially more effective and protective than those at the bottom.

Figure 6.2-1 The hierarchy of controls as applied for occupational risks by the United States National Institute for Occupational Safety and Health (NIOSH). NIOSH (2015).

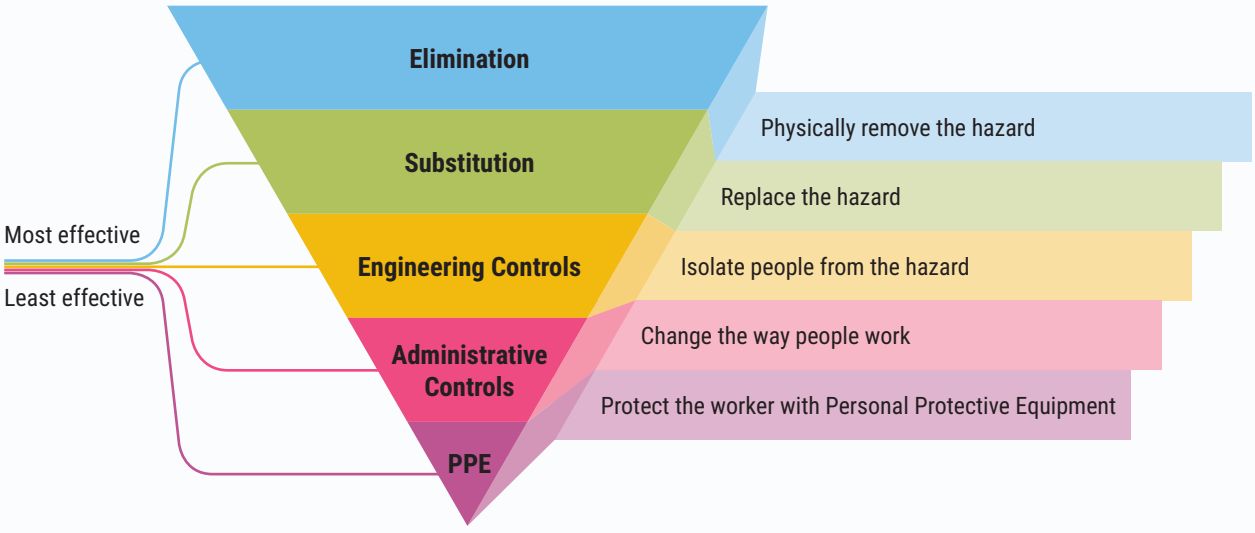


Table 6.2-5 Examples of pesticide risk mitigation measures for environmental and human health risks at different levels in the hierarchy of controls.

Stage in the hierarchy of controls	Human health risks			Environmental risks
	Occupational	Resident/ bystander	Dietary	
Elimination	Do not use pesticides (e.g. use resistant crop varieties, grow a healthy crop, conduct mechanic weed control, promote natural biological control) or Do not register the use of a high-risk pesticide (for a specific use) (e.g. ban, non-approval, cancellation, non-renewal)			
Substitution	Replace pesticide with high human health risk with a lower risk one			Replace pesticide with high environmental risk with a lower risk one
Engineering controls	Drift reducing nozzles, equipment calibration, deflectors		Equipment calibration	Drift reducing nozzles, equipment calibration, container rinse system, precision spray technology
	Closed tractor cabin, closed pesticide transfer system, container rinse system, hand wash water supply			
Administrative controls	Training, certification, best practices, application rate and frequency			
	Worker rotation, re-entry intervals	Notification of neighbouring properties	Pre-harvest intervals	Unsprayed buffers, vegetated filter strips, timing of applications during the day or season, notification of beekeepers
Personal protective equipment (PPE)	Require or recommend PPE as part of registration (e.g. on the label)	n.a.	n.a.	n.a.
Rehabilitation	n.a.	n.a.	n.a.	Multifunctional field margins, ecological recovery areas

Based on Coffman *et al*. (2009); Weinberg, Bunin and Das (2009); NIOSH (2015); Alix *et al*. (2017); Lukey and Paras (2017)
n.a. = not applicable

Similar hierarchies of controls are used to determine waste management options according to what is best for the environment (the "waste hierarchy") (e.g., Department for Environment, Food and Rural Affairs 2011) or to identify best practices to reduce environmental impacts (the "mitigation hierarchy") (Lukey and Paras 2017). The hierarchy of controls is further developed for the environmental and human health effects of pesticides in Table 6.2-5. A distinction is made between occupational, resident/bystander, dietary and environmental risks. While the general principles of the hierarchy of controls apply in the case of all of these risks, the risk reduction measures taken may differ.

Based on the assessments in other parts of this report, the effectiveness of various pesticide risk reduction measures is summarized in Table 6.2-6,. The table focuses on the use of these measures in low and lower-middle income countries. Country income has served as a proxy for available human and financial resources. For each measure listed in the table, an appraisal was carried out of whether it had become more or less effective in reducing the environmental and human health risks of pesticide use during the last two decades. Evaluations of effectiveness and trends over time are necessarily approximate, as they are highly dependent on the situation in individual countries.

Table 6.2-6 Effectiveness of pesticide risk reduction measures during the last two decades and overall trend, with a focus on low and lower-middle income countries.

Measure	Effectiveness in reducing environmental and human health and risks			Overall trend
	High	Moderate	Limited	
Regulatory measures				
National pesticide legislation	H	M, L		↑
Pest and pesticide management policy		H	M, L	↑
Control and enforcement		H	M, L	↓
International conventions		M, L	H	↔
Voluntary international instruments		M, L	H	↔
Regional instruments and mechanisms		H, M, L		↑
HHP risk reduction		H, M, L		↔
Pesticide registration	H	M, L		↑
Post-registration monitoring		H	M, L	↔
Economic measures				
Pesticide subsidies		Not evaluated		
Pesticide taxes		H	M, L	↑
Private standards		Not evaluated		
Training and awareness building				
Awareness about pesticide risks		H, M	L	↔
Training and extension		H	M, L	↓
Engineering controls				
Spray technology		H, M	L	↑
PPE		H	M, L	↔

Trend: ↑ = improved; ↓ = declined; ↔ = unchanged
H= high income countries, M= middle income countries, L= low income countries

Many current pesticide risk reduction approaches and measures have had only limited effectiveness, particularly in low and lower-middle income countries. The most successful ones mainly been successful in high income and, to a lesser extent, middle income countries.

The availability of resources (financial, technical, human, information) is a critical variable determining the success of specific risk reduction options. Risk reduction measures that have been effective in middle or high income countries will not necessarily be effective in countries, situations or communities where resources are inadequate.

When comparing the effectiveness of current pesticide risk reduction measures in Table 6.2-5 with those shown in the hierarchy of controls in Table 6.2-5, some commonalities and clear divergences can be distinguished.

Training, awareness building and use of PPE are relatively low in the hierarchy of controls. They have not been found to be very effective in current pesticide risk reduction, particularly in lower income countries. Some regulatory measures which are high in the hierarchy of controls have been found to be moderately or highly effective in pesticide risk reduction, but this has often been limited to higher income countries. The effectiveness of regulatory measures has had mixed outcomes, with many being only moderately effective. Engineering controls, again only moderately effective in the hierarchy of controls, were considered effective in reducing pesticide

risks, but primarily in high income countries where such controls are available and cost-effective.

Clearly no silver bullet exists that will reduce pesticide risks in all circumstances: one size does not fit all.

6.3 Strengthening pesticide management: options for actions

In this section actions are proposed that could strengthen existing pesticide risk reduction and risk management measures.

As described above (and in more detail in Chapters 2, 3, 4 and 5), numerous measures are being taken in countries to promote the sound management of pesticides and reduce their risks to the environment and human health. Nevertheless, the degree to which such measures have been implemented or are successful varies greatly among countries and regions. This diversity of choices and levels of progress creates opportunities to draw lessons from countries that have implemented measures with more or less success.

In the following chapter options are proposed to further progressively strengthen measures which, based on the assessments in this report, appear likely to contribute effectively to pesticide risk reduction. Depending on the degree of current implementation, and local regulatory, economic and cultural situations, countries may choose to develop and adopt specific measures. However, experience has shown that it is essential to develop a coherent and comprehensive set of measures based on a concrete policy vision if these measures are to be effective.

Single and/or fragmented measures are rarely adequate and may even be contradictory if implemented by different government or private entities. The elaboration or updating of a comprehensive national pest or pesticide management policy may be a good starting point for setting risk reduction goals and developing roadmaps or action plans leading to them.

It is recognized that the various options introduced will be more or less applicable to a specific situation prevailing in a given country or region.

6.3.1 Strengthening governance of the production, trade and use of pesticides

Elaborate and implement national sustainable pest and pesticide management policies

By adopting a national pest and pesticide management policy, sound management of pesticides can be placed and/or maintained on the national government's policy agenda. Such a policy should define clear objectives and targets for pest and pesticide management within a foreseeable time period, and be aimed at promoting sustainable pest management and reducing pesticide risks. This can be achieved through a stand-alone pesticide management policy which includes pesticides, but also by using other approaches to pest management or by incorporating elements of such policies in national legislation, regulations or policies on related topics (see Chapter 3.4.5). All stakeholders need to be involved in elaborating, implementing and monitoring this policy in order to secure wide support for its objectives and measures. Such a policy needs to be put into practice through effective legislation and their enforcement.

Develop and update national pesticide legislation to include all elements of the pesticide life cycle

Most countries have adopted dedicated pesticide legislation. In principle, this legislation should

cover all types of pesticides and all aspects of the pesticide life cycle in order to further strengthen the legal basis for sound pesticide management. This could involve ensuring the inclusion of essential elements such as administration, registration, import and export, licensing, packaging and labelling use, advertising, storage, transport and disposal, information collection, monitoring and incident reporting, inspection offences and penalties.

It is crucial that pesticide legislation be harmonized with national environmental, agricultural, health and economic regulations and policies aimed at pest and chemicals management, and that policy incoherence is avoided. To accomplish this, a national regulatory framework for control of pesticides may encompass a much broader set of legislation (e.g., legislation on environmental protection, public and occupational health, food safety, water, wildlife, plant protection and general chemicals management) than that which only directly addresses pesticides (see Chapter 3.4.1).

Enhance enforcement capacity to bolster effective implementation of national legislation

In many countries there is an urgent need to increase the effectiveness of the inspection and control of pesticide-related activities during manufacturing, importation, distribution, sales, disposal, and use. Ways to increase effectiveness include strengthening inspection services, tightening collaboration with other law enforcement entities, and establishing laboratory capacity for pesticide quality control. In addition, a culture of compliance by all stakeholders should be cultivated, for example through information exchange, compliance incentives, and appropriate forms of self-certification. Increased effectiveness of inspections and enforcement can be achieved by developing guidance and knowledge products with respect to what is required for compliance and increasing awareness in the regulated community. There is also a need to increase government capacity and enable regulatory entities to effectively carry out inspections, and to prosecute and adjudicate violations of pesticide legislation (see Chapter 3.4.5).

Establish or strengthen regional collaboration in pesticide management and, where relevant, harmonize pesticide legislation regionally

Regional collaboration can be an effective way to address transboundary or regional issues related to pesticides, such as cross-border trade in both legal and illegal pesticides, efficacy and residue testing of pesticides, pesticide residues in regionally traded commodities, pesticide pollution of transboundary watersheds, and information sharing on the observed efficacy and effects of pesticides and alternative pest management options. The regionalization of activities may include collaboration on the evaluation, authorization, inspection and control of pesticides and may occur through simple information exchange mechanisms or through more complex common decision-making. Regional collaboration can enhance the possibilities to control cross-border (illegal) trade in pesticides.

Regional harmonization of pesticide legislation goes a step further than collaboration. While more complex to establish and implement, regional harmonization can be beneficial for governments as well as the private sector, particularly in areas such as pesticide registration, pesticide quality standards, maximum residue levels (MRLs), and the control of illegal trade in pesticides. Regions may choose, for example, to harmonize their pesticide testing protocols, set up regional pesticide management action plans or introduce regional pesticide registration. (see Chapter 3.3).

6.3.2 Strengthening monitoring of pesticide use and effects

Collect statistics on the manufacturing, importation and sales of pesticides

Monitoring pesticides' use and effects is critical to inform decision-making and policy development. Countries should consider establishing or strengthening, as a minimum, the collection of national statistics on manufacturing, importation and sales of pesticides. One key area to achieve this would be to include information collection in pesticide legislation. Specifically designating powers and responsibilities, including the ability to impose reporting requirements on manufacturers,

importers, distributors and sellers of pesticide (see Chapter 3.4.1). Where possible, data on the use and disposal of pesticides should be compiled. After a pesticide is marketed, data collection and assessment may take place through regular monitoring, specific scientific studies, or feedback about incidents (see Chapter 4.2).

Establish pesticide residue monitoring systems and poison centres

Countries may consider establishing, where they do not exist, national pesticide residue monitoring programmes for food, feed and drinking water. Although more complex, countries should also consider establishing post-registration monitoring of pesticides and their transformation products in the environment, especially in areas where there is high use intensity. In addition, national or regional poison centres that operate effectively are essential to better understand the health effects of pesticide use (see Chapter 4.4.2).

Ensure feedback into policy- and decision-making

An analysis and reporting system needs to be put into place to ensure that the results obtained by monitoring and surveillance will inform policy- and decision-making on the authorization and use of pesticides. For example, post-registration monitoring and studies that complement prospective risk assessments are therefore important tools, especially given that data on environmental settings and human populations can only become available after the market authorization of a compound. Pesticide registrations should be subject to a periodic review process for re-authorization if consecutive evaluations show that use of a pesticide under local conditions results in unacceptable risks based on the outcomes of post-registration monitoring and studies (see Chapter 4.2).

6.3.3 Strengthening pesticide evaluation

Develop more integrated approaches to pesticide evaluation

Pesticides are currently evaluated on their individual merits and risks. If a pesticide is found to be effective and does not pose unacceptable risks, it will generally be authorized for use. However, its risks and benefits compared to those of other pesticides or pest management approaches are not assessed, nor is its contribution to the long-term sustainability of pest or vector management.

This approach to pesticide registration can be transformed to promote broader sustainable pest management solutions. Alternative pest management options should be evaluated as part of the decision-making process, along with their environmental and economic impacts. Evidence-based and interdisciplinary decision-making should drive pest management choices, while explicitly taking into account uncertainties and knowledge gaps in a precautionary manner (see Chapter 4.4).

Establish robust pesticide risk assessment methods for low and middle income countries

Since the human and financial resources available for pesticide evaluation are limited in many low- and middle-income countries, the extensive pesticide risk assessment approaches developed by some high-income countries generally cannot be applied there. There is an urgent need to develop robust and pragmatic environmental and human health risk assessment methods applicable to conditions of use in major world regions, with a focus on tropical, sub-tropical and semi-arid regions. This need may be addressed by extrapolating existing methods for use in neglected ecosystems and regions (see Chapter 4.4). Particular attention should be given to vulnerable groups and specific risks related to gender.

Optimize resources for pesticide evaluation

To optimize the use of limited resources for pesticide evaluation and minimize duplication of work, further internationalization of efficacy and risk assessments is needed: evaluate hazards globally – assess efficacy and risks regionally – authorize pesticides nationally/regionally.

Pesticide hazards are typically independent of use conditions. They can therefore be evaluated globally. Pesticide efficacy and risks are influenced

by conditions of use, as well as by environmental and agronomic circumstances, but they can be evaluated on a regional or (agro-)ecosystem basis rather than repeatedly at the national level. Registration decisions will also be influenced by the economic and social circumstances in which a pesticide is being used. Although based on international or regional efficacy, hazard and risk assessments, these decisions are therefore best taken at the national or regional levels (see Chapter 4).

Fill gaps for pesticide risk assessment

Environmental and human health risk assessment has greatly improved in the past decades. Nevertheless, evaluating certain pesticide risks requires further scientific inputs, including the environmental and human health risks of pesticide mixtures and of endocrine disruptors, pesticides' health effects during child development, and their environmental risks in (sub-)tropical and (semi-)arid regions. Other principal directions for improving pesticide environmental and human health risk assessments are proposed in Chapter 4.3.7 and 4.4.8.

Increase transparency in decision-making

In a number of recent cases, confusion has occurred as a result of apparent inconsistencies in evaluations conducted by different regulatory and scientific bodies. This has partly been due to use of different data sets and lack of access by non-regulators to data considered confidential.

To maintain public trust in the independence and comprehensiveness of regulatory risk assessments, there is a need to redefine the mechanisms that ensure the confidentiality of business information, but also to ensure public access to and third-party verification of environmental and human health information. These may be embedded in the legislation through, for example, registration and Information collection, monitoring and incident reporting, (see Chapter 3.4.1)

6.3.4 Innovate pest management through targeted research and development (R&D)

Promote multi-stakeholder innovation

Through the identification, development and implementation of innovative, context-appropriate and cost-effective technologies for pest and vector management requires collaboration between farmers or other pesticide users, public research bodies and the private sector. Such collaboration should not be limited to new pest control products. It can be broadened to innovative pest management approaches, including associated new skills that need to be adopted by farmers and other pesticide users. This can lead to more successful reductions in environmental and human health risks as a result of reduced application of pesticides, expanded use of reduced risk products, improved pesticide application practices, or better use of precautionary measures.

Develop low-risk pest control technologies

Public and private sector actors should prioritize and facilitate the development of low-risk pesticides and bioprotectants. This could include fast-track regulatory procedures, but also the establishment of local production facilities for beneficial macroorganisms and biopesticides, as well as services for the placement and monitoring of non-chemical pest control systems (e.g., traps, robotics, predator release) (see Chapter 2.7).

Innovative pest management technologies, such as RNAi for gene silencing in pests or CRISPR for breeding disease and pest resistant crops, may yield promising pest management approaches. They should be evaluated for efficacy and safety, as well as for access by smallholder farmers. More efficient pesticide application technologies, which reduce pesticide use and risks, also require further investment in R&D.

Assess costs and benefits

There is an urgent need to conduct comprehensive assessments of all economics variables associated with the use of pesticides, including direct and indirect costs and benefits, with the aim

of designing pest management strategies that are cost-effective at both the private and societal levels. Studies should also take into account other pest management options or the costs of externalities. For example, assessments may be made on the economic benefits of pesticide use at either individual pesticide level or a larger geographical scale as well as more comparisons between the overall costs of pesticide use with their estimated benefits (see Chapters 5.3.1 and 5.3.2).

Regularly review existing knowledge

To make better use of the large body of research that continues to be generated on the efficacy and risks of pesticides, there is a need for regular systematic reviews of new knowledge and insights. Sound scientific approaches should be applied in order to conduct such reviews, ensure the quality of the data involved, and ensure the inclusion of these reviews in the regulatory (re-)evaluations of pesticides. For example, more systematic reviews on pesticide risks and their mitigation options can be carried out (see Chapter 4.3.7 and 4.4.8).

6.3.5 Inform and educate for change

Ensure independent information provision to farmers and other pesticide users

To allow farmers and other pesticide users to make informed and objective decisions about pest management options, public agricultural extension systems and other independent advisory services need to be strengthened further. Integrated pest management (IPM) and integrated vector management (IVM) should be promoted through the training of agronomists, extension agents, input distributors and farmers.

Participatory and experience-based learning has been shown to be effective in ensuring the long-term adoption of sustainable behaviours and technologies, with specific attention given to building the capacities of youth and women. Better use can also be made of digital technology and social media to provide information and complement education and training, while ensuring the inclusion of marginalized groups.

Include sustainable pest management in educational curricula

Sustainable agriculture, agroecology, integrated pest management (IPM) and integrated vector management (IVM) could be included in the curricula and courses of agricultural schools, universities and other relevant training providers to a much greater extent. A highly participatory model which has been applied to build capacity in IPM management is farmer field schools (see Chapter 2.7.21).

Training in IPM, IVM and biocontrol should be considered a fundamental requirement for licensing of pest control operators.

Train health care and environmental professionals about pesticide risks

Health care professionals need to be prepared to diagnose and treat cases of pesticide poisoning, particularly in areas where there is high pesticide use. Furthermore, environmental professionals (e.g., those engaged in carrying out environmental inspection and monitoring, or staff at watershed authorities) should be trained on the identification and remediation of pesticides' environmental effects (see Chapters 4.3 and 4.4).

All environmental and human health incidents documented by professionals should be reported, so that they feed back into the pesticide decision-making process.

6.3.6 The need for policy coherence

Many strategies, programmes and projects have been developed and implemented during the last few decades to strengthen pest and pesticide management in all regions of the world. However, they have often addressed only a specific aspect of pest or pesticide management such as updating pesticide legislation, promoting integrated pest management (IPM) or biocontrol, strengthening pesticide registration, upgrading pesticide residue laboratories, disposing of obsolete pesticide stocks, training farmers in good agricultural practices, or transforming pesticide taxes or subsidies.

While such activities individually may have been quite successful in most parts of the world, they have not fundamentally changed how pests are managed or reduced the risks posed by pesticides. For example, pesticide use intensity is increasing, pesticide residues are ubiquitous in the environment, pesticide resistance continues to increase, occupational pesticide poisoning still occurs, surface water and groundwater are polluted by pesticides, and unanticipated environmental and health effects of pesticides are observed after the pesticides are authorized for use.

There is a need for integrated national and regional policies which address all aspects of pest management and aim to make it sustainable in the long term. Such policies, which have been rare so far (Chapter 3.4.5), need to combine regulatory and economic measures, ensure engagement to provide independent and relevant information, find effective ways to reach and train farmers and extension staff, develop innovative technologies accessible to stakeholders, and level the economic playing field for low-risk pesticides and pest management approaches. By their nature, these policies must be interdisciplinary, inter-ministerial and multi-stakeholder.

As shown by Lee, den Uyl and Runhaar (2019) (Chapter 3.4.5), pesticide risk reduction policies require an intelligent combination of measures which should be implemented in parallel or consecutively to achieve the goals that have been established. Integrated, more holistic approaches which reduce reliance on pesticide use are needed, including measures and incentives from the local farm level to the national (and even international) levels. Such approaches should set clear goals and targets, identify measures that contribute to achieving these targets, and include an assessment of measures that may be counterproductive.

References

Alix, A., Brown, C., Capri, E., Goerlitz G, Golla B, Knauer K *et al*. (eds.). (2017). *Mitigating the Risks of Plant Protection Products in the Environment (MAgPIE)*. Pensacola, FL and Brussels: Society for Environmental Toxicology and Chemistry (SETAC). https://www.setac.org/store/ViewProduct.aspx?id=9006489.

Alli, B.O. (2008) *Fundamental Principles of Cccupational Health and Safety. Second Edition*. Geneva: International Labour Office (ILO). https://www.ilo.org/global/publications/ilo-bookstore/order-online/books/WCMS_093550/lang--en/index.htm.

Coffman, C.W., Stone, J.F., Slocum, A.C., Landers, A.J., Schwab, C.V., Olsen, L.G. and Lee, S. (2009). Use of engineering controls and personal protective equipment by certified pesticide applicators. *Journal of Agricultural Safety and Health* 15(4), 311-326. https://doi.org/10.13031/2013.28886.

Department for Environment, Food and Rural Affairs (United Kingdom) (2011). *Guidance on Applying the Waste Hierarchy*. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/69403/pb13530-waste-hierarchy-guidance.pdf.

European Agency for Safety and Health at Work (2021). Directive 89/391/EEC - OSH "Framework Directive. https://osha.europa.eu/en/legislation/directives/the-osh-framework-directive/1. Accessed 17 January 2021.

Lee, R., den Uyl. R. and Runhaar, H. (2019). Assessment of policy instruments for pesticide use reduction in Europe; Learning from a systematic literature review. *Crop Protection* 126, 104929. https://doi.org/10.1016/j.cropro.2019.104929.

Lukey, P. and Paras, S. (2017). The environmental impact mitigation hierarchy as a hierarchy of risk. *Journal of Biodiversity, Bioprospecting and Development* 4(2), 1000166. https://doi.org/10.4172/2376-0214.1000166.

National Institute for Occupational Safety and Health (United States) (2015). The hierarchy of controls, 13 January. *United States Centers for Disease Control and Prevention*. https://www.cdc.gov/niosh/topics/hierarchy/. Accessed 17 January 2021.

Office of Technology and Investment (United States) (1985). *Preventing Illness and Injury in the Workplace*. United States Congress, Office of Technology Assessment. https://ota.fas.org/reports/8519.pdf.

United Nations Environment Programme (2013). *Costs of Inaction on the Sound Management of Chemicals*. Geneva. https://www.unenvironment.org/resources/report/costs-inaction-initiative.

United Nations Environment Programme (2019). *Global Chemicals Outlook II – From Legacies to Innovative Solutions: Implementing the 2030 Agenda for Sustainable Development*. https://www.unenvironment.org/resources/report/global-chemicals-outlook-ii-legacies-innovative-solutions.

Weinberg, J.L., Bunin, L.J. and Das, R. (2009). Application of the industrial hygiene hierarchy of controls to prioritize and promote safer methods of pest control: A case study. *Public Health Reports* (issue 4 suppl. 1) 124, 53-62. https://doi.org/10.1177/00333549091244S107.

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them

Envisioning A Chemical-Safe World

Chapter 7 of 12

Status and trends of fertilizer use

Contents

1 Global drivers, actors and policies affecting pesticides and fertilizer use

2 Status and trends of pesticide use

3 The regulatory and policy environment for pesticide management

4 Environmental and health effects of pesticide use

5. The environmental, human health and economic impacts of pesticides

6 Current pesticide risk reduction and risk management

About ii

7 Status and trends of fertilizer use 1

7.1 Overview 1

7.2 Types of fertilizers and fertilizer management 2

7.2.1 The nutrients required by plants 2
7.2.2 Major fertilizer types 2
7.2.3 Cropping systems and fertilizer use 4
7.2.4 Enhanced efficiency fertilizers 4
7.2.5 Nutrient recycling 6
7.2.6 The 4Rs of nutrient management 8
7.2.7 Fertilizer storage 9
7.2.8 Hazard classification and fertilizer compatibility 10

7.3 Fertilizer demand, sales and use 12

7.3.1 Fertilizer production and supply (current and projected) 13
7.3.2 Fertilizer demand: N from inorganic fertilizer and manure, P and K from inorganic fertilizer 15
7.3.3 Fertilizer exports and imports 18

7.4 Fertilizer distribution and sales mechanisms 19

7.4.1 Distribution and marketing 19
7.4.2 Organization of the fertilizer industry 20
7.4.3 Transportation 20
7.4.4 Landlocked countries 20

7.4.5 Commercial sales and subsidies 20
7.4.6 Fertilizer quality and adulteration 20
7.4.7 Cross-border trade 21

7.5 Drivers of fertilizer use 21

7.5.1 Factors influencing fertilizer usage: examples 21
7.5.2 Reasons for decreased fertilizer use: examples 28

References 31

8 The regulatory and policy environment of fertilizer management and use

9 Environmental and health effects of fertilizer use

10 The impact of fertilizer use

11 Current fertilizer risk reduction and risk management

12 Transformative actions to minimize the adverse impacts of pesticide and fertilizers

List of Figures

Figure 7.2-1	Proportion of nutrients used by crop categories globally.	4
Figure 7.2-2	Simplified overview of nitrogen and phosphorus flows highlighting major anthropogenic sources, and the associated environmental concerns.	7
Figure 7.2-3	Percentage of biosolids produced nationally used in agriculture in developed countries.	8
Figure 7.2-4	Compatibility of various fertilizer materials.	10
Figure 7.3-1	Comparison of FAOSTAT and IFASTAT data on fertilizer use in agriculture for the period 2002- 2017.	12
Figure 7.3-2	Fertilizer production (Mt).	13
Figure 7.3-3	Estimated supply and fertilizer demand for nitrogen, phosphorus and potassium in 2019 (Mt).	14
Figure 7.3-4	Nitrogen (N), phosphorus (P2O5) and potassium (K2O) fertilizer use in agriculture (Mt), 2002-2017.	15
Figure 7.3-5	Amount of nitrogen (N) excreted in manure which was left in pastures and applied to soil (Mt) (a) between 1961 and 2017, globally, and (b) in different regions (averages for 2003-2017).	16
Figure 7.3-6	Estimated net anthropogenic N inputs in the world's main river catchments (1 km2=100 ha).	17
Figure 7.3-7	Fertilizer exports minus imports, for nitrogen (N), phosphorus (P2O5) and potassium (K2O), in 2017 (Mt).	18
Figure 7.4-1	Simplified diagram of the fertilizer supply chain. The orange and green lines represent inorganic and organic fertilizers, respectively.	19

List of Tables

Table 7.1 Classification of inorganic fertilizers for transportation, storage and handling. 11

About

In December 2017, Resolution 4 of the 3rd Session of the United Nations Environment Assembly (UNEA 3) requested "the Executive Director to present a report on the environmental and health impacts of pesticides and fertilizers and ways of minimizing them, given the lack of data in that regard, in collaboration with the World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO) and other relevant organizations by the fifth session of the United Nations Environment Assembly". In response to this request, UNEP published a *Synthesis Report on the Environmental and Health Impacts of Pesticides and Fertilizers and Ways to Minimize Them*[1] in February 2022 (United Nations Environment Programme [UNEP] 2022).

The overall goal of the synthesis report is to provide the information base to enable other advocacy actions to be taken by stakeholders to minimize the adverse impacts of pesticides and fertilizers. Specific objectives of the synthesis report are to:

- Update understanding of current pesticide and fertilizer use practices;
- Present major environmental and health effects of pesticides and fertilizers, during their life cycle, and identify key knowledge gaps;
- Review current management practices, legislation and policies aimed at reducing risks in the context of the global chemicals, environmental and health agenda;
- Identify opportunities to minimize environmental and health impacts, including proven and innovative approaches.

This chapter on "The regulatory and policy environment for pesticide management" is the 7th in a series of 12 chapters that make up a comprehensive compilation of scientific information. The chapters were developed to both inform and further elaborate on the information provided in the synthesis report. Please note that the disclaimers and copyright from the synthesis report apply.

1 The Synthesis report is available at https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

Status and trends of fertilizer use

7

7.1 Overview

Soil is a major source of nutrients needed for plant growth. However, it may not provide adequate amounts of all the essential nutrients that plants require in correct proportions. Fertilizers can address nutrient deficiencies, maintain soil fertility, support crop growth, and maximize economic returns. [Chapter 7.2.1] They are loosely grouped as inorganic and organic, depending on their source and how they are produced. Nitrogen, phosphorus and potassium are the primary nutrients which fertilizers supply (Fairhurst 2012; Abdou *et al*. 2016; European Union [EU] 2019). [Chapter 7.2.2]

Enhanced efficiency fertilizers (EEFs) can play a significant role in sustainable agricultural production, but they may be expensive and some of the coating materials used are non-degradable (Naz and Sulaiman 2016). Moreover, EEFs may not perform better than conventional fertilizers (Li, T. *et al*. 2017). Research gaps exist for some of these fertilizers. [Chapter 7.2.4] Nutrient losses can be reduced by taking the state of the soil into account (Johnson and Bruuslsema 2014), targeting plant requirements, applying the right fertilizer, adopting practices that improve nutrient use efficiency (Johnson and Bruuslsema 2014; Nkebiwe *et al*. 2016; Wohab *et al*. 2017; Mutegi, Kiwia and Zingore 2019), and using good storage practices (Teenestra *et al*. 2015; Albadarin, Lewis and Walker 2017; Ndambi *et al*. 2019). [Chapters 7.2.6, 7.2.7]

About 190 million tons of inorganic fertilizer was used in agriculture in 2018 (Food and Agriculture Organization of the United Nations [FAO] 2020a). By 2025 demand for inorganic fertilizer for agricultural use is expected to reach 208 Mt (International Fertilizer Association [IFA] 2021a). [Chapter 7.3.2] Only a small fraction of organic waste materials (e.g., 5 per cent in the European Union) is used as fertilizer (United Nations Environment Programme [UNEP] 2019). Proper waste treatment, combined with recycling, has the potential to reduce demand for inorganic fertilizers (Christodoulou and Stamatelatou 2016; Schneider *et al*. 2019; Rutgersson *et al*. 2020). [Chapter 7.2.5]

Use of both inorganic and organic fertilizers for crop production tends to be rather high in some countries. In a number of them, estimates of nitrogen from inorganic and organic fertilizers are more than 200 kg of nitrogen per hectare (ha) of cropland per year (FAO 2020b). However, fertilizers use is particularly low in sub-Saharan Africa (SSA) (e.g., <50 kg N/ha/year) (Sutton *et al*. 2013). [Chapter 7.3.2]

The nitrogen in inorganic fertilizers is derived using the Haber-Bosch artificial nitrogen fixation process (Erisman *et al*. 2008). However, nitrogen and other nutrients can be recycled or transferred through the use of organic inputs (e.g., manure) (Fairhurst 2012; Bouwman *et al*. 2013; FAO 2020a; FAO 2020b). [Chapter 7.3.2]

Annually, livestock manure left on pasture and applied to soil contains about the same amount of nitrogen as that used from inorganic fertilizers (FAO 2020a; FAO 2020b). Manure probably contributes at least half the phosphorus used in crop and pasture production (Bouwman *et al*. 2013). The amount of manure nitrogen left in pastures could be about three times the amount applied to soil (FAO 2020b). [Chapter 7.3.2]

The amount of applied nitrogen taken up by cereal plants during the first year of fertilizer application can range from 30-50 per cent. In well-managed fields cereals take up about 40-65 per cent of fertilizer nitrogen, 15-25 per cent of fertilizer phosphorus, and 30-50 per cent of fertilizer potassium in the first year of application. Subsequent crops benefit from some of the fertilizer nutrients left in the soil by the first crop. For example, most of the phosphorus applied can be used by subsequent crops. [Chapter 7.4.1]

Fertilizers should be handled with care during transportation, storage and field application to avoid losses and exposure to hazards. [Chapters 7.2.6, 7.2.7, 7.2.8, 7.4.1, 7.4.2]

The fertilizer distribution chain is especially long in SSA, above all in landlocked countries. In general, fertilizer prices are highest in SSA compared with other regions. A large portion of retail costs are for transport, as fertilizer is mostly carried by road within countries and throughout the continent (Wanzala and Groot 2013; Gro Intelligence 2016; Benson and Mogues 2018) [Chapters 7.4.3, 7.4.4, 7.5.1]

Direct drivers that tend to increase the use of fertilizers include fertilizer subsidies, practices and technologies that make fertilizer use profitable, access to input credit and markets for inputs and produce and use of information and communication technology. Factors that can contribute to decreased fertilizer use include policies, practices and technologies that improve nutrient use efficiency, as well as dietary choices and efforts to reduce food loss [Chapters 7.5.1, 7.5.2].

7.2 Types of fertilizers and fertilizer management

7.2.1 The nutrients required by plants

To grow well, plants require an adequate supply of primary nutrients (nitrogen [N], phosphorus [P] and potassium [K]), secondary nutrients (calcium [Ca], magnesium [Mg], sodium [Na] and sulphur [S]), and micronutrients or trace elements (boron [B], copper [Cu], chlorine [Cl], iron [Fe], manganese [Mn], molybdenum [Mo] and zinc [Zn]) (EU 2019). Primary and secondary nutrients are also referred to as "major nutrients" or "macronutrients" (Roy *et al*. 2006).

Soil is a main source of nutrients, but it may not provide adequate amounts of all the nutrients required by plants in the correct proportions. Fertilizers can be applied to address deficiencies in essential nutrients, maintain soil fertility, support productive and nutritious crops, and maximize economic returns.

7.2.2 Major fertilizer types

Fertilizers are loosely grouped as inorganic or organic, depending on how they are produced and their sources. Fertilizing products can also be grouped into a larger number of categories. For example, in the 2019 European Union (EU) regulation concerning the marketing of fertilizing products, Annex I on Product Function Categories (PFCs), seven PFCs are designated: 1) fertilizer (organic, organo-mineral and inorganic); 2) liming material; 3) soil improver (organic and inorganic); 4) growing medium; 5) inhibitor; 6) plant biostimulant; and 7) fertilizing product blend (European Union [EU] 2019).

Inorganic and organic fertilizers, organo-mineral fertilizers (which combine inorganic and organic fertilizers), biostimulants (products that stimulate plant growth but do not provide nutrients)

and enhanced efficiency fertilizers (whose sources can be inorganic or organic) are described below.

Inorganic fertilizers

Inorganic fertilizers are nutrient-rich. They are produced industrially by chemical processes, mineral extraction or mechanical grinding (FAO 2019a). They are also referred to as "mineral fertilizers", "synthetic fertilizers" (when they are the result of a chemical process), "chemical fertilizers" and "conventional fertilizers". Urea, which provides nitrogen, is a "synthetic organic fertilizer" (University of Tennessee 1999) which is usually considered an inorganic fertilizer (Fairhurst 2012).

Inorganic fertilizers can be "straight fertilizers" (containing declarable amounts of only one of the three primary nutrients N, P and K) or "compound fertilizers" (containing declarable amounts of at least two of these nutrients) (FAO 1991). The 2019 EU regulation defines a "straight solid inorganic macronutrient fertilizer" as one with a declared content of either one macronutrient (N, P, K, Ca, Mg, Na, S) or only one primary macronutrient (N, P, K) and one or more secondary macronutrients (Ca, Mg, Na, S); it defines a "compound solid inorganic macronutrient fertilizer" as one with a declared content of more than one primary macronutrient (N, P, K) or more than one secondary macronutrient (Ca, Mg, Na, S) and no primary macronutrient (N, P, K) (EU 2019, pp. L 170/45-L 170/46).

Solid compound inorganic fertilizers may be further divided into "mixed fertilizers", produced by a physical process, and "complex fertilizers", produced by a process of chemical reaction (FAO 1991). All the nutrients in complex fertilizers are present in the same granule.

Globally, the inorganic fertilizers most commonly used in agriculture include urea, NPK fertilizers, ammonium phosphates and potassium chloride. Inorganic fertilizers that do not contain nitrogen, phosphorus or potassium include those supplying magnesium (e.g., dolomite), boron (e.g., boric acid), copper (e.g., copper sulphate), iron (iron sulphate), manganese (sulphate of manganese), molybdenum (ammonium molybdate) and zinc (zinc sulphate) (Fairhurst 2012).

Organic fertilizers

Organic fertilizers are derived from once-living organisms and are rich in carbon. They include animal manures (the most important fertilizer input for agricultural soils in many countries), composts, sewage sludge, green manures, peat, crop residues and by-products of industries handling agricultural produce (Codex Alimentarius Commission 2013). It should be noted that the use of nutrient sources like manures and crop residues entails the recycling of nutrients, and that the transfer of nutrients can occur, for example between fields and between farms.

Many organic fertilizers are used as soil amendments since they improve the soil's physical fertility (e.g., providing higher porosity or better infiltration capacity) and biological properties, as well as being a steady source of nutrients (Edmeades 2003; Abdou *et al*. 2016).

Organic fertilizers can largely be categorized as belonging to one of two groups: those that are end-products of processing, for example composts and digestates (Weithmann *et al*. 2018), and those that are used unprocessed, such as fresh livestock manures applied to the soil, green manures (e.g., cover crops) and crop residues left in the field.

Industrial processing of organic fertilizers presents an opportunity for refinement. For example, it makes possible products with higher and more reliable (instead of variable) nutrient concentrations, as well as the removal of certain contaminants.

Organo-mineral fertilizers

Organo-mineral fertilizers are a combination of organic and inorganic fertilizers (Kominko, Gorazda and Wzorek 2017). They probably represent a very small share of the global fertilizer market. Their market value is about 2 per cent of that of all fertilizers sold in the EU (European Commission 2016; Fertilizers Europe 2019).

Biostimulants

Biostimulants are not fertilizers *per se*. According to the definition in the International Code of Conduct for the Sustainable Use and Management of Fertilizers (FAO 2019a), they are products that stimulate plant growth through the synthesis of growth-promoting substances and/or plant nutrition processes independently of nutrient content; their aim is to improve plant nutrient use efficiency or uptake, plant tolerance to abiotic stress, or crop quality traits. While biostimulants (according to the 2019 EU regulation) "are not as such inputs of nutrients, [they] nevertheless stimulate plants' natural nutrition processes. Where such products aim solely at improving the plants' nutrient use efficiency, tolerance to abiotic stress or quality traits, or increasing the availability of confined nutrients in the soil or rhizosphere, they are by nature more similar to fertilizing products than to most categories of plant protection products. They act in addition to fertilizers, with the aim of optimizing the efficiency of those fertilizers and reducing the nutrient application rates" (EU 2019, p. L 170/4).

Diverse organic and inorganic substances and/or microorganisms can be considered biostimulants (Rouphael and Colla 2018; EU 2019). Examples of biostimulants include seaweed extracts, humic substances, amino acids, bacteria that promote plant growth (Halpern *et al*. 2015) and inorganic salts (Torre, Battaglia and Caradonia 2016).

7.2.3 Cropping systems and fertilizer use

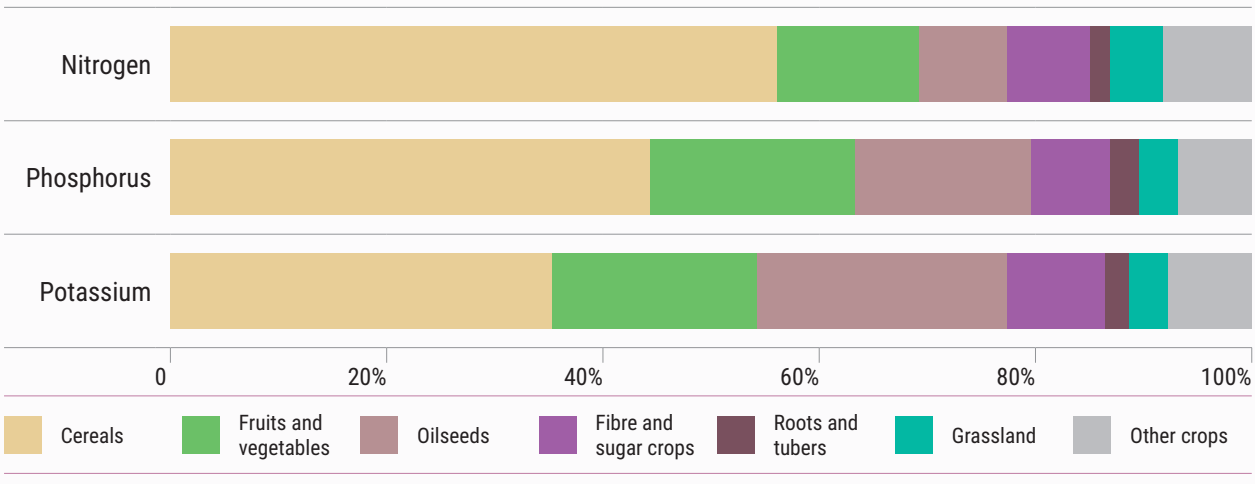
Of the fertilizers applied to crops globally, about half (49 per cent) are applied to cereals, 16 per cent to fruits and vegetables, and 13 per cent to oil crops (Heffer, Guère and Roberts 2017) (Figure 7.2-1). Partitioning of fertilizers among crops varies from one country to another. For example, about 32 per cent and 4 per cent of the N fertilizer used in China and India, respectively, were reported to be applied to fruits and vegetables (Heffer, Guère and Roberts 2017).

Global data on partitioning of organic fertilizers among crops are lacking. One study has reported that compared with wheat-maize rotation, greenhouse vegetables received about four times more N from inorganic fertilizer and about 30 times more N from manure in farmers' fields in China (Ju *et al*. 2007).

7.2.4 Enhanced efficiency fertilizers

Enhanced efficiency fertilizers (EEFs) have been defined by the Association of American Plant Food Control Officials (AAPFCO) as "fertilizer products with characteristics that allow increased plant uptake and reduce the potential of nutrient losses to the environment (e.g., gaseous losses, leaching, or run-off) when compared with an appropriate reference product" (Halvorson *et al*. 2014). They allow slow release or prolonged availability

Figure 7.2-1 Proportion of nutrients used by crop categories globally. Heffer, Guère and Roberts (2017).



of nutrients to plants. However, according to a review of the use of EEFs in nitrogen management (Li, T. *et al*. 2017), they do not always have higher use efficiencies than conventional fertilizers and they can sometimes be less efficient. The authors reported that the effectiveness of EEFs depends on the type of fertilizer, the cropping system and biophysical conditions. They concluded that while EEFs can play a significant role in sustainable agricultural production, prudent use of these fertilizers requires eliminating fertilizer mismanagement and implementing knowledge-based N management practices.

EEF systems can be categorized based on type of coating material (organic-and inorganic), mode of action (including urease activity inhibition and nitrification inhibition), composite nutrient formulations, use of naturally occurring minerals or their synthetic versions, and the technologies used in their development (e.g., nanotechnology) (Dimpka *et al*. 2020). Those already on the market or under development include controlled release fertilizers, slow release fertilizers, stabilized fertilizers and nanofertilizers.

Controlled release fertilizers

Controlled release fertilizers (CRFs) are mostly urea and NPK based (Mordor Intelligence 2021a). They are coated with a polymer (typically resin, plastic or biopolymers), which facilitates nutrient release according to plant requirements at different growth stages over weeks to months (Trenkel 2010; International Fertilizer Association [IFA] 2017). Examples of CRFs include polymer coated urea, polymer-sulphur coated urea, and polymer coated NPK (Mutegi, Kiwia and Zingore 2019).

CRFs typically require a minimum level of persistence in the environment (12-18 months) to achieve their intended purpose (European Chemicals Agency 2019). An issue with these fertilizers could be the degradation rate of the coating material. For example, the criteria for polymers in the 2019 EU fertilizer regulation include the requirement that at least 90 per cent of the organic carbon should be converted into carbon dioxide (CO_2) no more than 48 months after the end of its claimed functionality period (EU 2019, L 170/131). However, some of the coating materials used are non-degradable (Naz and Sulaiman 2016). Coated fertilizers also tend to be more expensive than traditional ones (Naz and Sulaiman 2016).

Slow-release fertilizers

Slow-release inorganic fertilizers are not coated with polymers, but they release their nutrients (e.g., condensed chemical forms of urea) slowly, over weeks to months (Trenkel 2010; IFA 2017). They can also have organic sources (Mutegi, Kiwia and Zingore 2019).

Urea supergranules/briquettes (weighing about 1-3 grams), used together with Urea Deep Placement (UDP) technology, are an example. This technology was developed by the International Fertilizer Development Center (IFDC) and its collaborators to reduce nitrogen losses. It is mainly used in the production of irrigated rice in Bangladesh (International Fertilizer Development Center [IFDC] 2017). Urea supergranules can also contain phosphorus and potassium (Chien, Prochnow and Cantarella 2009).

Another example is direct application of phosphate rock. According to several authors (Chien *et al*. 2011), the solubility of phosphate rock varies depending on its source. Although the application of slow release types may not be a suitable option for annual crops, it could be suitable for perennials in acid soils where rainfall is prevalent (Weeks and Hettiarachchi 2019). In addition, partial acidulation of phosphate rock has been reported to improve phosphate use efficiency (Bationo and Kumar 2002).

Stabilized fertilizers

Stabilized fertilizers are nitrogen fertilizers to which urease inhibitors and/or nitrification inhibitors are added (Trenkel 2010; Ribeiro *et al*. 2016; IFA 2017). For example, neem oil, which is known to have nitrification inhibition properties, is used to coat urea in India (Singh 2016). A meta-analysis of the effect of urease and nitrification inhibitors reported improved crop yields and improved nitrogen use efficiency, but their effectiveness depended on environmental and management factors

(Abalos *et al*. 2014). In meta-analyses, nitrous oxide (N_2O) emissions were less for fertilizers with both urease and nitrification inhibitors than for fertilizers without inhibitors, while urease inhibitors alone did not appear to reduce emissions (Snyder *et al*. 2014).

Nanofertilizers

Nanofertilization is a recent technology that involves the use of particles approximately 1-100 nanometres in size (Mikkelsen 2018). (One nanometre is one billionth of a metre.) Fertilizers (e.g., urea fertilizers) can be coated with nanomaterials to reduce the rate of dissolution of nutrients (Duhan *et al*. 2017; Iqbal, Umar and Mahmooduzzafar 2019).

Studies have suggested that nanofertilizers could be more beneficial than conventional fertilizers. Nanotechnology can enhance crop productivity and reduce nutrient losses (Dimkpa and Bindraben 2018). Coating microbial plant biostimulants with plyometric nanoparticles can improve their shelf life (Duhan *et al*. 2017). Zinc applied to leaves has a low degree of penetration and limited mobility in the phloem. Nanotechnology can improve plant uptake and movement in the phloem when the zinc is applied to plant leaves (Rios, Garcia-Ibañez and Carvajal 2019).

Despite the potential benefits of nanotechnology, its use by farmers could be constrained by lack of technical know-how as well as by economic factors (Hosseini, Nazrai and Lashgarara 2011). In India farmers may soon have access to nanofertilizers, as they have been researched and developed in that country and there are plans to apply them on farms under controlled conditions (*The Economic Times* 2019). According to Dimkpa and Bindraban (2018), large-scale industrial production of nanofertilizers has yet to take off. Furthermore, there is need to develop multi-nutrient fertilizers that provide balanced nutrition to plants and humans (Dimpka *et al*. 2020). At the same time, the possibility of negative effects and a paucity of research information on this technology have raised concerns among some scientists (Achari and Kowshik 2018; Dimkpa and Bindraban 2018; Kah *et al*. 2018; Zulfiqual *et al*. 2019).

7.2.5 Nutrient recycling

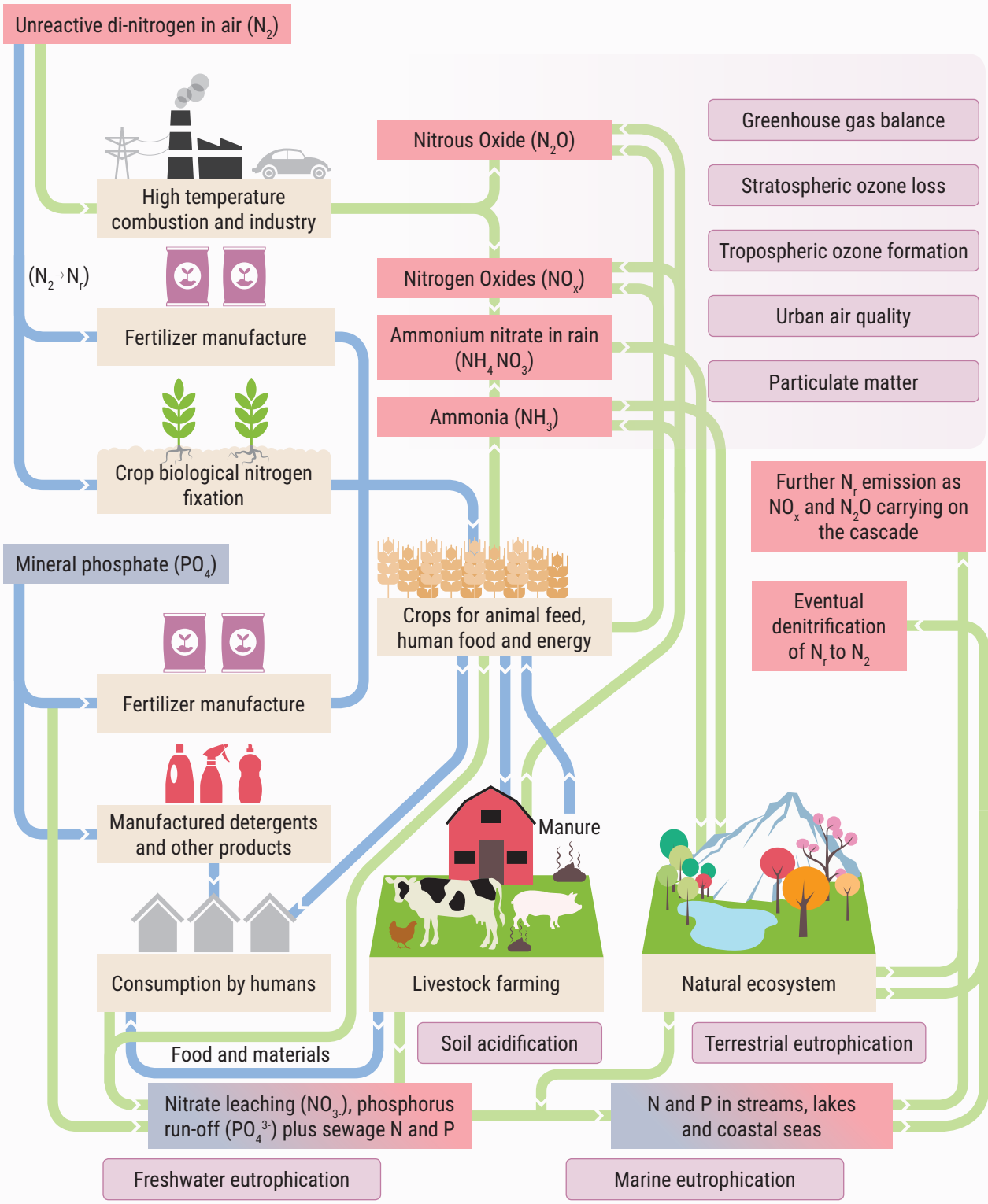
When inorganic or organic N, P and K are applied to crops, some of the edible plant parts are consumed by humans and animals, whose waste can be applied to soil to provide nutrients (Figure 7.2-2). Care should be taken to use waste that is safe for this purpose (see below).

There have been concerns about the long-term availability of phosphorus, as phosphate rock resources are finite (Li, B., Bicknell and Renwick 2019; Alewell *et al*. 2020; Bennett 2020). However, a recent report the United States Geological Survey concluded that phosphate rock shortages are not imminent. World resources of phosphate rock are estimated to be more than 300 billion tons(United States Geological Survey [USGS] 2020).

More nutrients are currently lost than recycled. According to a review of the literature by Schneider *et al*. (2019), studies have estimated that phosphorus from animal and human sources could potentially meet a large proportion of crop requirements. In the EU only 5 per cent of organic waste materials are recycled and used as fertilizer (UNEP 2019). Nevertheless, a substantial share of sewage sludge (biosolids) produced in some countries is used as fertilizer (Figure 7.2-3) (Christodoulou and Stamatelatou 2016; Rutgersson *et al*. 2020).

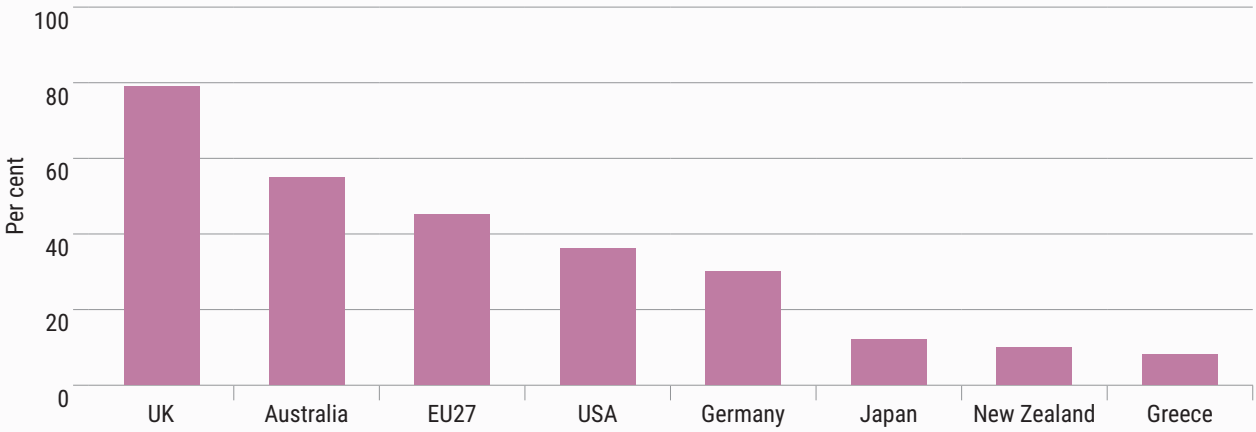
The use of untreated or contaminated organic inputs could present environmental and human health risks (see Chapter 9, e.g., for the potential effects of pathogens and endocrine disrupters in organic inputs on human health [Chapter 9.2.2] and the potential effects of endocrine disrupters in organic inputs on soil invertebrates [Chapter 9.3.2]). Moreover, the nutrients in such inputs may not match plant nutrient requirements. It has been estimated that recycling all the excreta produced in Sweden in 2017 could have met up to 75 per cent of crop nitrogen and 81 per cent of phosphorus needs, but would have exceeded crop potassium needs by 67 per cent (Akram *et al*. 2019).

Figure 7.2-2 Simplified overview of nitrogen and phosphorus flows highlighting major anthropogenic sources, and the associated environmental concerns. Sutton *et al.* (2013).



N_r= Reactive Nitrogen and NO_x = Collective reference term for nitric oxide (NO) and nitrogen dioxide (NO_2).

Figure 7.2-3 Percentage of biosolids produced nationally used in agriculture in developed countries.
Christodoulou and Stamatelatou (2016).



7.2.6 The 4Rs of nutrient management

Use by farmers of the 4R nutrient stewardship (4RNS) approach – the right nutrient source, the right application rate, the right time and the right place – is key to minimizing nutrient losses. This approach is applied, for example, in precision agriculture and is a component of integrated soil fertility management (Chapter 7.5.1). A recent development is a global 4RNS framework which embeds the 4Rs in a cropping system and landscape that support the environmental, economic and social dimensions of sustainability (Fixen 2020).

Nutrient sources

Fertilizers should match crop needs and soil properties (Johnson and Bruuslsema 2014). Balanced nutrition ensures maximum benefits. Factors influencing the choice of fertilizers include their form (e.g., dry granular products, fluids, straight vs. blends) and their availability (Johnson and Bruuslsema 2014).

Application rates

In general, the rate at which fertilizers are applied should meet crop needs, maintain soil fertility, and not exceed plant requirements (to avoid wastage and pollution) (Mutegi, Kiwia and Zingore 2019).

Application rates are specific to each site and cropping system. They are estimated based on the crop's nutrient requirements, the capacity of the soil to supply nutrients, the target yield, the attainable yield under local climatic conditions, fertilizer cost, and the value of the crop products (Fairhurst 2012). Environmental pollution resulting from fertilizer use should also be considered. Determining application rates that meet farmers' needs, ensure profitability for the fertilizer industry and reduce agricultural pollution can be challenging (Kanter, Zhang and Mauzerall 2014).

Nutrient placement

Fertilizers should be applied where crops can use them (Mutegi, Kiwia and Zingore 2019). Methods of fertilizer application for both inorganic and organic fertilizers, and for fertilizers in solid, liquid or gaseous form, include applying the fertilizer to soil by broadcasting or top dressing; applying it in the same furrow or hole as the seed at planting; applying it on the soil surface or in subsoil, either in the same row as seeds or between seed rows; spot application; and deep placement and side dressing (Nkebiwe *et al*. 2016).

Applying fertilizer close to a seed or plant increases plant uptake and, in the case of nitrogen, reduces the risk of volatilization. However, some fertilizers damage the seed or plant if they are placed too close to it. Broadcasting requires less investment in labour and equipment than placing fertilizer close to a seed or plant.

For example, deep placement of fertilizer briquets (e.g., in irrigated rice) demands more labour than the traditional practice of broadcasting (Pasandaran *et al*. 1999). On the other hand, using applicators for deep placement can reduce labour requirements (Wohab *et al*. 2017).

Water soluble fertilizers may be dissolved and applied in the form of foliar sprays, fertigation (e.g., via drip irrigation) or injection into soil. Plants sometimes respond better to application of liquid fertilizers than to that of granular ones. Soil application of liquid inorganic sources of phosphorus, nitrogen and zinc has been shown to produce significantly more grain than application of granular inorganic fertilizers in wheat grown on calcareous soil (Holloway *et al*. 2001).

Foliar application of zinc and iron has been reported to increase concentrations of these micronutrients in harvested grain (Melash *et al*. 2019). The concentration of zinc in wheat almost doubled with foliar application of zinc, but no noticeable change was observed with soil application (Cakman and Kutman 2018). In a review of selenium fertilization strategies, foliar application tended to have more positive responses than soil application for selenium while residual effects lasted longer in the case of soil-applied selenium (Ros *et al*. 2016). Foliar application is more beneficial than soil fertilization when plant demand for nutrients exceeds the roots' capacity to absorb nutrients or soil conditions limit plant uptake of nutrients (Oosterhuis 2009). In a study on vegetables, nitrogen leaching was less in the case of drip irrigation than in that of flooding irrigation (Lv *et al*. 2018). In a study on maize, fertilizer use efficiency was greater with drip irrigation than under rain-fed conditions (Wu, Xu *et al*. 2019). Fertigation allows matching nutrient application with crop demand, as well as better distribution of nutrients in the root zone (Kafkafi and Tachitzky 2011).

Aerial spraying is another method of applying inorganic and organic fertilizers in either solid or liquid form. This method allows application over large areas in a short time. However, it is expensive, requires care to avoid contamination of open waters (Roy *et al*. 2006), and results in losses through volatilization.

Timing of applications

Fertilizers should be available when a crop needs them (Mutegi, Kiwia and Zingore 2019). Basal ("starter") fertilizer, mostly containing N, P and K, is applied at planting or during early stages of plant growth; N fertilizers are also applied during later stages of crop growth as top dressing (Fairhurst 2012). Some crops may not require nitrogen at the early stages of plant growth. In direct seeded rice, for example, basal nitrogen was not necessary until the fourth leaf stage (Chen *et al*. 2018).

Matching nutrient supply with crop requirements improves use efficiency. Basal fertilizers tend to have minimal amounts of nitrogen in order to minimize nitrogen losses, which are more likely when plant roots are small or non-existent than when the roots are well established and nitrogen uptake is high.

7.2.7 Fertilizer storage

Inorganic fertilizers

Fertilizer quality may decline during storage. Some inorganic granulated fertilizers can absorb moisture during storage, making application difficult. Ammonium nitrate and urea are more hygroscopic than most other fertilizers (i.e., they take up moisture from their surroundings more readily) (Albadarin, Lewis and Walker 2017). Such fertilizers should be stored under dry conditions.

Organic fertilizers

Proper storage of organic fertilizers is necessary to minimize nutrient losses and, subsequently, environmental pollution. A global meta-analysis of measures to reduce ammonia emissions from agricultural systems (Ti *et al*. 2019) reported that manure storage management could reduce ammonia emissions by 70-82 per cent. Storage methods that contribute to a reduction of nutrient losses include stacking and compressing manure heaps and protecting manure from wind, water and sunlight, for example, by storing it in a structure with an impermeable floor, a roof and manure covering (Teenestra *et al*. 2015; Ndambi *et al*. 2019).

Poor storage conditions can contribute to an increased risk of fire (e.g., for composts) or the growth of fungi. Accidental explosions of compost being treated have been reported (Miyake 2006). Increased water content can stimulate the development of fungi in stored organic fertilizers (Kasprzycka *et al*. 2018).

7.2.8 Hazard classification and fertilizer compatibility

Based on the literature search conducted for this report, little information on hazard classification or the compatibility of fertilizer materials is available for organic fertilizers.

Hazard classification

Ammonium nitrate based fertilizers are thermally stable under normal conditions, but external heat can initiate their decomposition and, for example, cause detonation and the release of toxic gases (European Fertilizer Manufacturers Association [EFMA] 2007). Guidelines exist for the transport, storage and management of such fertilizers (see Table 7.1 for examples).

Regulation EC 2003/2003 of the European Parliament and the Council states that "ammonium nitrate fertilizers of high nitrogen content should conform to certain characteristics to ensure that they are harmless. Manufacturers should ensure that all high nitrogen content ammonium nitrate fertilisers have passed a test of resistance to detonation before those fertilizers are placed on the market" (EU 2003, p. L 304/2).

Under the International Maritime Dangerous Goods (IMDG) code (International Maritime Organization 2019), copper granulates, manganese sulphate, zinc sulphate monohydrate, and blends containing 2 per cent copper granules, 1 per cent manganese sulphate granular or 1 per cent zinc sulphate monohydrate are classified as marine pollutants (see, for example, the Kenya Bureau of Standards code of practice for blending fertilizers; Kenya Bureau of Standards 2014).

Figure 7.2-4 Compatibility of various fertilizer materials. Adapted from Fertilizers Europe (2006).

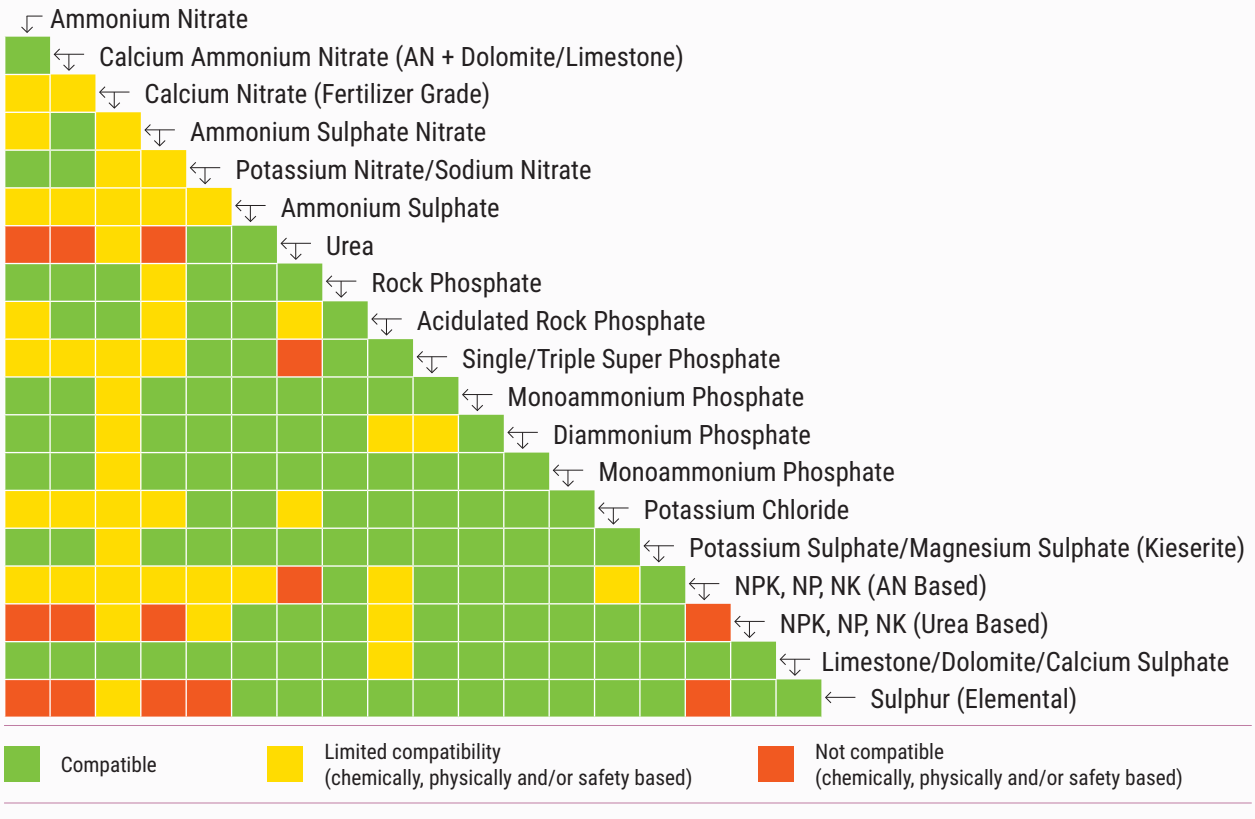


Table 7.1 Classification of inorganic fertilizers for transportation, storage and handling. European Fertilizer Manufacturers Association (EFMA) (2004); EFMA (2007); Fertilizers Europe (2006); Fertilizers Europe (2007); United States Environmental Protection Agency (US EPA) (2015).

Fertilizers	Hazard description
Sea transport classification	
Type A: Straight N fertilizers containing ammonium nitrate (AN) >80% and dolomite, limestone and/or calcium carbonate ($CaCO_3$); straight N fertilizers containing AN >70% and other inorganic substances not covered above; compound fertilizers containing AN >70%; straight N fertilizers containing AN and ammonium sulphate (AS) in which 45%<AN<70% and AN+AS>70%	Oxidizers, class 5.1, UN 2067. These fertilizers need to pass the UN resistance to detonation test.
Type B: NPK fertilizers with 45%<AN<70%, with ≤0.4% combustible/ organic materials; NPK fertilizers with AN<45%, with unrestricted combustible/organic materials and capable of self-sustaining decomposition	Class 9, UN 2071
Type C:	Non-hazardous
Rail transport classification	
Anhydrous ammonia: NH_3	Gas, toxic and corrosive substance (class 2, hazard identification number 268)
Ammonia solution: relative density <0.880 at 15°C in water, with >50% ammonia	Gas, toxic and corrosive substance (class 2, hazard identification number 268)
Ammonia solution: relative density <0.880 at 15°C in water, with >35% but not ≤ 50% ammonia	Toxic gas (class 2, hazard ID 20)
Ammonia solution: relative density 0.880-0.957 at 15°C in water, with >10% but ≤35% ammonia	(class 8, hazard ID 80)
Storage, handling and transportation classification	
Ammonium sulphate: no specific danger known	Not classified as hazardous
Urea:	Not detonable or inflammable
Potassium nitrate: KNO_3	Oxidizer, class 5.1, UN 1486
Sodium nitrate: $NaNO_3$	Oxidizer, class 5.1, UN 1498
Magnesium nitrate: $Mg(NO_3)_2$ or MgN_2O_6	Oxidizer, class 5.1, UN 1474
Calcium nitrate: $Ca(NO_3)_2$	Oxidizer, class 5.1, UN 1454

Some organic inputs, too, are considered potentially dangerous during transportation (International Maritime Organization 2006). For example, seed cake, the residue that remains after oil is extracted from oil-bearing seeds and is mainly used as animal feed or fertilizer, should be substantially free from flammable solvent and properly aged before shipment (BMT n.d).

Compatibility of fertilizer materials

Although most fertilizer materials are compatible when they are blended, some have limited compatibility or are incompatible (Figure 7.2-4). For example, ammonium nitrate and ammonium sulphate are incompatible as this mixture is potentially detonable (Fertilizers Europe 2006).

7.3 Fertilizer demand, sales and use

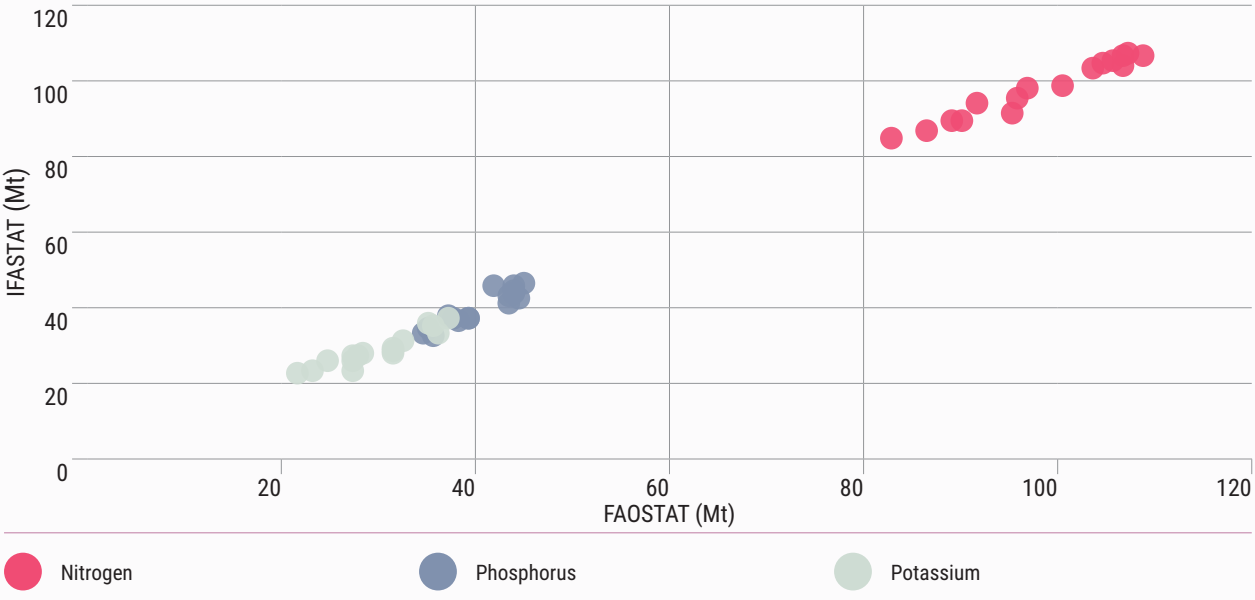
Key fertilizer data sources are the Food and Agriculture Organization of the United Nations (FAO) and the International Fertilizer Association (IFA). FAO data are collected directly from countries as part of official reporting to FAO and disseminated in the FAOSTAT database. IFA data are collected from national fertilizer industry correspondents and disseminated in the Association's IFASTAT database. For the purposes of this analysis, where the focus is largely on global and regional trends for the three main macronutrients, FAO and IFA data are fairly comparable (Figure 7.3-1).

The fertilizer data in FAOSTAT, at the time of this report cover years up to 2018. FAO's most recent *World Fertilizer Trends and Outlook* report covers the period 2017-2022 (FAO 2019b). Similar recent reports by IFA cover 2019-2023 (IFA 2019a; IFA 2019b) and 2020-2024 (IFA 2020a; IFA 2020b). The FAO report was developed by the Fertilizer Outlook Expert Group, comprising representatives of the FAO, IFA, the Fertilizer Association of India (FAI), the International Fertilizer Development Center (IFDC) and The Fertilizer Institute (TFI). IFA outlook reports are developed by IFA with input from the IFA Agriculture Committee and IFA members. The present report uses the FAO database for statistics up to 2018. It uses FAO (2019b), IFA (2019a and 2019b), and IFA (2020a and 2020b) for projections.

This section focuses mainly on inorganic fertilizers, for which considerable data are available. Some estimates for organic fertilizers are also included, but the only publicly available data on quantities are for nitrogen from animal manure (based on statistics concerning animal stocks and the application of the Guidelines of the Intergovernmental Panel on Climate Change [IPCC]) (FAO 2020b). Regional data for Europe include some countries in Eastern Europe and Central Asia (EECA), such as the Russian Federation and Belarus. Data for Latin America include countries in South and Central America and the Caribbean.

The quantities of nitrogen, phosphorus and potassium presented in this report are for elemental nitrogen (N), phosphorus pentoxide

Figure 7.3-1 Comparison of FAOSTAT and IFASTAT data on fertilizer use in agriculture for the period 2002- 2017. IFA (n.d.); FAO (2020a).



Note: the points indicate different years, and values are in million tonnes (Mt)

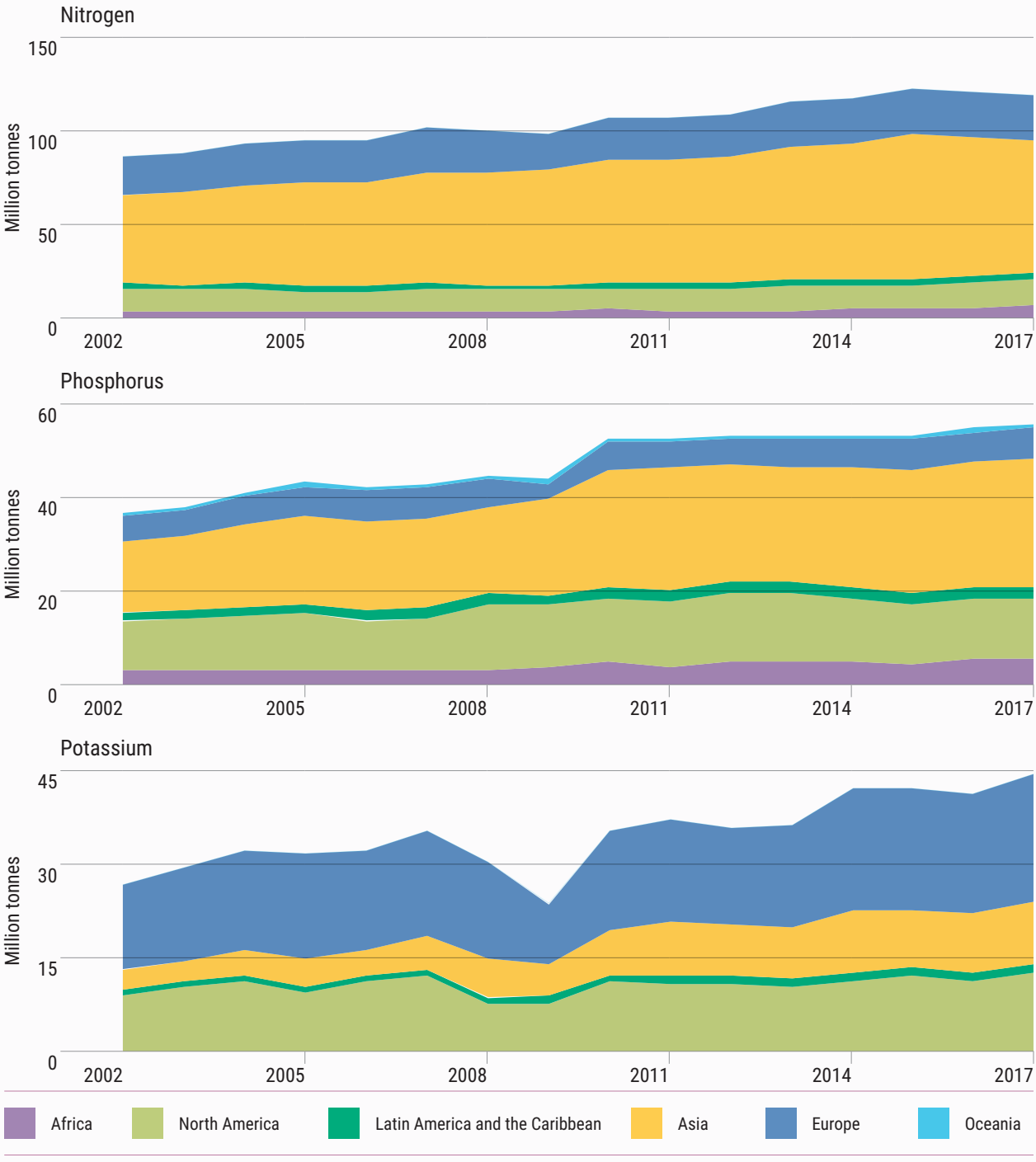
(P_2O_5) and potassium oxide (K_2O). Phosphoric acid (H_3PO_4) is an intermediate product in liquid form used in the manufacture of a wide variety of fertilizers and chemical products (IFA 2020c), the amount of phosphorus in the acid is expressed as P_2O_5. Potassium is also commonly referred to as potash (CropLife 2017; IFA 2020d). As such, the quantities of phosphoric acid and potash presented are for P_2O_5 and K_2O, respectively.

7.3.1 Fertilizer production and supply (current and projected)

Inorganic fertilizer

For the production data from FAO presented below, "production data represent the tonnes of nutrients manufactured into fertilizer products" (FAO 2020c). On the other hand supply data includes not only

Figure 7.3-2 Fertilizer production (Mt). FAO (2020a).



finished products used for plant nutrition, but also finished products used for animal feed and for industrial uses, intermediate products and fertilizer raw materials (IFA 2020e). Supply refers to "the capability or maximum achievable production (potential), derived by multiplying capacity by the highest achievable operating rate" (IFA 2019b).

Between 2002 and 2018 it is estimated that global inorganic fertilizer production increased from 87 to 120 million tons (Mt) for nitrogen, 34 to 44 Mt for phosphorus, and 27 to 45 Mt for potassium. Total production of the three nutrients is estimated to have increased from about 150 to 210 Mt (FAO 2020a) (Figure 7.3-2).

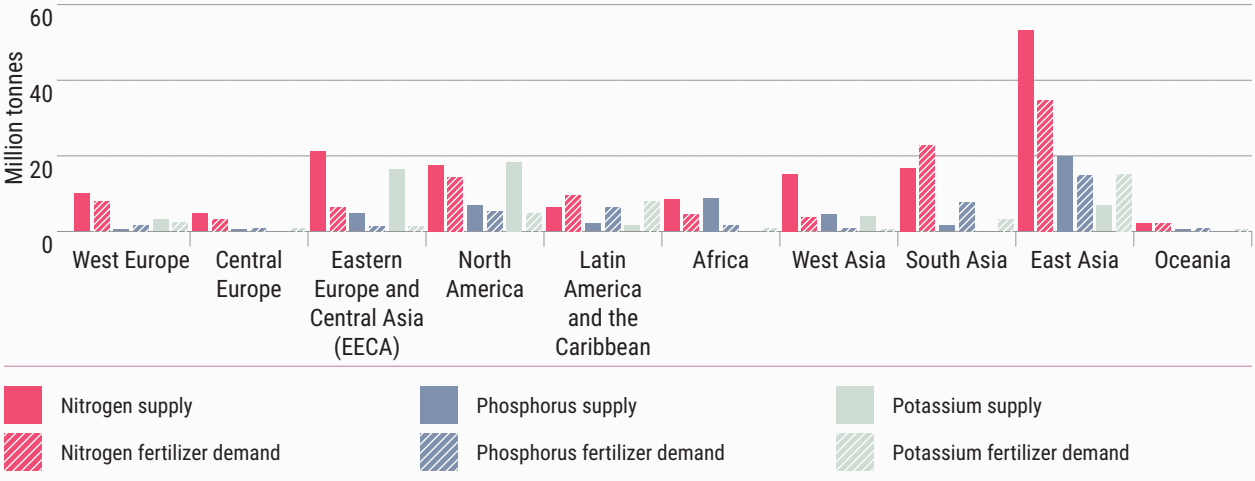
The FAO trends and outlook report and the IFA outlook report for 2019-2023 projected that global supply of ammonia in 2019 would be about 162 Mt and 155 Mt of N respectively, while both reports projected the supply of P_2O_5 and K_2O would be about 50 Mt each (FAO 2019b, IFA 2019b). Both the FAO and IFA reports projected a global nitrogen supply of about 162-163 Mt by 2022. They also projected that the supply of phosphorus and potassium (supplied as phosphoric acid and potash), would be about 52 Mt and 54 Mt, respectively, in 2022.

In general, Asia is the world's largest supplier of nitrogen and phosphorus; North America and EECA are the largest potassium suppliers (Figure 7.3-3). In China nitrogen production was about 41 Mt in 2015, but fell to 33 Mt in 2018 (FAO 2020a). Nitrogen production is expected to decline further in China between 2019 and 2023 because of continued capacity restructuring, and to increase in EECA, Latin America, Africa and India due to new capacity (IFA 2019b). During the same period phosphorus production is expected to increase in Africa, West Asia, Latin America and South Asia while potassium production is expected to increase in EECA, Canada and China (IFA 2019b).

Organic fertilizer, organo-mineral and biostimulant markets

Inorganic fertilizer is estimated to represent 80 per cent of the total market value of the EU fertilizer industry, with organic fertilizer representing around 5 per cent and organo-mineral fertilizer and biostimulants each representing about 2 per cent (EC 2016; Fertilizers Europe 2019). The economic value of biostimulants in the EU was estimated at between 200 and 400 million euros by the European Biostimulants Industry Council, according to Xu and Gellen (2018).

Figure 7.3-3 Estimated supply and fertilizer demand for nitrogen, phosphorus and potassium in 2019 (Mt).
IFA (2019b).



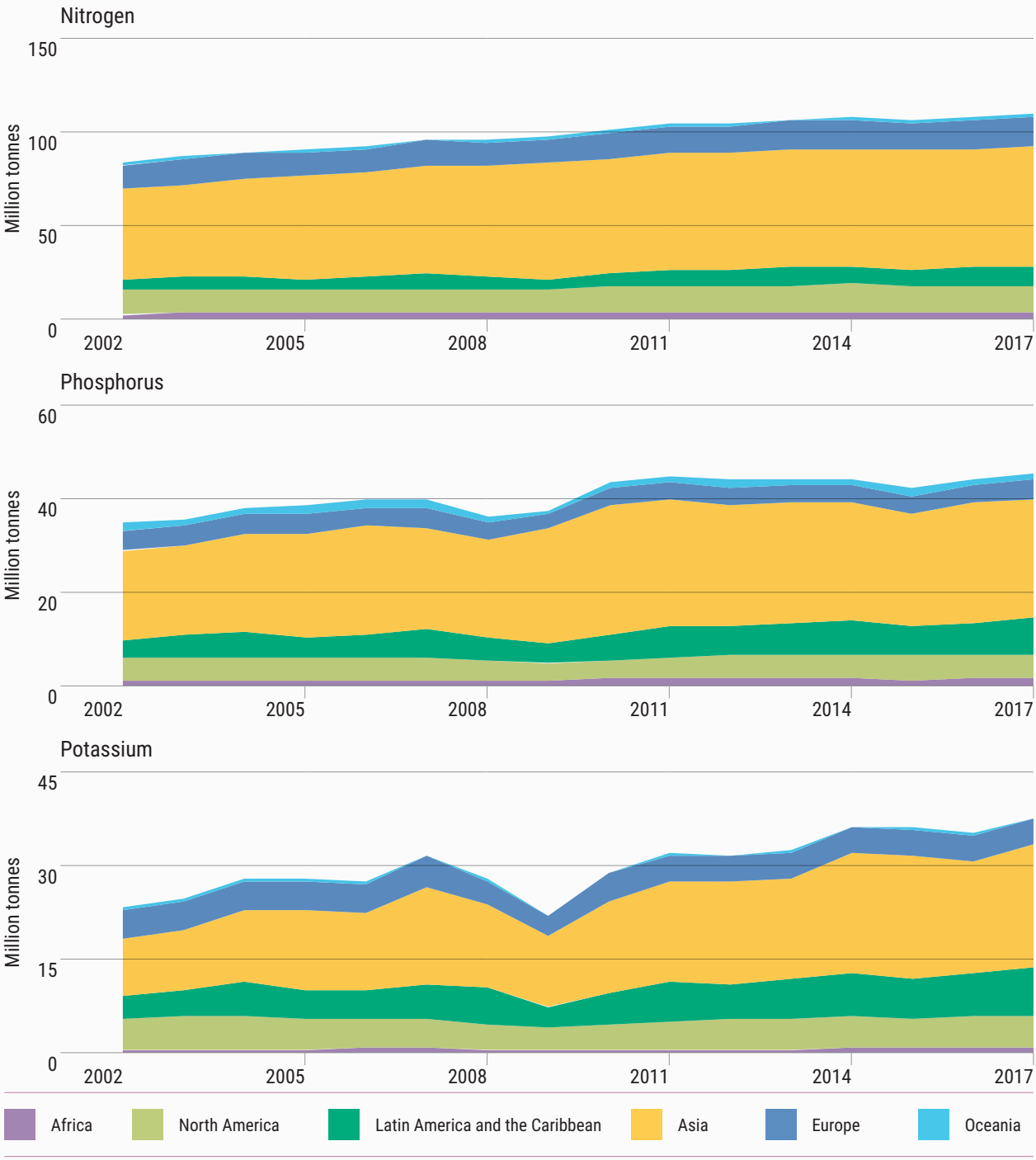
Note: Data for supply of phosphorus and potassium refers to P_2O_5 and K_2O from phosphoric acid and potash, respectively.

7.3.2 Fertilizer demand: N from inorganic fertilizer and manure, P and K from inorganic fertilizer

In this chapter "fertilizer demand" refers to fertilizer use in agriculture. Most of the fertilizer produced in the world is used for agricultural purposes (crop/plant production). For example, according to FAO data for 2018 the nitrogen, phosphorus and potassium used in agriculture were approximately 91, 92 and 87 per cent of the total amount produced in that year, respectively (FAO 2020a).

Between 2002 and 2018 the total amount of nutrients from inorganic fertilizer (nitrogen, phosphorus and potassium) used for agricultural

Figure 7.3-4 Nitrogen (N), phosphorus (P_2O_5) and potassium (K_2O) fertilizer use in agriculture (Mt), 2002-2017. FAO (2020a).



purposes globally increased from 140 Mt to 190 Mt of nutrients (FAO 2020a) and the amount of inorganic fertilizer used per hectare increased by about 23 per cent for nitrogen, 13 per cent for phosphorus and 56 per cent for potassium (FAO 2020e). Asia is the largest consumer of nitrogen, phosphorus and potassium from inorganic fertilizers, while Oceania and Africa consume the least (FAO 2020a) (Figure 7.3-4). Latin America is expected to be the main contributor to global growth in the medium term and Africa is expected to be the fastest growing market (International Fertilizer Association [IFA] 2021a). Between 2019 and 2022 agricultural demand for nitrogen was projected to increase from 107 Mt to 112 Mt, that for phosphorus from 47 Mt to 49 Mt, and that for potassium from 38 Mt to 40 Mt, according to the FAO trends and outlook report for 2017-2022. In 2025/2026 global demand for inorganic fertilizer is forecast by the to reach 208.3 Mt nutrients, compared with 198.2 MT in 2020/2021, with Latin America expected to be the main contributor to global growth in the medium term and Africa expected to be the fastest growing market (IFA 2020a; IFA 2020b; IFA 2021a).

Comparison of FAO estimates of the amount of nitrogen in manure left on pastures and in manure applied to the soil (based on statistics on animal stocks and Intergovernmental Panel on Climate Change [IPCC] conversion factors) with the amount of nitrogen used from inorganic fertilizers (based on data provided by countries) suggests that globally about 50 per cent of nitrogen supplied to the soil is from animal manure left on pastures and applied to soil annually(FAO 2020a; FAO 2020b). In general, the amount of nitrogen from manure left in pastures could be about three times the amount of nitrogen from manure applied to soil (Figure 7.3-5) (FAO 2020b). However, it should be noted that use of manure involves the transfer of nutrients across landscapes, while the nitrogen in inorganic fertilizers is a "new input" derived from the Haber-Bosch process which converts atmospheric nitrogen to ammonia by combining nitrogen with hydrogen (Erisman *et al*. 2008).

Figure 7.3-5 **Amount of nitrogen (N) excreted in manure which was left in pastures and applied to soil (Mt) (a) between 1961 and 2017, globally, and (b) in different regions (averages for 2003-2017).** FAO (2020b).

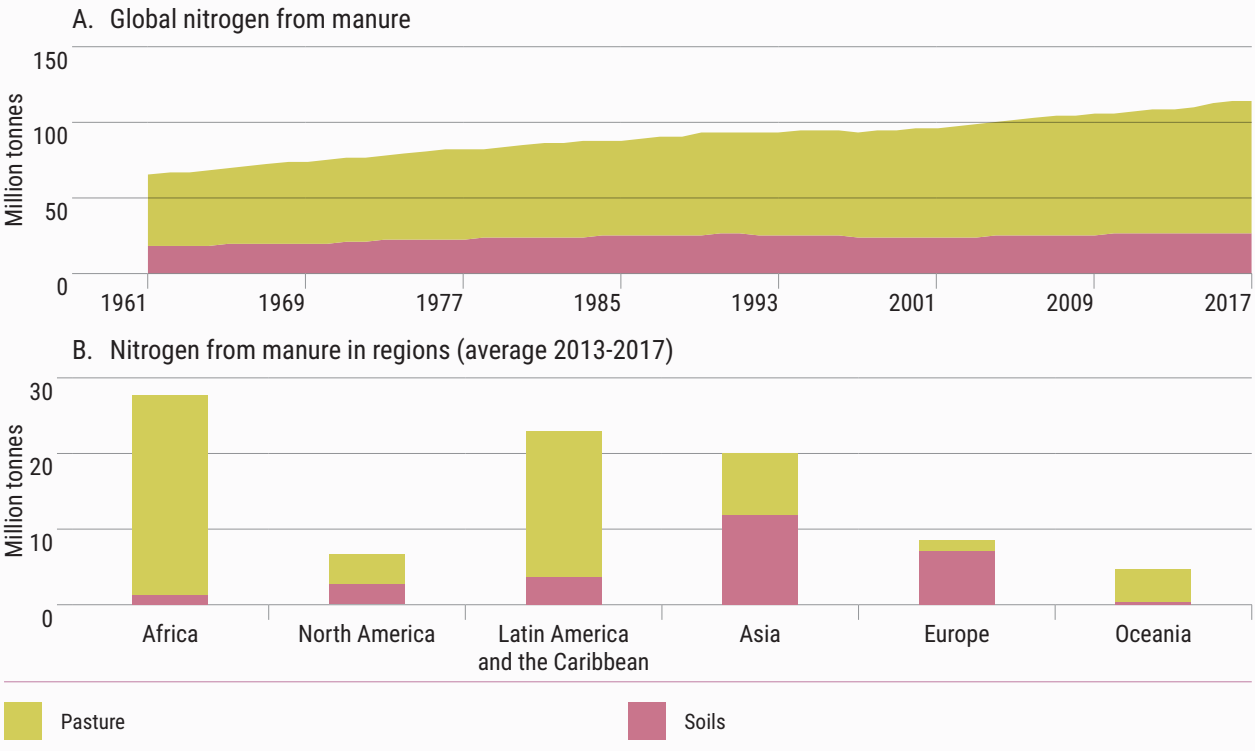
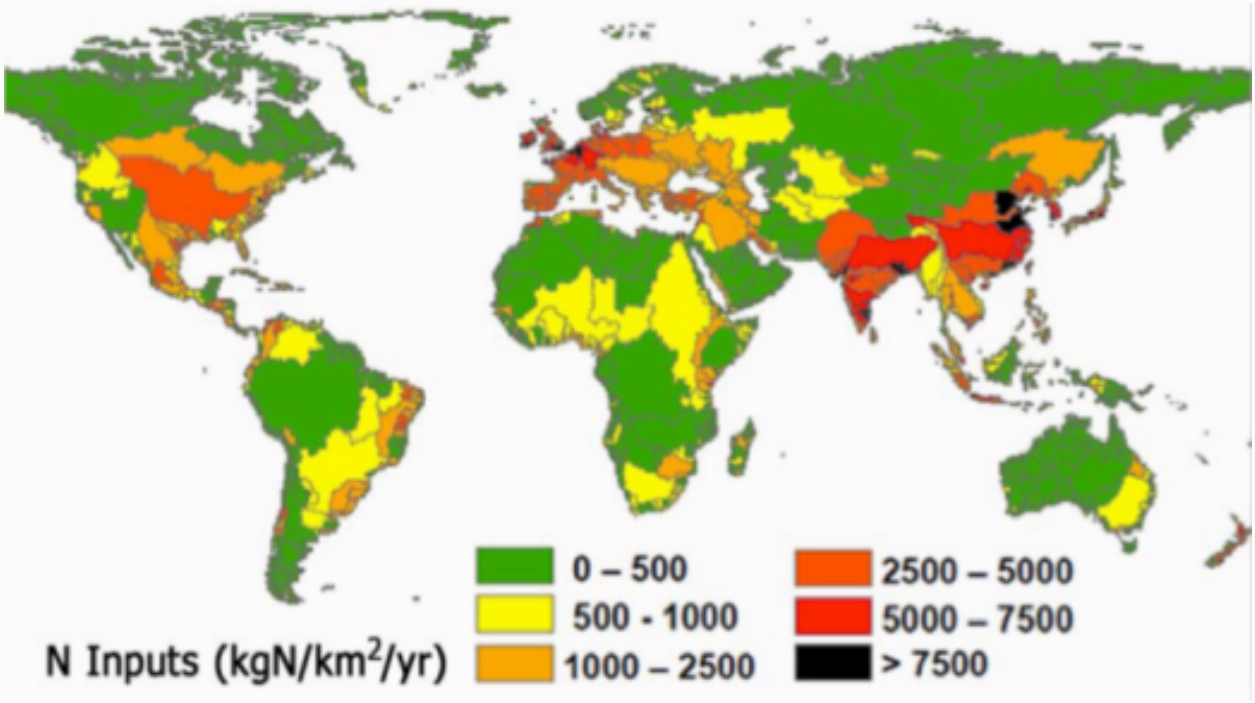


Figure 7.3-6 Estimated net anthropogenic N inputs in the world's main river catchments (1 km2=100 ha).



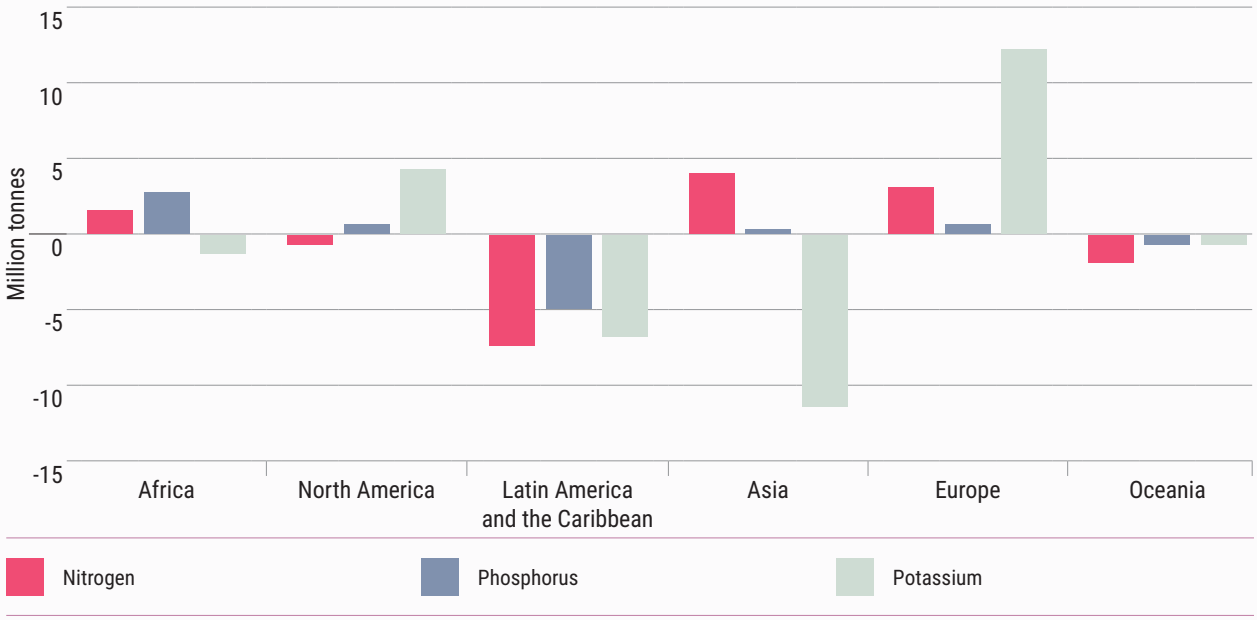
Estimates of the amounts of phosphorus and potassium supplied by animal manure globally are lacking. However, estimations of the nutrient composition of animal manure reported in the literature. Fairhurst (2012) suggest that it could also contribute substantial amounts of phosphorus and potassium to soils. Animal manure has been estimated to contribute approximately 55 per cent of phosphorus supplied by fertilizers in 2000 (Bouwman *et al*. 2013). According to a phosphorus flow chart showing global phosphorus flows for 2009 by Koppelaar and Weikard (2013), presented in Smits and Woltjer (2018), cropland and pasture may receive more phosphorus from animal manure (about 28 million tons) and other organic sources (7 million tons, mainly from crop loss, residues, food waste and human excreta) than from inorganic fertilizers (about 17 million tons). At the same time, it should be remembered that a proportion of nutrients in manure is or could originally be from inorganic sources.

The leading countries in terms of agricultural consumption of inorganic fertilizer are Brazil, China, India and the United States. About 25 per cent of nitrogen, phosphorus and potassium inorganic fertilizer produced in the world was used in agricultural production in China in 2016 (FAO 2020a). In 2017 the countries with the largest areas of cropland, in declining order, were India, the United States, China, the Russian Federation and Brazil (FAO 2020d). In the same year, among countries with more than 0.5 million ha of cropland, those with the highest rate of nitrogen from inorganic fertilizers compared to their area of cropland included Egypt (342 kg N/ha), China (219 kg N/ha), the Netherlands (209 kg N/ha), Belgium (198 kg N/ha) and the United Kingdom (170 kg N/ha) (FAO 2020d; FAO 2020e)[2].

The amount of manure N applied to cropland was estimated by dividing FAOSTAT data on "total amount of N in the manure applied to soils in a country" (FAO 2020b) by "area under cropland in a country" (FAO 2020d). Among countries with at least 0.5 million ha of cropland, those with the resulting highest rates of nitrogen from

2 The cut-off of 0.5 million ha of cropland is an arbitrary value used to identify countries with the potential to make substantial contributions to global pollution from nitrogen use.

Figure 7.3-7 **Fertilizer exports minus imports, for nitrogen (N), phosphorus (P_2O_5) and potassium (K_2O), in 2017 (Mt).** FAO (2020a).



manure were the Netherlands (284 kg/ha), Belgium (161 kg/ha), New Zealand (126 kg/ha), Mongolia (122 kg/ha) and the United Kingdom (83 kg/ha).

Estimates of the amount of nitrogen, summed up for both inorganic and organic fertilizers, applied to soil was least in sub-Saharan countries (less than 50 kg/ha/year applied in many countries). Nitrogen inputs from anthropogenic sources are estimated to be highest in Europe, North America, parts of South and East Asia, and Latin America (Sutton *et al*. 2013) (Figure 7.3-6).

7.3.3 Fertilizer exports and imports

Between 2014 and 2016 the leading net exporting regions were Asia and Europe (including the EECA countries) for nitrogen, Africa for phosphorus, and Europe (including the EECA countries) and North America for potassium (Figure 7.3-7). The leading net importing regions were Latin America and the Caribbean (LAC) for nitrogen and phosphorus and Asia and LAC for potassium.

Global sales revenues for inorganic fertilizers in 2018 were reported to be about USD 151 billion; the global fertilizer market has been projected to grow at a compound annual growth rate (CAGR) of 3.8% in the period 2020-2025 (Mordor Intelligence 2020, as cited by Ilinova, Dmitrieva and Kraslawski 2021) and at a CAGR of 2.1 per cent between 2021 and 2026 (Mordor Intelligence 2021b).

There is not much information available on trade in organic fertilizers. However, indications are that the contribution of organic fertilizers to the fertilizer market is relatively small. For example, it was estimated that global sales of organic fertilizers would be about USD 6.52 billion by the end of 2019 (Bloomberg Business 2019). According to this Bloomberg Business report, the EU represents a large proportion of the global organic fertilizer market (estimated to be about 38 per cent in 2019).

7.4 Fertilizer distribution and sales mechanisms

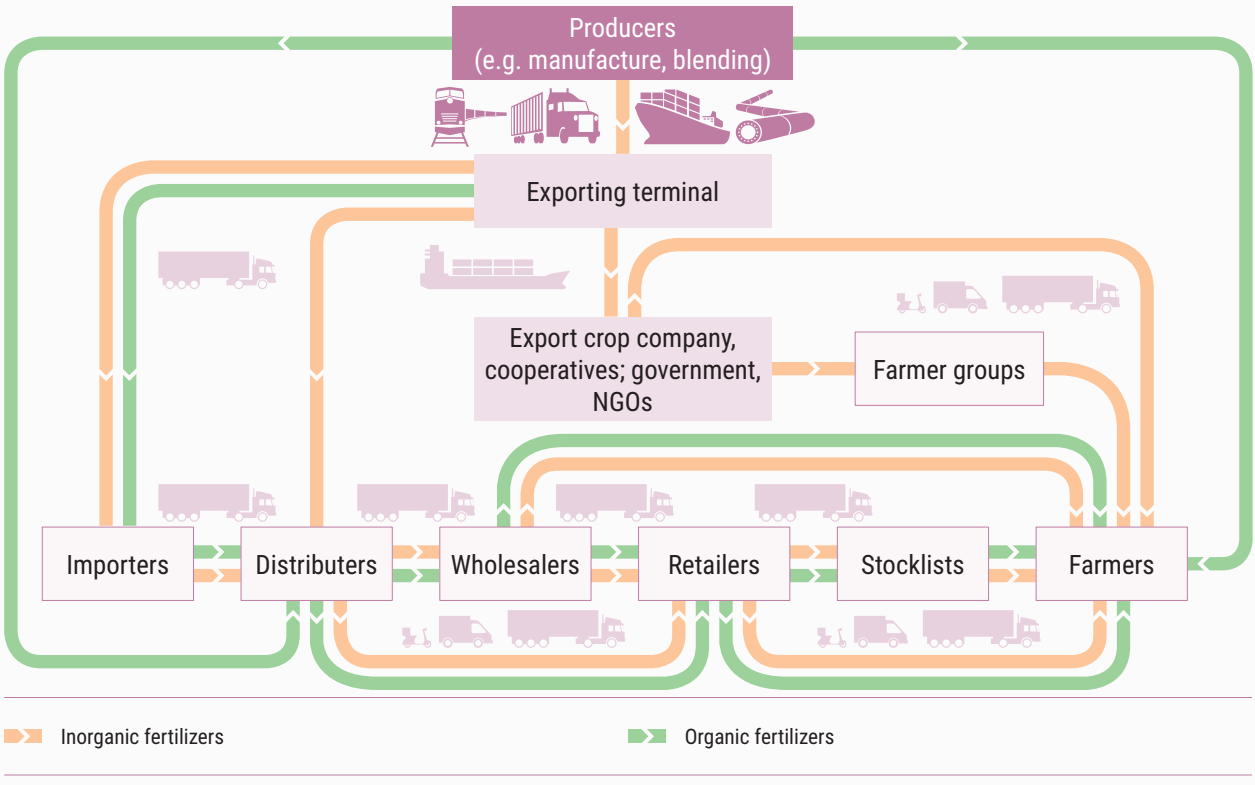
7.4.1 Distribution and marketing

In general, the organic fertilizer supply chain can be shorter than that of inorganic fertilizers, for example where farmers recycle products generated on their own farms or purchase products such as manure from neighbouring farmers.

The fertilizer distribution chain starts with manufacturers/producers (global, regional or national), followed by importers which operate at regional or national level, wholesalers and retailers which operate at national level, and finally farmers who are the fertilizer end users (Figure 7.4-1). Importing can be carried out by private sector importers or government ministries (Hernandez *et al*. 2018). The supply chain can be simple. For example, cooperatives purchase fertilizers to be used by members (e.g., on export crops like tea and coffee) from fertilizer producers (AfricaFertilizer 2012). Governments can purchase fertilizers from producers and importers for distribution to farmers (Hernandez *et al*. 2018).

In high income countries fertilizers are often distributed in bulk or in large bags (500-1,000 kg). In low income countries importers often bag fertilizers before they are transported to distributers. Sometimes the fertilizers are repackaged into smaller quantities and information about them is provided through retailers or stocklists. In Africa some fertilizer producers have blending plants or are investing substantially in building such plants (Diallo *et al*. 2019). Blending plants produce customized fertilizers adapted to local conditions and requirements (Tsujimoto *et al*. 2019). The popularity of blends, which are cheaper to produce than compound fertilizers, has been increasing in sub-Saharan Africa (Mutegi, Kiwia and Zingore 2019).

Figure 7.4-1 Simplified diagram of the fertilizer supply chain. The orange and green lines represent inorganic and organic fertilizers, respectively. AfricaFertlizer (2012) and Li *et al*. (2013).



7.4.2 Organization of the fertilizer industry

Globally the inorganic fertilizer industry is largely represented by the (IFA 2021b). A few IFA members are also involved in organic and organo-inorganic fertilizer production. IFA members represent different sectors of the fertilizer industry. In 2020 there were over 400 members in 70 countries. At regional level there are organizations such as Fertilizers Europe, the Arab Fertilizer Association (AFA), The Fertilizer Institute (TFI) in the United States, Fertilizer Canada, the Fertilizer Association of India (FAI), the Associação Nacional para Difusão de Adubos (ANDA), Fertilizer Australia, and the West Africa Fertilizer Association.

The organic fertilizer industry does not have a global association. In Europe national associations including Acteurs d'une terre plus verte (Afaïa) and L'Union des industries de la fertilisation (UNIFA) in France, BELFORM in Belgium, FOMA in Spain, Assofertilizzanti in Italy and the BMA in the Netherlands cover refined organic fertilizers. The European Consortium of the Organic-Based Fertilizer Industry (ECOFI) provides a representative voice for manufacturers. The lack of a global association for the organic fertilizer industry could be due in part to its traditionally local nature (with regard to both raw materials and markets) and the fact that most producers are small and medium-sized enterprises.

7.4.3 Transportation

Transportation from producers to farmers is mostly carried out by a combination of transport modes, for example by sea from the country of origin to a port close to the importing country, by rail from the port to distributors, and by road from distributers to retailers.

In the United States railroads and waterborne vessels often move fertilizers from a production facility to and throughout the country to warehouses, terminals and retailers (TFI 2016). These forms of transport are especially important for long distance transportation, while roads may be important for short distances, particularly as the product gets closer to farmer customers (TFI 2016). A large share of the fertilizers used in the Corn Belt is shipped on barges on the Mississippi River (TFI 2016; Plume 2019).

In sub-Saharan Africa a large portion of retail costs are for transport, as fertilizer is mostly carried by road within countries and through the continent (Benson and Mogues 2018). According to Gro Intelligence (2016) and Wanzala and Groot (2013), transport costs are a major reason for high fertilizer prices in sub-Saharan Africa. Fertilizer prices there are reported to be the highest in the world, e.g., four times those in Europe (Hernandez and Torero 2018). Data presented in a World Bank report on trade and transport in developing countries (World Bank 2014) suggest that rail transport is cheaper than road transport.

7.4.4 Landlocked countries

Distribution costs tend to be high in landlocked countries, partly due to additional transport costs. For example, in 2007 about 32 per cent of retail prices of fertilizer in landlocked Mali and 22 per cent of those in Tanzania (which has a coastline) were attributed to these costs (Wanzala and Groot 2013). In sub-Saharan Africa the highest urea prices have been reported in three landlocked countries (Cedrez *et al*. 2020). Landlocked countries also have to maintain relatively higher stocks of imported commodities than other countries, incurring greater costs (World Bank 2014).

7.4.5 Commercial sales and subsidies

Subsidies can contribute to a reduction of the quantity of a commercial fertilizer purchased and crowd out the private sector (Mather and Jayne 2018). A major objective of current "smart subsidies" is to boost sustainable development of input markets (Tiba 2011).

Additional information on fertilizer subsidies can be found in Chapter 8.

7.4.6 Fertilizer quality and adulteration

The literature on the status of fertilizer quality and adulteration, published in indexed journals, is minimal or lacking. Hence, the information

presented here is based on reports from FAO and IFDC.

A 2011 FAO report stated that 52 per cent of 5,053 fertilizer samples tested in South Asia were substandard (FAO 2011). An assessment study carried out by IFDC in five countries in West Africa in 2010, on samples collected from retailers, found that blends were more likely to be of poor quality than compound fertilizers (12-96 per cent of blended fertilizers vs. 1-16 per cent of compound fertilizers) and that some fertilizers had severe granule degradation. (This report assumed that "agro-dealers", the term used in IFDC reports, were synonymous with "retailers"); poor quality was mainly attributed to uneven distribution of nutrients in bags due to segregation of granules, as well as to poor handling and storage (Sanabria, Dimithe and Algnikou 2013). More recent IFDC studies, on samples taken from retailers shops in Kenya and Tanzania, indicated that some of the fertilizer sold had less nutrient content than indicated on the bags and that some bags could be underweight, although fertilizer was not adulterated (Sanabria *et al*. 2018a; Sanabria *et al*. 2018b).

Information of existing fertilizer regulations and policies is presented in Chapter 8.

7.4.7 Cross-border trade

In sub-Saharan Africa cross-border trade of fertilizer is limited by poor infrastructure, weak economic integration, and delays in crossing borders due to inefficient custom procedures and document requirements (United Nations Economic Commission for Africa and African Development Bank [UNECA and AfDB] 2018). Trading fertilizer across borders largely takes place through formal markets, but informal trading also exists (UNECA and AfDB 2018).

7.5 Drivers of fertilizer use

Drivers of fertilizer use exist at farm level or outside the farm. They can be technological (e.g., fertilizer use efficiency), associated with consumers (e.g., demand for food and food choices), or can even be associated with sustainability aims (e.g., environmental policies). Some factors can encourage increased use of fertilizers and others can discourage it, while still others may have either effect depending on the circumstances.

7.5.1 Factors influencing fertilizer usage: examples

Demand for food (in terms of quantity and quality) and income

Global demand for food continues to grow with the increasing population (Bodirsky *et al*. 2015). Unlike in the past, when growth in agricultural production was largely due to increases in the amount of cropped land, expansion of agricultural area is no longer feasible in many regions and higher production is largely due to agricultural intensification. According to FAO data, between 2002 and 2018 the global population increased by about 21 per cent (FAO 2020g) and cereal production increased by 44 per cent, but the total area harvested for cereals increased by only about 10 per cent (FAO 2020f). The use of fertilizers has been a key component of intensification (Cassman 1999). For example, small-scale farmers in Asia and Latin America increased production dramatically through use of improved seed, greater use of inorganic fertilizers, and good management practices (Voortman, Sonneveld and Keyzer 2003).

Fertilizers may improve the quality of the edible crop part (e.g as a biofortification strategy). Selenium (Se) applied to the soil or to crops can improve crop Se levels, thereby contributing to improved nutrition for human beings and animals (Ros *et al*. 2016).

Policies, subsidies and government support

Developing countries in which there are high levels of hunger tend to have the lowest levels of

inorganic fertilizer use. For example, according to the World Bank (2019), in 2018 the prevalence of undernourishment (per cent population) averaged 19 per cent and 18 per cent in least developed countries and in SSA, respectively, but was 3 per cent in Europe and North America. Improving farmers' access to and lowering the costs of inputs such as high-quality seeds, fertilizers and pesticides can contribute to achieving Sustainable Development Goal (SDG) 2 (End hunger, achieve food security and improved nutrition, and promote sustainable agriculture) (United Nations Conference on Trade and Development 2018). Agricultural growth in China has been linked to policies that support fertilizer production and use (Li, Y. *et al*. 2013).

Policies that promote recycling and include standards for fertilizers encourage production by the fertilizer industry of quality fertilizer products from waste and manures. For example, the 2019 EU regulation states that "Promoting increased use of recycled nutrients would further aid the development of the circular economy and allow a more resource-efficient general use of nutrients…" (EU 2019, p. L 170/1). Policies that promote a "clean environment" can also encourage nutrient recycling. In a study carried out in four EU countries (Hou *et al*. 2018), pressure from government regulations was perceived as a key factor stimulating manure treatment. However, some countries lack policies that encourage access to high-quality inorganic and organic fertilizers (as discussed in greater detail in Chapter 8) or nutrient recycling.

Input subsidies can contribute to greater use of fertilizers (Holden 2018), as well as to increasing yields and national production (Jayne *et al*. 2018). They may also have drawbacks. In China subsidies contributed to an adequate supply of affordable fertilizer, which in turn led to fertilizer overuse (Li *et al*. 2013. In India, through heavy subsidization the price of urea has been kept low compared with the prices of phosphorus and potassium fertilizers, leading to unbalanced use of nutrients (overuse of nitrogen) and soil and other types of environmental degradation (Huang, Gulati and Gregory 2017). Ndambi *et al*. (2019) have pointed out that subsidization of the cost of synthetic fertilizers in sub-Saharan Africa could discourage use of manure as fertilizer if there are not incentives for manure use.

Soil degradation

Soil degradation involves deterioration of the soil's physical, chemical and biological properties and is a reason for reduced production. Degradation processes include soil erosion, decline of soil organic carbon, compaction, salinization (accumulation of water-soluble salts), sodification (accumulation of sodium), contamination, and loss of soil biodiversity (Louwagie, Gay and Burrell 2009; Govers *et al*. 2013; Bach *et al*. 2020; Baveye *et al*. 2020).

Overuse of inorganic fertilizer can contribute to desertification and land degradation through changes in the chemical properties that cause soil acidification and reduction of soil organism biodiversity, which may further change soil structure through increased soil compaction and decreased water and air retention capacity. Overuse of inorganic and organic fertilizers can also increase chemical accumulation in soil water and affect water cycling. For example, over-application of nitrogen fertilizers can reduce soil pH, resulting in soil degradation (Kopittke *et al*. 2019), and increase greenhouse gas (GHG) emissions. On the other hand, use of low amounts of fertilizers has been partially associated with soil degradation, e.g., in sub-Saharan Africa (Zingore *et al*. 2015).

Judicious use of fertilizers enhances soil organic matter, reduces conversion of natural land to cropland, and minimizes the risks of additional land degradation. Practices addressing soil degradation include use of organic fertilizers, localized fertilizer application, intercropping (e.g., growing both cereals and legumes), conservation agriculture, ridge tillage, contour farming, terracing (Louwagie, Gay and Burrell 2009) and precision conservation (Delgado, Barrera *et al*. 2019; Delgado, Short *et al*. 2019; Delgado, Sassenrath and Mueller 2020).

Several studies have been carried out on the relationship between fertilizers and soil carbon. Organic fertilizers remediate degraded soils by maintaining and enhancing soil organic matter,

improving their physical fertility, and supplying nutrients to crops (Medina *et al*. 2015). In a review of organic inputs and fertilizers by Chivenge, Vanlauwe and Six (2011) soil carbon was associated with increased use of organic inputs; yields were high when inorganic fertilizers were used, and highest when organic inputs were used in combination with inorganic fertilizer. A review by Gram *et al*. (2020) reported that increases in soil carbon were only significant in the case of organic inputs and were more pronounced when these inputs were of high quality. Fertilizer use contributes to an increase in biomass, while incorporation of crop residues in the soil can contribute to increased soil carbon (Hijbeek, van Loon and van Ittersum 2019b). The authors of these three publications point out that using a combination of inorganic and organic fertilizers can be beneficial with respect to both yields and soil carbon. Many other authors (e.g., Bharari *et al*. 2017 and Kumari *et al*. 2019) have drawn similar conclusions.

Fertilizer use efficiency

High fertilizer use efficiency is linked to increased yields, which encourages farmers to invest in fertilizers. Nevertheless, Fixen *et al*. (2015) suggests that a substantial proportion of the fertilizer applied to the soil is lost and not taken up by crop.

There are several ways to estimate nutrient use efficiencies (Fixen *et al*. 2015). In this section "nutrient use efficiency" (or "efficiency") refers to the percentage of fertilizer nutrient recovered in the aboveground plant biomass used (i.e., for a nutrient, the amount of nutrient taken up by the fertilized crop minus the amount taken up by the unfertilized crop, divided by the amount of nutrient applied). In cereals the proportion of applied nitrogen taken up by plants in farmers' fields in the first year ranged from 30-50 per cent in one study (Cassman, Dobermann and Walters 2002). According to Fixen *et al*. (2015), efficiencies in well-managed cereal fields are 40-65 per cent for nitrogen, 15-25 per cent for phosphorus and 30-50 per cent for potassium during the first year of application.

After the first season or year, some nutrients remain in the soil and can be taken up by crops in subsequent seasons or years. For example, according to Syers, Johnson and Curtin (2008) it is possible that in some cases a relatively large amount of phosphorus from fertilizer (up to 90 per cent) is taken up by crops in the long term. Dobermann and Cassman (2005) estimated that to meet an anticipated 38 per cent increase in global cereal demand by 2025 a 60 per cent increase in nitrogen use on cereals might be required, partly due to large nutrient losses.

Fertilizer use efficiency is improved by using nutrient-saving technologies such as integrated soil fertility management and precision agriculture. For example, in the Netherlands the level of nitrogen applied in recent years has been reduced to the same level as in the 1960s while yields have doubled (Lassaletta *et al*. 2014).

Improving fertilizer use efficiency may not be enough to meet environmental goals. According to a recent study on nitrogen use efficiency (NUE) in the EU-27 (de Vries and Schulte-Uebbing 2020), in some regions protecting surface water quality would require increasing NUE (to about 72 per cent) while maintaining current levels of nitrogen inputs, whereas in other regions it would require increasing NUE and reducing nitrogen inputs.

Integrated soil fertility management and other approaches

Integrated soil fertility management has been defined as a "set of soil fertility management practices that necessarily include the use of fertilizer, organic inputs, and improved germplasm combined with the knowledge on how to adapt these practices to local conditions, aiming at maximizing agronomic use efficiency of the applied nutrients and improving crop productivity. All inputs need to be managed following sound agronomic principles" (Vanlauwe *et al*. 2010). Integrated soil fertility can be used to describe the package of technologies and practices that contribute to improved fertilizer use efficiency, soil health and crop productivity.

Nutrient management practices that reduce nutrient losses include using the right nutrient source and applying the fertilizer at the right rate, the right place and the right time (the 4R approach to nutrient stewardship) (The Fertilizer Institute [TFI] 2017; IFA 2020g) (see Chapter 7.2). The 4R approach contributes to improved fertilizer use efficiency. There is a need to optimize fertilizers for crop mixtures and rotations. For example, in cereal-legume systems the cereal can benefit from the residual effects of the legume (Franke *et al*. 2018) and may therefore need less fertilizer nitrogen.

Some technologies are promoted in packages that include fertilizer use. For example, recommendations for some maize varieties include use of inorganic fertilizers to achieve good yields (Nyagnena and Juma 2014). Some countries provide subsidies for inputs. As reported in a review by Hemming *et al*. (2018), farmers in India were offered subsidized agricultural inputs in the form of mini-kits containing seeds for rice, oilseeds and potatoes, fertilizers and pesticides, and farmers in Mozambique were offered voucher subsidies for an improved maize seed and fertilizer package. Agronomic practices also influence crop demand for fertilizers. For example, fertilizer demand can increase with the size of plant populations (Yang and Fang 2015) and placing the fertilizer close to seed, instead of broadcasting, can increase fertilizer uptake and yields (Nkebiwe *et al*. 2016).

Other examples of soil fertility management options include:

- Improving fertilizer use efficiency by growing cover crops, applying nitrogen fertilizer in splits, using slow-release fertilizers, precision agriculture (see below), and site-specific nutrient management (SSNM) options such as the use of leaf colour charts to match nitrogen with crop demand (Fairhurst 2012).

- Nitrogen leaching which can be reduced by growing cover crops (Abdalla *et al*. 2019) and including crops with deeper root systems in rotations (Delgado *et al*. 2007). Incorporating biochar in soil can reduce both N leaching (Feng *et al*. 2019) and N_2O emissions (Li, Y. *et al*. 2018). However, the costs can be prohibitive for farmers, especially in the short term (Spokas *et al*. 2012; Dickinson *et al*. 2015).

- Using liming materials, for example limestone (calcium carbonate, $CaCO_3$), limestone which contains magnesium carbonate ($CaCO_3$ and $MgCO_3$), and gypsum ($CaSO_4 \cdot 2H_2O$). These are often used as soil amendments (also referred to as "soil conditioners") (Fairhurst 2012). When used to correct the pH of acid soils (Fairhurst 2012), these materials can contribute to increased phosphorus recovery (Kisinyo *et al*. 2014). However, they may increase CO_2 releases to the atmosphere from inorganic carbon (Zamanian and Kuzyakov 2019). Gypsum is used to rehabilitate sodic soils, while each of these amendments is used to supply calcium to plants (Fairhurst 2012).

Precision agriculture

Precision agriculture (or precision farming) improves the targeting of nutrients to plant requirements, enabling farmers to make agricultural management decisions that consider fields' heterogeneity (Finger *et al*. 2019). It reduces nutrient losses, hence contributing to greater economic benefits from fertilizers (Bhakta, Phadikar and Majumer 2019). For example, ammonia emissions were less with precision application of organic inputs than with broadcasting (Nicholson *et al*. 2018) and nitrate leaching losses were lower when remote sensing and management zones were used than with traditional practices (Delgado *et al*. 2005). Luther, Swinton and Deynze (2020) found that in the United States objectives other than income were important drivers for the adoption of conservation and precision technologies; for example, farmers who participated in "working lands" environmental stewardship programmes were considered more likely to adopt both cover cropping and precision soil testing.

The types of technologies adopted can differ with the level of operations. On smallholder farms in West Africa, for example, Aune, Coulibaly and Giller (2017) proposed using good quality seeds primed and treated with a mixture of pesticides and fungicides; low doses of inorganic

fertilizers; accurate distribution of seeds and fertilizers; and mechanized sowing and weeding. Large-scale farmers are likely to adopt more advanced technologies (Carli, Xhakollari and Tagliaventi 2017). A review of state-of-the-art precision technologies by Bhakta, Phadikar and Majumder (2019) lists global positioning systems (GPS), remote sensing, wireless on-the go sensors and yield monitors among technologies used for data collection; geospatial tools, soft computing and modern software among the technologies used for data analysis; and variable rate fertigation and variable rate pesticides as means of applying variable rates of inputs. These high-end technologies are probably most suitable for large-scale farmers with high financial capacity, but tools that can be used by smallholder farmers also exist (see below).

Decision support tools for fertilizer recommendations

In view of the need to improve fertilizer efficiency and reduce pollution, fertilizer recommendations have increasingly taken spatial and temporal heterogeneity between and within fields into consideration. Approaches for developing fertilizer recommendations and decision support tools have therefore shifted towards cloud-based tools and site-specific nutrient management (SSNM) based on scientific principles (Ahmad and Mahdi 2018).

Tools like the web-based Nutrient Manager for Rice (NMR) (Bado, Dhaman and Mel 2018), the Nutrient Expert (NE) (Rurinda *et al*. 2020), and the Corn N Calculator (CNC) and Adapt-N (Sela *et al*. 2017) have increased the efficiency with which fertilizer recommendations are developed. In comparison with recommendations based on soil tests, for example, the NE approach has been shown to contribute to the improvement of grain yield, nutrient uptake and fertilizer use efficiency (Yang *et al*. 2017). Sela *et al*. (2017) compared the Corn N Calculator (CNC), a static nitrogen recommendation tool, and Adapt-N, a tool that combines soil, crop, and management information with real-time weather data to estimate optimum N application rates for maize. They found that the Adapt-N tool contributed to increased farmer profits and reduced N application rates in comparison with the CNC, which in turn resulted in substantially lower simulated nitrogen losses to the environment.

Use of fertilizer recommendation tools in association with tools for other inputs and practices, for example RIDEV in rice production, enhances fertilizer use efficiency even more. RIDEV can be used to determine optimum timing of nitrogen fertilizer application and the timing of drainage and harvesting (Wopereis *et al*. 2003). More recently, RiceAdvice has been developed for rice production systems in Africa. It is ready for use or being tested in some countries (RiceAdvice 2019). However, as demonstrated by Cotter *et al*. (2020), it may need data for adaptation to new environments. A key characteristic of these tools is their emphasis on splitting nitrogen for reduced N losses to the environment, better yields, and improved fertilizer use efficiency. In a study using wheat as a test crop, Belete *et al*. (2018) showed that by splitting nitrogen into three doses (one-quarter at sowing, one-half at tillering, and the remaining one-quarter at booting) yields and nitrogen recovery could be improved significantly. When the results of such tools are combined with the application of nutrients in accordance with 4R nutrient stewardship, the effectiveness of fertilizer recommendations on crop yields and the reduction of nutrient releases to the environment is increased.

The adoption of some decision support tools could be constrained by their complexity. In Australia, where most farmers learn about nitrogen fertilizer requirements from commercial crop advisers, these advisers preferred simple decision support systems rather than decision support tools requiring detailed inputs and soil characterization (Schwenke *et al*. 2019). Furthermore, accessing and interpreting data can be a challenge (Weersink *et al*. 2018; Trendov, Varas, and Zeng 2019). The existence of suitable open-access databases and access to internet with sufficient bandwidth for transmission of data can contribute to the increased use of such tools (Delgado, Short *et al*. 2019). Their adoption can also be encouraged by the technical capacities of the research teams and end-users (Bado, Dhaman and Mel 2018).

Fertilizer quality

Awareness that fertilizers could be of poor quality may encourage farmers to exceed recommended rates. For example, the perception that the quality of inorganic fertilizers might be inferior to that of organic ones contributed to increased fertilization rates in China (Yang and Fang 2015). In the case of organic fertilizers, uncertainty about nutrient content can be a barrier to use (Case *et al*. 2017)

According to farmers surveyed in Denmark, the most important barriers to the use of animal manure are unpleasant odour for neighbours, uncertainty about nutrient content, and difficulty in planning and use (Case *et al*. 2017). Concerns about transmission of pathogens and the resulting sanitary requirements (e.g., as evidenced by organic fertilizer regulations) can create additional obstacles to use of organic fertilizers. Hence, the availability of processed organic inputs could encourage use of organic fertilizers. Processing has the advantage of providing inputs with reliable nutrient concentrations and reduced concentrations of contaminants. Bulkiness may be a constraint on the use of unprocessed organic fertilizers. Processing can reduce moisture content, thereby encouraging the use of such fertilizers (Mehta *et al*. 2015).

Farm and household characteristics and objectives

Farm and household characteristics that influence intensification include household income, age and gender of the head of the household, household size, farm size, availability and affordability of inputs, access to equipment and knowledge of technologies (Fairhurst 2012), farmers' engagement in off-farm economic activities, contact with agricultural extension services, and experience in agriculture (Ali, Awumi and Danso-Abbeam 2018). In sub-Saharan Africa male-headed households were reported to be more likely to use modern inputs than female-headed households (Sheahan and Barrett 2017); in addition, young farmers were more likely to invest in intensification than old and poor ones (Wairegi *et al*. 2018). Increased farm size has been associated with a decrease in fertilizer application rates (Wu, Xi *et al*. 2018). However, there seems to be little or no evidence directly linking farmer's margins and their willingness to invest in/adopt technologies and practices that improve fertilizer use efficiency.

Supply (availability of inputs, markets included) and infrastructure

Constraints on inorganic fertilizer supply are partially associated with manufacturing, the enabling environment (Ariga *et al*. 2019), importation, distribution and pricing (Mwangi 1996). In sub-Saharan Africa, where fertilizer prices are among the highest in the world, these prices have largely been attributed to poor infrastructure (e.g., ports, roads, distribution networks) (Mwangi 1996).

The costs of inputs increase with greater distance to input markets (Aggrarwal *et al*. 2018), while farm gate prices fall with greater distance to markets for cultivated products. Proximity to markets has been shown to have a significant positive correlation with fertilizer adoption in Brazil (Morello *et al*. 2018).

Inorganic fertilizers may not be available, may not be delivered on time, or may be packaged in large quantities (which can be a constraint for poor farmers) (Mwangi 1996; Dersseh *et al*. 2016). An adequate supply of inorganic fertilizers that match crop recommendations is lacking in some countries (Stewart *et al*. 2020).

Constraints on the supply of organic fertilizers include lack of sufficient amounts, competition between uses (Nhamo, Kintche and Chikoye 2017), and high transport costs due to weight, bulk and distance from the (Akram *et al*. 2019).

According to a mapping study by Powers *et al*. (2019), the intensification of livestock production and high population densities present opportunities for phosphorus recovery. However, use of the recovered nutrients for crop production can be hampered by distances between croplands and livestock production areas, and between croplands and densely populated areas. The authors showed that there are areas where croplands occur next to areas with dense human populations or close to dense animal populations, creating opportunities for phosphorus recycling.

However, there are cases in which the amounts of organic products exceed demand although they are still spread on soils, not because of their value as fertilizer but just in order to get rid of them. For example, in Europe, where there is spatial separation of livestock systems from cropping systems due to intensification, livestock producers have excessive amounts of manure compared with available land (EIP-AGRI 2017). Hence, in some of these areas excess manure and slurries have contributed to significant pollution of water.

Consumer preferences for organic products

Organic agriculture has been defined as "a production system that sustains the health of soils, ecosystems and people. It relies on ecological processes, biodiversity and cycles adapted to local conditions, rather than the use of inputs with adverse effects. Organic agriculture combines tradition, innovation and science to benefit the shared environment and promote fair relationships and a good quality of life for all involved" (IFOAM – Organics International [IFOAM] n.d.a) Organic farmers are encouraged to use organic inputs such as manure and crop residues, and to adopt practices that mitigate environmental pollution (Sustainable Organic Agriculture Action Network 2013).

Products from organic agriculture are sometimes labelled "organic" by a certification body if they have been produced, stored, processed, handled and marketed in accordance with that body's precise technical specifications (standards) (IFOAM n.d. b). The FAO/WHO Codex Alimentarius Commission has produced Codex Alimentarius guidelines for organically produced foods to guide producers and protect consumers (FAO and WHO 2001).

While the production and marketing of organic products are growing, the area of land under organic agriculture remains relatively small. According to survey results based on available data and presented in Willer *et al*. (2020), the market for organic foods is estimated to be worth some USD 105.5 billion per year (having grown from almost nothing in the 1990s) while about 71.5 million ha was estimated to be under organic agriculture in 2018 (FAO 2020d).

Access to finance (credit)

Use of inputs is also influenced by access to credit (Sheahan and Barrett 2017). Credit-constrained farmers are less likely to purchase inorganic fertilizers (McIntosh, Sarris and Papadoupoulos 2013) and more likely to adopt the use of manure (Kassie *et al*. 2015). For farmers in sub-Saharan Africa, credit is attached to strict collateral and high costs, as annual interest rates are around 30 per cent (Ariga *et al*. 2019).

Access to information, social networks, and membership in cooperatives

Training of extension staff and farmers is necessary for increased adoption of new technologies and practices to achieve sustainable nutrient management. Access to information can contribute to either increased or reduced use of fertilizers. For example, in Ethiopia visits to farmers by extension staff were associated with increased use of fertilizer (Tigabu and Gebeyehu 2018) and in China provision of guidance to farmers in the field was associated with reduced use of excessive amounts (Pan *et al*. 2017).

Social networks are useful for exchange of knowledge between farmers, such as on improved farming practices. Social networks, for example being in a group or knowing fertilizer traders, can act as drivers of intensification by improving the flow of information and access to credit, and subsequently adoption of intensification practices (e.g., fertilizer use) (Kassie *et al*. 2015). Farmers are more likely to adopt technologies if other farmers in their social networks have also adopted them. A study by Wang *et al*. (2018) concluded that membership in agricultural cooperatives can encourage farmers to use organic instead of chemical fertilizers. In coffee cooperatives which consolidate coffee from farmers for sale, farmers may be pressured by other members and extension agents to improve the quality of their berries through intensification for better coffee prices (Bennett *et al*. 2016). At the same time, in such cooperatives farmers with better quality coffee are not rewarded, so that some farmers may be unwilling to invest in inputs (e.g., in use of fertilizers).

Information and communications technology (ICT) is increasingly being used to disseminate new information on agriculture. Even in developing countries, transmission of agricultural information through mobile technologies has been associated with increased adoption of recommended practices (Fabregas, Kremer and Schilbach 2019). Some are less likely to own ICT tools due to lack of resources (Gumucio *et al*. 2018). They are also likely to have less access to information about technologies and to extension services (Ragasa 2012).

Climate change

Fertilizer use has been associated with GHG emissions either directly (e.g., through nitrogen fertilization, liming, and methane emissions from cropping and livestock activities) or during production and transportation (see Chapter 9). Furthermore, climate change and extreme weather events will likely increase the potential for nutrient losses through erosion and leaching (Lal *et al*. 2011). On the other hand, the Green Revolution, in which fertilizer played a large part (Pingali 2012), (see Chapter 10).

Climate change can influence fertilizer consumption. Using a model, Rurinda *et al*. (2015) predicted that the yield decline expected due to climate change could be reduced through use of fertilizers, but the response to fertilizers would decline with climate change. According to a review by Olesen and Bindi (2002), climate change may or may not lead to increased demand for fertilizers in agricultural production in Europe. The use of real time nutrient management strategies that respond to variations in climate events can encourage fertilizer use.

Organic fertilizer industry associations

When the inorganic fertilizer supply side at national and regional levels is under an umbrella association, the fertilizer market is strengthened and effective dialogue with governments or with regional governmental organizations in order to promote trade is made possible (Ariga *et al*. 2019). With regard to the organic supply side, where they are mostly absent, the formation of such associations may need intervention from governments.

7.5.2 Reasons for decreased fertilizer use: examples

Health and environmental concerns, including policies and regulations

Global objectives such as those set out in the Sustainable Development Goals (SDGs) (e.g., SDG 3 on Good Health and Wellbeing, SDG 6 on Clean Water and Sanitation, and SDG 14 on Life Below) are a strong indication of intentions to reduce pollution from nutrients and other sources. Many countries have aligned (or are in the process of aligning) their national agendas to the 2030 Agenda for Sustainable Development through actions, policies, laws and programmes (United Nations Environment Programme and Forum for Law, Environment, Development and Governance 2018). For example, enforcement of the EU Nitrates and National Emission Ceilings Directives reduced fertilizer use in Europe (van Grinsven *et al*. 2015). International and regional partnerships can also align their agenda with the United Nations' 2030 Agenda for Sustainable Development. For example, IFA has embarked on developing an Industry code of practice on nutrient stewardship to support implementation of the 2019 *International Code of Conduct for the Sustainable Use and Management of Fertilizers* (FAO 2019a) (Williams 2019).

More information on Conventions and policies influencing (or with the potential to influence) fertilizer use is presented in Chapter 8.

Adoption of approaches and technologies that improve fertilizer use efficiency

Approaches and technologies that improve fertilizer use efficiency (such as the 4R approach) can contribute to reduced fertilizer use. For example, cost saving was one reason given by farmers in the United States for using precision agriculture technology (Thompson *et al*. 2019), which suggests that this technology contributes to reduced use of inputs. In a study on urea deep placement (UDP) technology in Bangladesh,

the rate of urea fertilizer application to rice was reduced by about 50 per cent and profitability was higher when UDP technology was used (Rahman and Barmon 2015). In a study in China, reducing nitrogen fertilization rates by about 15-18 per cent combined with enhanced management practices contributed to an increase in yields (by about 11 per cent) as well a reduction of the amount of nitrogen used (by about 1.2 Mt) (Cui *et al*. 2018).

Despite the existence of technologies that improve fertilizer use efficiency, their adoption has not always been successful. For example, the traditional training approach has not contributed to a reduction in the use of nitrogen fertilizers (Huang *et al*. 2015).

Improved varieties (e.g., plant breeding, gene editing)

Yields are higher when nitrogen is applied to improved varieties than to traditional varieties (Hurley, Koo and Tesfaye 2018). The use of organic fertilizer has been positively correlated with growing improved varieties in West Africa (Kpadonou *et al*. 2015). There have been studies on the contribution of gene editing to improved nutrient use by crops (McAllister, Beatty and Good 2012). An example is a study on improving nitrogen use efficiency in barley (Han *et al*. 2016). Future study topics proposed include plant breeding to optimize the benefits of microbial biofertilizers (Trivedi *et al*. 2017). Even where such crops were grown, however, there would be a need to use fertilizers to replace nutrients lost through the removal of plant biomass from fields. Despite their potential benefits, use of these crops could pose environmental risks (Raina *et al*. 2018). The perceived risks have raised concerns among consumers (McFadden and Smyth 2019). Research efforts have sometimes been unbalanced. For example, crop breeding has resulted in high-yield crops while root biomass has remained almost unchanged, e.g., in soybeans (Li, S. *et al*. 2019) and wheat (Junaidi *et al*. 2018).

Crop mixtures and rotations

Legume-based mixed cropping and intercropping/rotation can reduce the need for external inputs of fertilizers/manures. Herridge, Peoples and Boddey (2008) estimated that in 2005 legumes (pulses and oilseeds) fixed about 21 million tons of nitrogen. A review by Crews and Peoples (2004) concluded that obtaining nitrogen from legumes is potentially more sustainable than obtaining it from industrial sources. Biological nitrogen fixation (Figure 7.2-2) by legumes and the incorporation of legume residues in the soil improve soil fertility (Morgan *et al*. 2019). Subsidies have been reported to contribute to increased adoption of legume production. For example, in Malawi the subsidized input package covers legume seeds apart from fertilizer and maize seeds, and farmers who received the subsidy were more likely to grow legumes than other farmers (Koppmair, Kassie and Quaim 2017). Other incentives suggested include policies to make legume production more profitable, a "meat tax" to encourage reduction of meat consumption and encourage consumption of legumes, and a carbon tax that rewards farmers for reducing emissions and for increased soil carbon (Kuhulman, Helming and Linderhof 2017).

Reducing food loss and wastage, dietary choices

About 25 per cent of food produced may be lost along the food chain (Houlton *et al*. 2019). In low income countries more food waste occurs on the farm, for example during storage and transportation, while in high income countries more occurs outside the farm, for example at consumer level. Reducing food losses can make more food available to consumers, contribute to reduced nutrient losses, and reduce the need for fertilizers.

Looking at nutrient pollution from the point of view of the planetary boundaries concept described by Rockström *et al*. (2009), the food system affects several planetary boundaries, including the nitrogen and phosphorus cycles. A modelling study by Springmann *et al*. (2018) which looked at food-related environmental impacts in 2010 and 2050 concluded that keeping within the planetary boundaries associated with the food system requires making dietary changes towards more plant-based diets, improving management practices and technologies, and reducing food loss and waste.

This implies that minimizing the adverse impacts of fertilizers means making changes in the food system, not just in crop and livestock production. Thus, there is a need to include measures that encourage dietary changes and contribute to a more efficient "from-farm-to-fork" food chain in sustainability efforts.

References

Abalos, D., Jeffrey, S., Sanz-Cobena, A., Guardia, G. and Vallejo, A. (2014). Meta-analysis of the effect of urease and nitrification inhibitors on crop productivity and nitrogen use efficiency. *Agriculture, Ecosystems and Environment* 189, 136-144. https://doi.org/10.1016/j.agee.2014.03.036.

Abdalla, M., Hastings, A., Cheng, K., Yue, Q., Chadwick, D., Espenberg, M. *et al*. (2019). A critical review of the impacts of cover crops on nitrogen leaching, net greenhouse gas balance and crop productivity. *Global Change Biology* 25, 2530-2543. https://doi.org/10.1111/gcb.14644.

Abdou, G., Ewusi-Mensah, N., Nouri, M., Tetteh, F.M., Safo, E.Y. and Abaidoo, R.C. (2016). Nutrient release patterns of compost and its implication on crop yield under Sahelian conditions of Niger. *Nutrient Cycling in Agroecosystems* 105, 117–128. https://doi.org/10.1007/s10705-016-9779-9.

Achari, G. and Kowshik, M. (2018). Recent developments on nanotechnology in agriculture: Plant mineral nutrition, health, and interactions with soil microflora. *Journal of Agricultural and Food Chemistry* 66(33), 8647-8661. https://doi.org/10.1021/acs.jafc.8b00691.

AfricaFertilizer (2012). *Training Manual on Fertilizer Statistics in Africa*. Draft Version. http://www.inter-reseaux.org/IMG/pdf/AfricaFertilizer-org_-_Training_Manual_on_Fertilizer_Statistics_-_english_June_2012_.pdf.

Aggarwal, S., Giera, B., Jeong, D., Robinson, J. and Spearot, A. (2018). *Market Access, Trade Costs, and Technology Adoption: Evidence from Northern Tanzania*. NBER Working Paper Series. Working Paper 25253. Cambridge, MA: National Bureau of Economic Research. http://doi.org/10.3386/w25253.

Ahmad L. and Mahdi S.S. (2018). Site-specific nutrient management. In *Satellite Farming*. Springer. 97-109. https://doi.org/10.1007/978-3-030-03448-1_7.

Akram, U., Quttineh, N., Wennergren, U., Tondersk, K. and Metson, G. (2019). Enhancing nutrient recycling from excreta to meet crop nutrient needs in Sweden – a spatial analysis. *Scientific Reports* 910264. https://doi.org/10.1038/s41598-019-46706-7.

Albadarin, A., Lewis, T. and Walker, G. (2017). Granulated polyhalite fertilizer caking propensity. *Powder Technology* 308, 193-199. https://doi.org/10.1016/j.powtec.2016.12.004.

Alewell, C., Ringeval, B., Ballabio, C., Robinson, D.A., Panagos, P. and Borrelli, P. (2020). Global phosphorus shortage will be aggravated by soil erosion. *Nature Communications* 11, 4546. https://doi.org/10.1038/s41467-020-18326-7.

Ali, E., Awumi, J. and Danso-Abbeam, G. (2018). Determinants of fertilizer adoption among smallholder cocoa farmers in the Western Region of Ghana. *Cogent Food and Agriculture* 4, 1. https://doi.org/10.1080/23311932.2018.1538589.

Ariga, J., Mabaya, E., Waithaka, M. and Wanzala-Mlobela, M. (2019). Can improved agricultural technologies spur a green revolution in Africa? A multicountry analysis of seed and fertilizer delivery systems. *Agricultural Economics* 50, 63-74. https://doi.org/10.1111/agec.12533.

Aune, J., Coulibaly, A. and Giller, K. (2017). Precision farming for increased land and labour productivity in semi-arid West Africa. A review. *Agronomy for Sustainable Development* 37, 16, http://doi.org/10.1007/s13593-017-0424-z.

Bach, E.M., Ramirez, K.S., Fraser, T.D. and Wall, D.H. (2020). Soil biodiversity integrates solutions for a sustainable future. *Sustainability* 12(7), 2662. https://www.mdpi.com/2071-1050/12/7/2662.

Bado, V., Djaman, K. and Mel, V. (2018). Developing fertilizer recommendations for rice in SubSaharan Africa, achievements and opportunities. *Paddy and Water Environment* 16, 571-586. https://doi.org/10.1007/s10333-018-0649-8.

Balogun, A., Diallo, A. and Harawa, R. (2019). Fertilizer distribution. In *Feeding Africa's Soils: Fertilizers to Support Africa's Agricultural Transformation*. Nairobi: Alliance for a Green Revolution in Africa. 67-76. https://agra.org/wp-content/uploads/2019/11/FeedingAfrica'sSoils.pdf.

Bationo, A. and Kumar, K. (2002). Phosphorus use efficiency as related to sources of P fertilizers, rainfall, soil, crop management, and genotypes in the West African semi-arid tropics. In *Food Security in Nutrient-Stressed Environments: Exploiting Plants' Genetic Capabilities*. *Developments in Plant and Soil Sciences* vol. 95. Adu-Gyamfi, J.J. (ed.). Springer. 144-154. https://doi.org/10.1007/978-94-017-1570-6_16.

Baveye, P.C., Schnee, L.S., Boivin, P., Laba, M. and Radulovich, R. (2020). Soil organic matter research and climate change: Merely re-storing carbon versus restoring soil functions. *Frontiers in Environmental Science* 8, 579904. https://doi.org/10.3389/fenvs.2020.579904.

Belete, F., Dechassa, N., Molla, A. and Tana, T. (2018). Efect of split application of diferent N rates on productivity and nitrogen use efciency of bread wheat (Triticum aestivum L.). *Agriculture and Food Security* 7, 92. https://doi.org/10.1186/s40066-018-0242-9.

Bennett, C. (2020). Phosphorus time bomb for agriculture? Myth and reality, 25 August. *Farm Journal*. https://www.agprofessional.com/article/phosphorus-time-bomb-agriculture-myth-and-reality. Accessed 30 October 2020.

Bennett, M., de los Rios, C., Himmel, M., Wairegi, L., Ainembabaz, H., Ampaire, E. *et al*. (2016). *Impacts of Certification on Organized Small Coffee Farmers in Kenya*. *Baseline Results*. International Institute of Tropical Agriculture and Committee on Sustainability Assessment. https://www.isealalliance.org/sites/default/files/resource/2019-02/ISEAL%20DIPI%20Kenya%20baseline%20study%20report.pdf.

Benson, T. and Mogues, T. (2018). Constraints in the fertilizer supply chain: Evidence for fertilizer policy development from three African countries. *Food Security* 10, 1479-1500. https://doi.org/10.1007/s12571-018-0863-7.

Bhakta, I., Phadikar, S. and Majumder, K. (2019). State-of-the-art technologies in precision agriculture: A systematic review. *Journal of the Science of Food and Agriculture* 99(11), 4878-4888. https://doi.org/10.1002/jsfa.9693.

Bharari, A., Baruah, K., Bhattacharyya, P. and Gorh, D. (2017). Integrated nutrient management in wheat grown in a northeast India soil: Impacts on soil organic carbon fractions in relation to grain yield. *Soil and Tillage Research* 168, 81-91. https://doi.org/10.1016/j.still.2016.12.001.

Bloomberg Business (2019). Organic fertilizer market is expected to remain steady, with a volume CAGR of ~6% during 2019-2029 – PMR, 16 July. *Persistence Market Research*. https://www.bloomberg.com/press-releases/2019-07-16/organic-fertilizer-market-is-expected-to-remain-steady-with-a-volume-cagr-of-6-during-2019-2029-pmr. Accessed 24 October 2020.

BMT (n.d.). Seedcake. https://www.cargohandbook.com/Seedcake. Accessed 21 October 2020.

Bodirsky, B., Rolinski , S., Biewald, A., Weindl, I., Popp, A. and Lotze-Campen, H. (2015). Global food demand scenarios for the 21st century. *PLoS ONE* 10(11), e0139201. https://doi.org/10.1371/journal.pone.0139201.

Bouwman, L., Goldewijk, K., Van Der Hoek, K., Beusen, A., Van Vuuren, D., Willems, J. *et al*. (2013). Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by livestock production over the 1900-2050 period. *Proceedings of the National Academy of Sciences of the United States of America* 110(52), 20882-20887. https://doi.org/10.1073/pnas.1012878108.

Business Wire (2019). Global fertilizer market growth, trends and forecast (2019-2024) – ResearchAndMarkets.com. https://apnews.com/6fd9b90eccc242f4a83cf9846e50e273. Accessed 20 October 2020.

Cakman, I. and Kutman, U. (2018). Agronomic biofortification of cereals with zinc: A review. *European Journal of Soil Science* 69(1), 172-180. https://doi.org/10.1111/ejss.12437.

Canadian Fertilizer Institute (2014). *Reflecting on the Past – Focused on the Future*. *Annual Report 2014*. https://fertilizercanada.ca/wp-content/uploads/2015/07/Fertilizer-Canada-Annual-Report-2014-English.pdf.

Carli, C., Xhakollari, V. and Tagliaventi, M. (2017). How to model the adoption and perception of precision agriculture technologies. In *Precision Agriculture: Technology and Economic Perspectives*. Pedersen, S.M. and and Lind, K.M. (eds.). Springer. 223-240. https://doi.org/10.1007/978-3-319-68715-5.

Case, S., Oelofse, M., Hou, Y., Oenema, O. and Jensen, L. (2017). Farmer perceptions and use of organic waste products as fertilisers – A survey study of potential benefits and barriers. *Agricultural Systems* 151, 84-95. https://doi.org/10.1016/j.agsy.2016.11.012.

Cassman, K. (1999). Ecological intensification of cereal production systems: Yield potential, soil quality, and precision agriculture. *Proceedings of the National Academy of Sciences of the United States of America* 96(11), 5952-5959. https://doi.org/10.1073/pnas.96.11.5952.

Cassman, K., Dobermann, A. and Walters, D. (2002). *Agroecosystems, Nitrogen-use Efficiency, and Nitrogen Management*. Agronomy and Horticulture – Faculty Publications 356. University of Nebraska, United States. https://digitalcommons.unl.edu/agronomyfacpub/356.

Cedrez, C., Chamberlin, J., Guo, Z. and Hijmans, R. (2020). Spatial variation in fertilizer prices in Sub-Saharan Africa. *PLoS ONE* 15(1), e0227764. https://doi.org/10.1371/journal.pone.0227764.

Chen, Q., Peng, S., Dong, H., Wang, W. and Nie, L. (2018). Optimal nitrogen fertilizer management for direct seeding rice: A review. *International Journal of Agriculture and Biotechnology* 20(6), 1382-1390. http://www.fspublishers.org/published_papers/20603.pdf.

Chien, S., Prochnow, L. and Cantarella, H. (2009). Recent developments of fertilizer production and use to improve nutrient efficiency and minimize environmental impacts. *Advances in Agronomy* 102, 267-322. https://doi.org/10.1016/S0065-2113(09)01008-6.

Chien, S., Prochnow, L., Tu, S. and Snyder, C. (2011). Agronomic and environmental aspects of phosphate fertilizers varying in source and solubility: An update review. *Nutrient Cycling in Agroecosystems* 89(2), 229-255. https://doi.org/10.1007/s10705-010-9390-4.

Chivenge, P., Vanlauwe, B. and Six, J. (2011). Does the combined application of organic and mineral nutrient sources influence maize productivity? A meta-analysis. *Plant and Soil* 342, 1-30. https://doi.org/10.1007/s11104-010-0626-5.

Christodoulou, A. and Stamatelatou, K. (2016). Overview of legislation on sewage sludge management in developed countries worldwide. *Water Science and Technology* 73(3), 453-462. https://doi.org/10.2166/wst.2015.521.

Codex Alimentarius Commission (2013). *Guidelines for the Production, Processing, Labelling and Marketing of Organically Produced Foods*. Rome: Codex Alimentarius Commission, Joint FAO/WHO Food Standards Programme. http://www.codexalimentarius.net.

Cotter, M., Asch, F., Abera, B., Chuma, B., Senthilkumar, K., Rajaona, A. *et al*. (2020). Creating the data basis to adapt agricultural decision support tools to new environments, land management and climate change – A case study of the RiceAdvice App. *Journal of Agronomy and Crop Science* 206, 423-432. https://onlinelibrary.wiley.com/doi/pdf/10.1111/jac.12421.

Crews, T. and Peoples, M. (2004). Legume versus fertilizer sources of nitrogen: Ecological tradeoffs and human needs. *Agriculture, Ecosystems and Environment* 102, 279-297. https://doi.org/10.1016/j.agee.2003.09.018.

CropLife (2017). A look at potash origin and sources. https://www.croplife.com/crop-inputs/a-look-at-potash-origin-and-sources/. Accessed 24 October 2020.

Cui, Z., Zhang, H., Chen, X., Zhang, C., Ma, W., Huang, C. *et al*. (2018). Pursuing sustainable productivity with millions of smallholder farmers. *Nature* 555, 363-366. https://doi.org/10.1038/nature25785.

da Costa Simões, D., Caixeta-Filho, J.V. and Palekar, U.S. (2018). Fertilizer distribution flows and logistic costs in Brazil: Changes and benefits arising from investments in port terminals. *International Food and Agribusiness Management Review* 21 (3), 292-316. https://doi.org/10.22434/IFAMR2017.0051.

de Vries, W. and Schulte-Uebbing, L. (2020). *Required Changes in Nitrogen Inputs and Nitrogen Use Efficiencies to Reconcile Agricultural Activities with Water and Air Quality Objectives in the EU-27*. Cambridge: International Fertiliser Society at a Conference. Proceedings No. 842. https://www.fertilizerseurope.com/wp-content/uploads/2020/03/Proc-842-de-Vries-short-abstract_-21-Feb.pdf.

Delgado, J.A., Barrera, M., Escudero, L., Cartagena, A., Alwang, J., Stehouwer, R. *et al.* (2019). Conservation agriculture increases profits in an Andean region of South America. *Agrosystems, Geosciences and Environment* 2(1), 1-8. https://doi.org/10.2134/age2018.10.0050.

Delgado, J.A., Dillon, M., Sparks, R. and Essah, S. (2007). A decade of advances in cover crops: Cover crops with limited irrigation can increase yields, crop quality, and nutrient and water use efficiencies while protecting the environment. *Journal of Soil and Water Conservation* 62(5), 110A-117A. https://www.researchgate.net/publication/43261805_A_decade_of_advances_in_cover_crops.

Delgado, J.A., Khosla, R., Bausch, W., Westfall, D. and Inman, D. (2005). Nitrogen fertilizer management based on site specific management zones reduces potential for nitrate leaching. *Journal of Soil and Water Conservation* 60, 402-410. https://www.jswconline.org/content/60/6/402.

Delgado, J.A., Sassenrath, G.F. and Mueller, T. (2020). *Precision Conservation: Goespatial Techniques for Agricultural and Natural Resources Conservation*. Wiley. https://www.wiley.com/en-us/Precision+Conservation:+Goespatial+Techniques+for+Agricultural+and+Natural+Resources+Conservation-p-9780891183556.

Delgado, J.A., Short, N.J., Roberts, D. and Vandenburg, B. (2019). Big data analysis for sustainable agriculture on a geospatial cloud framework. *Frontiers in Sustainable Food Systems* 3, 54. https://doi.org/10.3389/fsufs.2019.00054.

Dersseh, W., Gebresilase, Y., Schulte, R. and Struik, P. (2016). The analysis of potato farming systems in Chencha, Ethiopia: Input, output and constraints. *American Journal of Potato Research* 93, 436-447. https://doi.org/10.1007/s12230-016-9521-9.

Diallo, A., Balogun, A., Ba, M.N. and Prud'homme, M. (2019). Fertilizer supply. *In Feeding Africa's Soils: Fertilizers to Support Africa's Agricultural Transformation*. Nairobi: Alliance for a Green Revolution in Africa. 51-66. https://agra.org/wp-content/uploads/2019/11/FeedingAfrica'sSoils.pdf.

Dickinson, D., Balduccio, L., Buysse, J., Ronsse, F., van Huylenbroeck, G. and Prins, W. (2015). Cost-benefit analysis of using biochar to improve cereals agriculture. *GCB-Bioenergy* 7, 850-864. https://doi.org/10.1111/gcbb.12180.

Dimkpa, C.O., and Bindraban, P. (2018). Nanofertilizers: New products for the industry? *Journal of Agricultural and Food Chemistry* 66(26), 6462-6473. https://doi.org/10.1021/acs.jafc.7b02150.

Dimpka, C.O., Fugice, J., Singh, U. and Lewis, T.D. (2020). Development of fertilizers for enhanced nitrogen use efficiency – Trends and perspectives. *Science of The Total Environment* 731, 139113. https://doi.org/10.1016/j.scitotenv.2020.139113.

Dobermann, A. and Cassman, K. (2005). Cereal area and nitrogen use efficiency are drivers of future nitrogen fertilizer consumption. *Science in China Series C: Life Sciences* 48, 745-758. https://pubmed.ncbi.nlm.nih.gov/20549431/.

Duhan, J., Kumar, R., Kumar, N., Kaur, P., Nehra, K. and Duhan, S. (2017). Nanotechnology: The new perspective in precision agriculture. *Biotechnology* 15, 11-23. https://doi.org/10.1016/j.btre.2017.03.002.

Edmeades, D.C. (2003). The long-term effects of manures and fertilisers on soil productivity and quality: a review. *Nutrient Cycling in Agroecosystems* 66, 165-180. https://doi.org/10.1023/A:1023999816690.

EIP-AGRI. (2017). *EIP-AGRI Focus Group on Mixed Farming Systems: Livestock/Cash Crops*. *Final Report May 2017*. https://ec.europa.eu/eip/agriculture/sites/agri-eip/files/fg16_mixed_farming_final-report_2017_en.pdf.

Erisman, J., Sutton, M., Galloway, J., Klimont, Z. and Winiwarter, W. (2008). How a century of ammonia synthesis changed the world. *Nature Geoscience* 1, 636-639. https://doi.org/10.1038/ngeo325.

European Chemicals Agency (2019). *Annex XV Restriction Report: Proposal for a Restriction*. *Intentionally Added Microplastics*. https://echa.europa.eu/documents/10162/05bd96e3-b969-0a7c-c6d0-441182893720.

European Commission (2016). *Proposal for a Regulation of the European Parliament and of the Council" Laying Down Rules on the Making Available on the market of CE Marked Fertilising Products and Amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009*. https://eur-lex.europa.eu/resource.html?uri=cellar:5aa49d31-ec29-11e5-8a81-01aa75ed71a1.0001.02/DOC_3&format=PDF#page=6.

European Fertilizer Manufacturers Association (2004). *Guidance for Sea Transport of Ammonium Nitrate Based Fertilizers*. https://issuu.com/efma2/docs/130917110758-5c7e4384a5b44f5e95136b44fd2a043a.

European Fertilizer Manufacturers Association (2007). *Guidance for the Storage, Handling and Transportation of Solid Mineral Fertilizers*. https://www.fertilizerseurope.com/wp-content/uploads/2019/08/Guidance_for_the_storage_handling_and_transportation_of_solid_mineral_fertilizers.pdf.

European Union (2003). Regulation (EC) No 2003/2003 of the European Parliament and of the Council of 3 October 2003 relating to fertilisers (Text with EEA relevance). *Official Journal of the European Union* L 304/1. https://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2003:304:0001:0194:en:PDF.

European Union (2019). Regulation (EU) 2019/1009 of the European Parliament and of the Council of 5 June 2019 laying down rules on the making available on the market of EU fertilising products and amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and repealing Regulation (EC) No 2003/2003 (Text with EEA relevance). *Official Journal of the European Union* L 170/1. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32019R1009.

Fabregas, R., Kremer, M. and Schilbach, F. (2019). Realizing the potential of digital development: The case of agricultural advice. *Science* 366(6471), 1328. http://doi.org/10.1126/science.aay3038.

Fairhurst, T. (ed.). (2012). *Handbook for Integrated Soil Fertility Management*. Nairobi: Africa Soil Health Consortium. https://www.cabi.org/cabebooks/ebook/20163213016.

Feng, Y., Yang, X., Singh, B., Mandal, S., Guo, J., Che, L. and Wang, H. (2019). Effects of contrasting biochars on the leaching of inorganic nitrogen from soil. *Journal of Soils and Sediments* 20, 3017-3026. https://doi.org/10.1007/s11368-019-02369-5.

Fertilizer Focus (2019). *Rail Transportation in North America: Time to Adapt and Lead*. https://www.kuritaamerica.com/Blog%20Images/FF_Jul_Aug_2019_lowres_spreads.pdf.

Fertilizers Europe (2006). *Guidance for the Compatibility of Fertilizer Blending Materials*. https://www.fertilizerseurope.com/wp-content/uploads/2019/08/Guidance_for_compatibility2.pdf.

Fertilizers Europe (2007). *Guidance for Transporting Ammonia by Rail*. 2007 Edition. https://www.fertilizerseurope.com/wp-content/uploads/2019/08/Guidance_for_transporting_ammonia_in_rail_4.pdf.

Fertilizers Europe (2019). *Industry Facts and Figures 2019*. https://www.fertilizerseurope.com/wp-content/uploads/2019/07/Industry-Facts-and-Figures-2019-Digital-version.pdf.

Finger, R., Swinton, S., El Benni, L. and Walter, A. (2019). Precision farming at the nexus of agricultural production and the environment. *Annual Review of Resource Economics* 11, 313-335. https://doi.org/10.1146/annurev-resource-100518-093929.

Fixen, P. (2020). A brief account of the genesis of 4R nutrient stewardship. *Agronomy Journal* 112(5), 4511-4518. https://doi.org/10.1002/agj2.20315.

Fixen, P., Brentrup, F., Bruulsema, T., Garcia, F., Norton, R. and Zingore, S. (2015). Nutrient/fertilizer use efficiency: Measurement, current situation and trends. In *Managing Water and Fertilizer for Sustainable Agricultural Intensification*. First Edition. Drechsel, P., Heffer, P., Magen, H., Mikkelsen, R., Singh, H. and Wichelns, D. (eds.). Paris: International Fertilizer Industry Association, International Water Management Institute, International Plant Nutrition Institute and International Potash Institute. http://www.iwmi.cgiar.org/Publications/Books/PDF/managing_water_and_fertilizer_for_sustainable_agricultural_intensification.pdf.

Food and Agriculture Organization of the United Nations (1991). *Manual on Fertilizer Statistics*. http://www.fao.org/fileadmin/templates/ess/ess_test_folder/Publications/ManualFertilizers.pdf.

Food and Agriculture Organization of the United Nations (2005). Fertilizer use by crop in India. Fertilizer distribution and credit. http://www.fao.org/3/a0257e/A0257E07.htm. Accessed 24 October 2020.

Food and Agriculture Organization of the United Nations (2011). *Case Studies on Policies and Strategies for Sustainable Soil Fertility and Fertilizer Management in South Asia*. Bangkok: FAO Regional Office for Asiz and the Pacific. http://www.fao.org/3/i2308e/i2308e00.htm.

Food and Agriculture Organization of the United Nations (2019a). *The International Code of Conduct for the Sustainable Use and Management of Fertilizers*. http://www.fao.org/3/ca5253en/CA5253EN.pdf.

Food and Agriculture Organization of the United Nations (2019b). *World Fertilizer Trends and Outlook to 2022*. http://www.fao.org/3/ca6746en/CA6746EN.pdf?eloutlink=imf2fao.

Food and Agriculture Organization of the United Nations (2020a). FAOSTAT. Fertilizers by nutrient. https://www.fao.org/faostat/en/#data/RFN. Accessed 24 October 2020.

Food and Agriculture Organization of the United Nations (2020b). FAOSTAT. Livestock manure https://www.fao.org/faostat/en/#data/EMN. Accessed 24 October 2020.

Food and Agriculture Organization of the United Nations (2020c). *Dataset Information*. *Domain: Fertilizers by Nutrient*. http://fenixservices.fao.org/faostat/static/documents/RFN/RFN_EN_README.pdf.

Food and Agriculture Organization of the United Nations (2020d). FAOSTAT. Land use. https://www.fao.org/faostat/en/#data/RL. Accessed 24 October 2020.

Food and Agriculture Organization of the United Nations (2020e). FAOSTAT. Fertilizers indicators. https://www.fao.org/faostat/en/#data/EF. Accessed 24 October 2020.

Food and Agriculture Organization of the United Nations (2020f). FAOSTAT. Crops. https://www.fao.org/faostat/en/#data/QC. Accessed 24 October 2020.

Food and Agriculture Organization of the United Nations (2020g). FAOSTAT. Annual population. https://www.fao.org/faostat/en/#data/OA. Accessed 24 October 2020.

Food and Agriculture Organization of the United Nations and World Health Organization (2001). Codex Alimentarius – Organically Produced Foods. http://www.fao.org/3/Y2772E/Y2772E00.htm. Accessed 24 October 2020.

Franke, A., van den Brand, G., Vanlauwe, B. and Giller, K. (2018). Sustainable intensification through rotations with grain legumes in SubSaharan Africa: A review. *Agriculture, Ecosystems and Environment* 261, 172-185. https://doi.org/10.1016/j.agee.2017.09.029.

Govers, G., Merckx, R., Van Oost, K. and van Wesemael, B. (2013). *Managing Soil Organic Carbon for Global Benefits: A STAP Technical Report*. Washington, D.C.: Global Environment Facility. https://thegef.org/sites/default/files/publications/STAP-SOC-Report-lowres-1_0.pdf.

Gram, G., Roobroeck, D., Pypers, P., Six, J., Merckx, R. and Vanlauwe, B. (2020). Combining organic and mineral fertilizers as a climate-smart integrated soil fertility management practice in sub-Saharan Africa: A meta-analysis. *PLoS ONE* 15(9), e0239552. https://doi.org/10.1371/journal.pone.0239552.

Gro Intelligence (2016). The world's most expensive fertilizer market: Sub-Saharan Africa, 2 December. https://gro-intelligence.com/insights/articles/fertilizers-in-sub-saharan-africa. Accessed 21 October 2020.

Gumucio, T., Hansen, J., Huyer, S., van Huysen, T. and Schwager, S. (2018). *Identifying Pathways for More Gender-Sensitive Communication Channels in Climate Services*. *Info Note*. CGIAR Research Program on Climate Change, Agriculture and Food Security. https://cgspace.cgiar.org/bitstream/handle/10568/96531/Info%20Note_Gender%20Comm_07.pdf.

Halpern, M., Bar-Tal, A., Ofek, M., Minz, D., Muller, T. and Yermiyahu, U. (2015). The use of biostimulants for enhancing nutrient uptake. *Advances in Agronomy* 130, 141-174. https://doi.org/10.1016/bs.agron.2014.10.001.

Halvorson, A., Snyder, C., Blaylock, A. and Del Grosso, S. (2014). Enhanced-efficiency nitrogen fertilizers: Potential role in nitrous oxide emission mitigation. *Agronomy Journal* 106(2), 715-722. https://doi.org/10.2134/agronj2013.0081.

Han, M., Wong, J., Su, T., Beatty, P. and Good, A. (2016). Identification of nitrogen use efficiency genes in barley: Searching for QTLs controlling complex physiological traits. *Frontier Plant Science 7,* 1587. https://doi.org/10.3389/fpls.2016.01587.

Heffer, P., Gruère, A. and Roberts, T. (2017). *Assessment of Fertilizer Use by Crop at the Global Level 2014-2014/15*. Paris: International Fertilizer Association and International Plant Nutrition Institute. https://www.fertilizer.org/images/Library_Downloads/2017_IFA_AgCom_17_134%20rev_FUBC%20assessment%202014.pdf.

Hemming, D., Chirwa, E., Ruffhead, H., Hill, R., Osborn, J., Langer, L. *et al*. (2018). Agricultural Input subsidies for improving productivity, farm income, consumer welfare and wider growth in low- and medium-income countries. A systematic review. *Campbell Systematic Reviews* 14(1) 1-153. https://doi.org/10.4073/csr.2018.4.

Hernandez, M. and Torero, M. (2018). *Promoting Competition in the Fertilizer Industry in Africa: A Global and Local Approach*. *IFPRI Issue Brief*. Washingon D.C.: International Food Policy Institute. https://www.ifpri.org/publication/promoting-competition-fertilizer-industry-africa-global-and-local-approach.

Hernandez, M., Paz, F., Adeoti, A., Adong, A., Jumbe, C., Lagat, K. and Kaiyatsa, S. (2018). *Overview of the Fertilizer Supply Chain and Market Structure in Africa: A Cross-country Assessment*. *Project Note*. Washington D.C.: International Food Policy Research Institute. https://www.ifpri.org/publication/overview-fertilizer-supply-chain-and-market-structure-africa-cross-country-assessment.

Herridge, D., Peoples, M. and Boddey, R. (2008). Global inputs of biological nitrogen fixation in agricultural systems. *Plant and Soil* 311, 1-18. https://doi.org/10.1007/s11104-008-9668-3.

Hijbeek, R., van Loon, M. and van Ittersum, M. (2019). *Fertiliser Use and Soil Carbon Sequestration: Opportunities and Trade-offs*. CCAFS Working Paper No. 264. Wageningen, the Netherlands: CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS). https://www.researchgate.net/publication/334491113_Fertiliser_use_and_soil_carbon_sequestration_trade-offs_and_opportunities.

Holden, S. (2018). Fertilizer and sustainable intensification in Sub-Saharan Africa. *Global Food Security* 18, 20-26. https://doi.org/10.1016/j.gfs.2018.07.001.

Holloway, R., Bertrand, I., Frischke, A., Brace, D., McLaughlin, M. and Shepperd, W. (2001). Improving fertiliser efficiency on calcareous and alkaline soils with fluid sources of P, N and Zn. *Plant and Soil* 236(2), 209-219. https://doi.org/10.1023/A:1012720909293.

Hosseini, S., Nazrai, R. and Lashgarara, F. (2011). Factor influencing the adoption of nanotechnology by farmers in Iran. *Australian Journal of Basic and Applied Sciences* 5(9), 1130-1134. http://www.ajbasweb.com/old/ajbas/2011/September-2011/1130-1134.pdf.

Hou, Y., Velthof, G., Case, S., Oelofse, M., Grignani, C., Balsari, P. *et al*. (2018). Stakeholder perceptions of manure treatment technologies in Denmark, Italy, the Netherlands and Spain. *Journal of Cleaner Production* 172, 1620-1630. https://doi.org/10.1016/j.jclepro.2016.10.162.

Houlton, B., Almaraz, M., Aneja, V., Austin, A., Bai, E., Cassman, K. *et al*. (2019). A world of cobenefits: Solving the global nitrogen challenge. *Earth's Future* 7, 865-872. https://doi.org/10.1029/2019EF001222.

Huang, J., Gulati, A. and Gregory, I. (eds.) (2017). *Fertilizer Subsidies – Which Way Forward? An IFDC/FAI Report*. International Fertilizer Development Center and Fertilizer Association of India. https://pdfslide.net/download/link/fertilizer-subsidies-which-way-forward-subsidies-which-way-forward.

Huang, J., Huang, Z., Jia, X., Hu, R. and Xiang, C. (2015). Long-term reduction of nitrogen fertilizer use through knowledge training in rice production in China. *Agricultural Systems* 135, 105-111. https://doi.org/10.1016/j.agsy.2015.01.004.

Hurley, T., Koo, J. and Tesfaye, K. (2018). Weather risk: How does it change the yield benefits of nitrogen fertilizer and improved maize varieties in sub-Saharan Africa? *Agricultural Economics* 49, 711-723. https://doi.org/10.1111/agec.12454.

IFOAM – Organics International (n.d. a). Definition of organic agriculture. https://www.ifoam.bio/en/organic-landmarks/definition-organic-agriculture. Accessed 24 October 2020.

IFOAM – Organics International (n.d. b). Why organic? https://www.ifoam.bio/en/our-library/organic-basics. Accessed 24 October 2020.

Ilinova, A., Dmitrieva, D. and Kraslawski, A. (2021). Influence of COVID-19 pandemic on fertilizer companies: The role of competitive advantages. *Resources Policy* 71, 102019. https://www.sciencedirect.com/science/article/pii/S0301420721000362.

International Fertilizer Association (2017). *Assessment of the Global Market for Slow- and Controlled-Release, Stabilized and Water-Soluble Fertilizers in 2016*. https://www.fertilizer.org/ItemDetail?iProductCode=10313Pdf&Category=FERT. (Available to IFA members only.)

International Fertilizer Association (2019a). *Fertilizer Outlook 2019-2023*. IFA Annual Conference, Montreal, 11-13 June 2019. https://www.ifastat.org/market-outlooks.

International Fertilizer Association (2019b). *Medium-Term Outlook for Fertilizers and Raw Materials Global Supply: 2019-2023. IFA annual conference 11-13 June 2019, Montreal, Canada*. https://www.ifastat.org/market-outlooks. (Available to IFA members only.)

International Fertilizer Association (2020a). *Fertilizer Outlook 2020-2024*. https://www.ifastat.org/market-outlooks.

International Fertilizer Association (2020b). *Global Medium-Term Outlook for Fertilizers and Raw Materials: 2020-2024*. https://www.ifastat.org/market-outlooks. (Available to IFA members only.)

International Fertilizer Association (2020c). Phosphate products. https://www.ifastat.org/supply/Phosphate%20Products/Processed%20Phosphates. Accessed 24 October 2020.

International Fertilizer Association (2020d). Potash stories. https://www.fertilizer.org/Public/About_fertilizers/Facts_and_Figures/Potash_Stories/Public/About_Fertilizers/Potash_Stories.aspx?hkey=7308fd82-077f-4a81-b983-697ef88a9f0f. Accessed 24 October 2020.

International Fertilizer Association (2020e). Frequently asked questions. Why is there a difference between the supply database and the consumption database? https://www.ifastat.org/faq. Accessed 24 October 2020.

International Fertilizer Association (2020f). *Global Sustainability Report 2019*. https://www.scribd.com/document/460442439/2020-IFA-Sustainability-Report-pdf-pdf.

International Fertilizer Association (2020g). Nutrient stewardship. https://www.fertilizer.org/Public/Stewardship/Nutrient_Stewardship/Public/Stewardship/Nutrient/Nutrient_Stewardship.aspx?hkey=a92ea40d-dcb7-4d14-b3a5-0ad42bd0929e. Accessed 24 October 2020.

International Fertilizer Association (2021a). *Public Summary: Medium-Term Fertilizer Outlook 2021-2025*. https://www.ifastat.org/market-outlooks.

International Fertilizer Association (2021b). Fertilizers, for productive and sustainable agriculture systems. https://www.fertilizer.org. Accessed 10 January 2022.

International Fertilizer Association (n.d.). Databases and charts. Consumption. https://www.ifastat.org/databases/plant-nutrition. Accessed 24 October 2020

International Fertilizer Development Center (2017). *Rapid Introduction and Market Development for Urea Deep Placement Technology for Lowland Transplanted Rice: A Reference Guide*. http://www.mediaterre.org/docactu,cGV4aW5lZy9kb2NzL3VyZWEtZGVlcC1wbGFjZW1lbnQtdGVjaG5vbG9neQ==,11.pdf.

International Maritime Organization. (2006). *International Maritime Dangerous Goods Code. Incoporating Ammendment 33-06* . https://law.resource.org/pub/us/cfr/ibr/004/imo.imdg.1.2006.pdf.

International Maritime Organization (2019). IMDG Code. https://www.imo.org/en/publications/Pages/IMDG%20Code.aspx. Accessed 27 October 2020.

Iqbal M., Umar S., and Mahmooduzzafar (2019). Nano-fertilization to enhance nutrient use efficiency and productivity of crop plants. In *Nanomaterials and Plant Potential*. Husen A. and Iqbal M. (eds.). Springer. https://doi.org/10.1007/978-3-030-05569-1_19.

Jayne, T., Mason, N., Burke, W. and Ariga, J. (2018). Review: Taking stock of Africa's second-generation agricultural input subsidy programs. *Food Policy* 75, 1-14. https://doi.org/10.1016/j.foodpol.2018.01.003.

Johnson, A. and Bruulsema, T. (2014). 4R Nutrient Stewardship for improved nutrient use efficiency. *Procedia Engineering* 83, 365-370. https://doi.org/10.1016/j.proeng.2014.09.029.

Ju, X., Kou, C., Christie, P., Dou, Z. and Zhang, F. (2007). Changes in the soil environment from excessive application of fertilizers and manures to two contrasting intensive cropping systems on the North China Plain. *Environmental Pollution* 145(2), 497-506. https://doi.org/10.1016/j.envpol.2006.04.017.

Junaidi, J., Kallenbach, C., Byrne, P. and Fonte, S. (2018). Root traits and root biomass allocation impact how wheat genotypes respond to organic amendments and earthworms. *PLoS ONE* 13(7), e0200646. https://doi.org/10.1371/journal.pone.0200646.

Kafkafı, U. and Tarchitzky, J. (2011). *Fertigation: A Tool for Efficient Fertilizer and Water Management*. Paris: International Fertilizer Industry Association (IFA) and International Potash Institute (IPI). https://www.haifa-group.com/sites/default/files/ifa_fertigation-Kafkafi-511.pdf.

Kah, M., Kookana, R., Gogos, A. and Bucheli, T. (2018). A critical review of nanopesticides and nanofertilizers against their conventional analogues. *Nature Nanotechnology* 13, 677-684. https://doi.org/10.1038/s41565-018-0131-1.

Kanter, D., Zhang, X. and Mauzerall, D. (2014). Reducing nitrogen pollution while decreasing farmers' costs and increasing fertilizer industry profits. *Journal of Environmental Quality* 44(2), 325-335. https://doi.org/10.2134/jeq2014.04.0173.

Kasprzycka, A., Lalak-Kańczugowska, J., Tys, J., Chmielewska, M. and Pawłowska, M. (2018). Chemical stability and sanitary properties of pelletized organo-mineral waste-derived fertilizer. *Archives of Environmental Protection* 44(3), 106-113. http://journals.pan.pl/Content/106586/PDF/Archives%20vol%2044%20no%203_12.pdf.

Kassie, M., Teklewold, H., Jaleta, M., Marenya, P. and Erenstein, O. (2015). Understanding the adoption of a portfolio of sustainable intensification practices in eastern and southern Africa. *Land Use Policy* 42 400-411. https://doi.org/10.1016/j.landusepol.2014.08.016.

Kenya Bureau of Standards (2014). *Blending fertilizers – Code of Practice*. *Draft Standard*. *Kenya Standard KS 2528:2014*. http://www.puntofocal.gov.ar/notific_otros_miembros/ken410_t.pdf.

Kisinyo, P., Othieno, C., Gudu, S., Okalebo, J., Opala, P., Ng'etich, W. *et al*. (2014). Immediate and residual effect of lime and phosphorus fertilizer on soil acidity and maize production in western Kenya. *Experimental Agriculture* 50(1), 128-143. https://doi.org/10.1017/S0014479713000318.

Kominko, H., Gorazda, K. and Wzorek, Z. (2017). The possibility of organo-mineral fertilizer production from sewage sludge. *Waste and Biomass Valorization* 8, 1781-1791. https://doi.org/10.1007/s12649-016-9805-9.

Kopittke, P., Menzies, N., Wang, P., McKenna, B. and Lombi, E. (2019). Soil and the intensification of agriculture for global food security. *Environmental International* 132, 105078. https://doi.org/10.1016/j.envint.2019.105078.

Koppelaar, R.H.E.M. and Weikard, H.P. (2013). Assessing phosphate rock depletion and phosphorus recycling options. *Global Environmental Change* 23(6), 1454-1466. http://dx.doi.org/10.1016/j.gloenvcha.2013.09.002.

Koppmair, S., Kassie, M. and Qaim, M. (2017). The influence of farm input subsidies on the adoption of natural resource management technologies. *Australian Journal of Agricultural and Resource Economics* 61, 539-556. https://doi.org/10.1111/1467-8489.12220.

Kpadonou, R., Barbier, B., Denton, F. and Owiyo, T. (2015). *Linkage Between and Determinants of Organic Fertilizer and Modern Varieties Adoption in the Sahel*. International Conference of Agricultural Economics. https://doi.org/10.22004/ag.econ.212016.

Kuhulman, T., Helming, J. and Linderhof, V. (2017). Optimizing legume cropping: The policy questions. In *Legumes in Cropping Systems*. Murphy-Bokern, D.,Stoddard, F. and Watson, C. Wallingford, United Kingdom: CABI International. 226-243. https://www.cabi.org/bookshop/book/9781780644981/.

Kumari, R., Kundu, M., Das, A., Rakshit, R., Sahay, S., Sengupta, S. and Ahmad, F. (2019). Long-term integrated nutrient management improves carbon stock and fruit yield in a subtropical mango (*Mangifera indica* L.) orchard. *Journal of Soil Science and Plant Nutrition* 20, 725-737. https://doi.org/10.1007/s42729-019-00160-6.

Lal, R., Delgado, J., Groffman, P., Millar, N., Dell, C. and Rotz, A. (2011). Management to mitigate and adapt to climate change. *Journal of Soil and Water Conservation* 66(4), 276-285. https://doi.org/10.2489/jswc.66.4.276.

Lassaletta, L., Billen, G., Grizzetti, B., Anglade, J. and Garnier, J. (2014). 50 year trends in nitrogen use efficiency of world cropping systems: The relationship between yield and nitrogen input to cropland. *Environmental Research Letters* 9(10), 105011. https://iopscience.iop.org/article/10.1088/1748-9326/9/10/105011/meta.

Li, B., Bicknell, K.B. and Renwick, A. (2019). Peak phosphorus, demand trends and implications for the sustainable management of phosphorus in China. *Resources, Conservation and Recycling* 146, 316-328. https://doi.org/10.1016/j.resconrec.2019.03.033.

Li, G., Huang, G., Li, H., van Ittersum, M., Leffelaar, P. and Zhang, F. (2016). Identifying potential strategies in the key sectors of China's food chain to implement sustainable phosphorus management: A review. *Nutrient Cycling in Agroecosystems* 104, 341-359. https://doi.org/10.1007/s10705-015-9736-z.

Li, S., Wang, W., Cao, Y., Wang, C., Yan, C., Dong, L. *et al*. (2019). How root traits would be affected by soybean yield improvement? An examination of historical cultivars grafted with record-yield cultivar scion. *Plant and Soil* 439, 19-30. https://doi.org/10.1007/s11104-018-3792-5.

Li, T., Zhang, W., Yin, J., Chadwick, D., Norse, D., Lu, Y. *et al*. (2017). Enhanced-efficiency fertilizers are not a panacea for resolving the nitrogen problem. *Global Change Biology* 24(2), e511-e521. https://doi.org/10.1111/gcb.13918.

Li, Y., Hu, S., Chen, J., Müller, K., Li, Y., Fu, W. *et al*. (2018). Effects of biochar application in forest ecosystems on soil properties and greenhouse gas emissions: A review. *Journal of Soils and Sediments* 18(2), 546-563. https://doi.org/10.1007/s11368-017-1906-y.

Li, Y., Zhang, W., Ma, L., Huang, G., Oenema, O., Zhang, F. and Dou, Z. (2013). An analysis of China's fertilizer policies: Impacts on the industry, food security, and the environment. *Journal of Environmental Quality* 42(2), 972-981. https://doi.org/10.2134/jeq2012.0465.

Louwagie, G., Gay, S. and Burrell, A. (eds.) (2009). *Addressing Soil Degradation in EU Agriculture: Relevant Processes, Practices and Policies*. *Report on the Project 'Sustainable Agriculture and Soil conservation (SoCo)'. JRC Scientific and Technical Reports*. European Commission Joint Research Centre, Institute for Prospective Technological Studies. https://publications.jrc.ec.europa.eu/repository/bitstream/JRC50424/jrc50424.pdf.

Luther, Z., Swinton, S. and Deynze, B. (2020). What drives voluntary adoption of farming practices that can abate nutrient pollution? *Journal of Soil and Water Conservation* 75 (5) 640-650. https://doi.org/10.2489/jswc.2020.00184.

Lv, H., Lin, S., Wang, Y., Lian, X., Zhao, Y., Li, Y. *et al*. (2018). Drip fertigation significantly reduces nitrogen leaching in solar greenhouse vegetable production system. *Environmental Pollution* 245, 694-701. https://doi.org/10.1016/j.envpol.2018.11.042.

Mather, D. and Jayne, T. (2018). Fertilizer subsidies and the role of targeting in crowding out: Evidence from Kenya. *Food Security* 10, 397-417. https://doi.org/10.1007/s12571-018-0773-8.

McAllister, C., Beatty, P. and Good, A. (2012). Engineering nitrogen use efficient crop plants: The current status. *Plant Biotechnology Journal* 10(9), 1011-1025. https://doi.org/10.1111/j.1467-7652.2012.00700.x.

McFadden, B. and Smyth, S. (2019). Perceptions of genetically engineered technology in developed areas. *Trends in Microbiology* 37(5), 447-451. https://doi.org/10.1016/j.tibtech.2018.10.006.

McIntosh, I., Sarris, A. and Papadopoulos, F. (2013). Productivity, credit, risk, and the demand for weather index insurance in smallholder agriculture in Ethiopia. *Agricultural Economics* 44, 399-417. https://www.povertyactionlab.org/sites/default/files/research-paper/623%20Weather%20Insurance%20Ethiopia.pdf.

Medina, J., Monreal, C., Barea, J., Arriagada, C., Borie, F. and Cornejo, P. (2015). Crop residue stabilization and application to agricultural and degraded soils: A review. *Waste Management* 42, 41-54. https://doi.org/10.1016/j.wasman.2015.04.002.

Mehta, C., Khunjar, W., Nguyen, V., Tait, S. and Batstone, D. (2015). Technologies to recover nutrients from waste streams: A critical review. *Critical Reviews in Environmental Science and Technology* 45(4), 385-427. https://doi.org/10.1080/10643389.2013.866621.

Melash, A., Mengistu, D., Aberra, D. and Tsegay, A. (2019). The influence of seeding rate and micronutrients foliar application on grain yield and quality traits and micronutrients of durum wheat. *Journal of Cereal Science* 85, 221-227. https://doi.org/10.1016/j.jcs.2018.08.005.

Mikkelsen, R. (2018). Nanofertilizer and nanotechnology: A quick look. *Better Crops* 102, 18-19. https://www.researchgate.net/publication/327708151_Nanofertilizer_and_Nanotechnology_A_quick_look.

Miyake, A., Wakakura, M., Uchida, T. and Ushikubo, A. (2006). Investigation of accidental explosion of raw garbage composting system. *Journal of Thermal Analysis and Calorimetry* 83(3), 643-649. https://doi.org/10.1007/s10973-006-7637-8.

Montwiłł A. (2014). Trends in the development of European inland freight transport. *Scientific Journals* 37(109), 6671. https://1library.net/document/qm8lk24z-trends-development-european-inland-freight-transport.html?utm_source=search_v3.

Mordor Intelligence (2020). Fertilizer market – share, growth, trends, and forecast (2020-2025). https://www.mordorintelligence.com/industry-reports/fertilizers-market. Accessed 25 October 2020.

Mordor Intelligence (2021a). Controlled release fertilizer market – growth, trends, COVID-19 impact, and forecasts (2021-2026). https://www.mordorintelligence.com/industry-reports/controlled-release-fertilizer-market. Accessed 20 May 2021.

Mordor Intelligence (2021b). Fertilizers market – growth, trends, COVID-19 impact, and forecasts (2021-2026). https://www.mordorintelligence.com/industry-reports/fertilizers-market. Accessed 20 May 2021.

Morello, T., Picketty, M., Gardner, T., Parry, L., Barlow, J., Ferreira, J. and Tancredi, N. (2018). Fertilizer adoption by smallholders in the Brazilian Amazon: Farm-level evidence. *Ecological Economics* 144, 278-291. https://doi.org/10.1016/j.ecolecon.2017.08.010.

Morgan, S., Mason, N., Levine, N. and Zulu-Mbata, O. (2019). Dis-incentivizing sustainable intensification? The case of Zambia's maize-fertilizer subsidy program. *World Development* 122, 54-69. https://doi.org/10.1016/j.worlddev.2019.05.003.

Mutegi, J., Kiwia, A. and Zingore, S. (2019). Types of fertilizers. In *Feeding Africa's Soils: Fertilizers to Support Africa's Agricultural Transformation*. Nairobi: Alliance for a Green Revolution in Africa (AGRA). 9-21. https://agra.org/wp-content/uploads/2019/11/FeedingAfrica'sSoils.pdf.

Mwangi, W. (1996). *Low Use of Fertilizers and Low Productivity in Sub-Saharan Africa*. *Natural Resources Group (NRG) Paper 96-05*. Mexico, D.F.: International Maize and Wheat Improvement Center (CIMMYT). https://repository.cimmyt.org/xmlui/bitstream/handle/10883/937/63491.pdf.

Naz, M. and Sulaiman, S. (2016). Slow release coating remedy for nitrogen loss from conventional urea: A review. *Journal of Controlled Release* 225, 109-120. https://doi.org/10.1016/j.jconrel.2016.01.037.

Ndambi, O., Pelster, D., Owino, J., de Buisonjé, F. and Vellinga, T. (2019). Manure management practices and policies in Sub-Saharan Africa: Implications on manure quality as a fertilizer. *Frontiers in Sustainable Food Systems* 3, 29. https://doi.org/10.3389/fsufs.2019.00029.

Nhamo, N., Kintche, K. and Chikoye, D. (2017). Advancing key technical interventions through targeted investment. In *Smart Technologies for Sustainable Smallholder Agriculture: Upscalling in Developing Countries*. Nhamo, N., Chikoye, D. and Gondwe, T. (eds.). London: Academic Press. 39-79. https://www.elsevier.com/books/smart-technologies-for-sustainable-smallholder-agriculture/chikoye/978-0-12-810521-4.

Nicholson, F., Bhogal, A., Rollett, A., Taylor, M. and Williams, M. (2018). Precision application techniques reduce ammonia emissions following food-based digestate applications to grassland. *Nutrient Cycling in Agroecosystems* 110, 151-159. https://doi.org/10.1007/s10705-017-9884-4.

Nkebiwe, P., Weinmann, M., Bar-Tal, A. and Müller, T. (2016). Fertilizer placement to improve crop nutrient acquisition and yield: A review and meta-analysis. *Field Crops Research* 196, 389-401. https://doi.org/10.1016/j.fcr.2016.07.018.

Nyangena, W. and Juma, O. (2014). *Impact of Improved Farm Technologies on Yields: The Case of Improved Maize Varieties and Inorganic Fertilizer in Kenya*. Environment for Development Discussion Paper Series 14-02. https://www.rff.org/publications/working-papers/impact-of-improved-farm-technologies-on-yields-the-case-of-improved-maize-varieties-and-inorganic-fertilizer-in-kenya/.

Olesen, J. and Bindi, M. (2002). Consequences of climate change for European agricultural productivity, land use and policy. *European Journal of Agronomy* 16(4), 239-262. https://doi.org/10.1016/S1161-0301(02)00004-7.

Oosterhuis, D. (2009). *Foliar Fertilization: Mechanisms and Magnitude of Nutrient Uptake*. Paper presented at the Fluid Fertilizer Foundation meeting in Scottsdale, Arizona, United States, 15-17 February 2009. http://www.fluidfertilizer.com/Forum%20Presentations/2009/2009%20Forum%20Proceedings/Derrick%20Oosterhuis.pdf.

Pan, D., Kong, F., Zhang, N. and Ying, R. (2017). Knowledge training and the change of fertilizer use intensity: Evidence from wheat farmers in China. *Journal of Environmental Management* 197, 130-139. https://doi.org/10.1016/j.jenvman.2017.03.069.

Pasandaran, E., Gultom, B., Adiningsih, J., Hapsari and Rochayati, S. (1999). Government policy support for technology promotion and adoption: A case study of urea tablet technology in Indonesia. In *Resource Management in Rice Systems: Nutrients*. Papers presented at the International Workshop on Natural Resource Management in Rice Systems: Technology Adaption for Efficient Nutrient Use, Bogor, Indonesia, 2-5 December 1996. Springer. 181-190. https://www.springer.com/gp/book/9780792351993.

Perin, G., Yunus, I., Valton, M., Alobwede, E. and Jones, P. (2019). Sunlight-driven recycling to increase nutrient use-efficiency in agriculture. *Algal Research* 41, 101554. https://doi.org/10.1016/j.algal.2019.101554.

Pingali, P.L. (2012). Green Revolution: Impacts, limits, and the path ahead. *Proceedings of the National Academy of Sciences of the United States of America* 109(31), 12302-12308. https://doi.org/10.1073/pnas.0912953109.

PLANCO Consulting (2007). *Economical and Ecological Comparison of Transport Modes: Road, Railways, Inland Waterways*. *Summary of Findings*. http://www.ebu-uenf.org/fileupload/SummaryStudy_engl.pdf.

Plume, K. (2019). Floods stall fertilizer shipments in latest blow to U.S. farmers, 25 April. *Reuters*. https://www.reuters.com/article/us-usa-weather-fertilizer-idUSKCN1S11BY. Accessed 5 November 2010.

Powers, S., Chowdhury, R., MacDonald, G., Metson, G., Beusen, A., Bouwman, A. *et al*. (2019). Global opportunities to increase agricultural independence through phosphorus recycling. *Earth's Future* 7(4), 370-383. https://doi.org/10.1029/2018EF001097.

Ragasa, C. (2012). Gender and institutional dimensions of agricultural technology adoption: A review of literature and synthesis of 35 case studies. Selected poster prepared for presentation at the International Association of Agricultural Economists (IAAE) Triennial Conference, Foz do Iguaçu, Brazil, 18-24 August 2012. https://ideas.repec.org/p/ags/iaae12/126747.html.

Rahman, S. and Barmon, B. (2015). Productivity and efficiency impacts of urea deep placement technology in modern rice production: An empirical analysis from Bangladesh. *The Journal of Developing Areas* 49(3), 119-134. https://doi.org/10.1353/jda.2015.0158.

Raina, M., Pandotra, P., Salgotra, R., Ali, S., Mir, Z., Bhat, J. *et al*. (2018). Genetic engineering and environmental risk. In *Modern Age Environmental Problems and their Remediation*. Oves, M., Ismail, M. and Ismail, I. (eds.). Springer. 69-82. https://link.springer.com/book/10.1007/978-3-319-64501-8.

Ribeiro, V.J., Andrade, F.V., Passos, R.R., Mendonça, E.S., da Silva, L.L. and Sartor, A.F. (2016). Slow-release stabilized nitrogen fertilizers on initial development and nutrition of coffee plants (Coffea arabica L.). *Australian Journal of Crop Science* 10(4), 497-502. https://doi.org/10.21475/ajcs.2016.10.04.p7229x.

RiceAdvice (2019). Tools for improving rice value chains in Africa. https://www.riceadvice.info/en/riceadvice/. Accessed 25 October 2020.

Rios, J., Garcia-Ibañez, P. and Carvajal, M. (2019). The use of biovesicles to improve the efficiency of Zn foliar fertilization. *Colloids and Surfaces B: Biointerfaces* 173, 899-905. https://doi.org/10.1016/j.colsurfb.2018.10.057.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F., Lambin, E. *et al*. (2009). A safe operating space for humanity. *Nature* 461, 472-475. https://doi.org/10.1038/461472a.

Ros, G., van Rotterdam, A., Bussink, D. and Bindraban, P. (2016). Selenium fertilization strategies for bio-fortification of food: an agro-ecosystem approach. *Plant and Soil* 404, 99-112. https://doi.org/10.1007/s11104-016-2830-4.

Rouphael, Y. and Colla, G. (2018). Synergistic biostimulatory action: Designing the next generation of plant biostimulants for sustainable agriculture. *Frontiers in Plant Science* 9, 1655. http://doi.org/10.3389/fpls.2018.01655.

Rouphael, Y. and Colla, G. (2020). Editorial: Biostimulants in agriculture. *Frontiers in Plant Science* 11(40), 1-7. https://www.frontiersin.org/articles/10.3389/fpls.2020.00040/full.

Roy, R., Finck, A., Blair, G. and Tandon, H. (2006). *Plant Nutrition for Food Security: A Guide for Integrated Nutrient Management*. *FAO Fertilizer and Plant Nutrition Bulletin 16.* Rome: FAO. http://www.fao.org/fileadmin/templates/soilbiodiversity/Downloadable_files/fpnb16.pdf.

Rurinda, J., van wijk, M., Mapfumo, P., Descheemaeker, K., Supit, I. and Giller, K. (2015). Climate change and maize yield in southern Africa: What can farm management do? *Global Change Biology* 21(12), 4588-4601. https://doi.org/10.1111/gcb.13061.

Rurinda, J., Zingore, S., Jibrin, J., Balemi, T., Masuki, K., Andersson, J. *et al*. (2020). Science-based decision support for formulating crop fertilizer recommendations in sub-Saharan Africa. *Agricultural Systems* 180, 102790. https://doi.org/10.1016/j.agsy.2020.102790.

Rutgersson, C., Ebmeyer, S., Lassen, S.B., Karkman, A., Fick, J. Kristiansson, E. *et al*. (2020). Long-term application of Swedish sewage sludge on farmland does not cause clear changes in the soil bacterial resistome. *Environment International* 137, 105339. https://doi.org/10.1016/j.envint.2019.105339.

Sanabria, J., Ariga, J., Fugice, J. and Mose, D. (2018a). *Fertilizer Quality Assessment in Markets of Kenya*. United States Agency for International Development (USAID) and International Fertilizer Development Center (IFDC). https://ifdc.org/wp-content/uploads/2019/02/FERTILIZER-QUALITY-ASSESSMENT-IN-MARKETS-OF-KENYA.pdf.

Sanabria, J., Ariga, J., Fugice, J. and Mose, D. (2018b). *Fertilizer Quality Assessment in Markets of Uganda*. Muscle Schoals, AL: United States Agency for International Development (USAID) and International Fertilizer Development Center (IFDC). https://ifdc.org/wp-content/uploads/2019/01/Uganda-Fertilizer-Quality-Assessment-Report-1-14-19-1.pdf.

Sanabria, J., Dimithe, G. and Alognikou, E. (2013). T*he Quality of Fertilizer Traded in West Africa: Evidence for Stronger Control*. Joint Economic Community of West African States (ECOWAS), West African Economic and Monetary Union/Union Economique et Monétaire Ouest Africaine (UEMOA) and International Fertilizer Developoment Center (IFDC) Report. Muscle Schoals, AL: IFDC. https://africafertilizer.org/wp-content/uploads/2017/04/The-Quality-of-Fertilizer-Traded-in-West-Africa-Evidence-for-Stronger-Control.pdf.

Schneider, K., Martens, J., Zvomuya, F., Reid, D., Fraser, T., Lynch, D. *et al*. (2019). Options for improved phosphorus cycling and use in agriculture at the field and regional scales. *Journal of Environmental Quality* 48(5), 1247-1264. https://doi.org/10.2134/jeq2019.02.0070.

Schwenke, G., Beange, L., Cameron, J., Bell, M. and Harden, S. (2019). What soil information do crop advisors use to develop nitrogen fertilizer recommendations for grain growers in New South Wales, Australia? *Soil Use and Management* 35(1), 85-93. https://doi.org/10.1111/sum.12469.

Sela, S., van Es, H., Moebius-Clune, B., Marjerison, R., Moebius-Clune, D., Schindelbeck, R. *et al*. (2017). Dynamic model improves agronomic and environmental outcomes for maize nitrogen management over static approach. *Journal of Environmental Quality* 46, 311-319. https://doi.org/10.2134/jeq2016.05.0182.

Sheahan, M. and Barrett, C. (2017). Ten striking facts about agricultural input use in Sub-Saharan Africa. *Food Policy* 67, 12-25. https://doi.org/10.1016/j.foodpol.2016.09.010.

Singh, B. (2016). *Agronomic Benefits of Neem Coated Urea – A Review. Technical Report*. https://www.researchgate.net/publication/309384701_Agronomic_Benefits_of_Neem_Coated_Urea_-_A_Review.

Smits, M. and Woltjer, G. (2018). *Phosphorus Recycling from Manure: A Case Study on the Circular Economy. Circular Impacts*. https://circular-impacts.eu/sites/default/files/D4.5_Case-Study-Nutrient-Recycling_FINAL.pdf.

Snyder, C., Davidson, E., Smith, P. and Venterea, R. (2014). Agriculture: Sustainable crop and animal production to help mitigate nitrous oxide emissions. *Current Opinion in Environmental Sustainability* 9-10, 46-54. https://doi.org/10.1016/j.cosust.2014.07.005.

Soe, J., Lwin, H.Y., The, N.E.M. and Mar, S.S. (2016). Fertilizer supply chain and market structure: a case study in Tatkon Township, Nay Pyi Taw. *Journal of Agricultural Research* 3, 90-97. https://www.cabi.org/gara/abstract/20173139029.

Spokas, K., Cantrell, K., Novak, J., Archer, D., Ippolito, J., Collins, H. *et al*. (2012). Biochar: A synthesis of Its agronomic impact beyond carbon sequestration. *Journal of Environmental Quality* 41, 973-989. https://doi.org/10.2134/jeq2011.0069.

Springmann, M., Clark, M., Mason-D'Croz, D., Wiebe, K., Bodirsky, B., Lassaletta, L. *et al*. (2018). Options for keeping the food system within environmental limits. *Nature* 562, 519-525. https://doi.org/10.1038/s41586-018-0594-0.

Stevenson, J., Villoria, N., Byerlee, D., Kelley, T. and Maredia, M. (2013). Green Revolution research saved an estimated 18 to 27 million hectares from being brought into agricultural production. *Proceedings of the National Academy of Sciences of the United States of America* 110(21), 8363-8368. https://doi.org/10.1073/pnas.1208065110.

Stewart, Z.P., Pirzynski, G.M., Middendorf, B.J. and Prasad, P.V. (2020). Approaches to improve soil fertility in sub-Saharan Africa. *Journal of Experimental Botany* 71(2), 7 632-641. https://doi.org/10.1093/jxb/erz446.

Sustainable Organic Agriculture Action Network (2013). *Best Practice Guideline for Agriculture and Value Chains. Public version 1.0*. http://www.fao.org/3/a-ax270e.pdf.

Sutton, M., Bleeker, A., Howard, C., Bekunda, M., Grizzetti, B., de Vries, W. *et al*. (2013). *Our Nutrient World: The Challenge to Produce More Food and Energy with Less Pollution. Global Overview of Nutrient Management*. Edinburg: Centre for Ecology and Hydrology (CEH) on behalf of the Global Partnership on Nutrient Management and the International Nitrogen Initiative. http://nora.nerc.ac.uk/id/eprint/500700/1/N500700BK.pdf.

Syers, J., Johnson, A. and Curtin, D. (2008). *Efficiency of Soil and Fertilizer Phosphorus Use: Reconciling Changing Concepts of Soil Phosphorus Behaviour with Agronomic Information*. Rome: FAO Fertilizer and Plant Protection Bulletin. http://www.fao.org/3/a1595e/a1595e00.pdf.

Teenstra, E., De Buisonjé, F., Ndambi, A. and Pelster, D. (2015). *Manure Management in the (Sub-)Tropics; Training Manual for Extension Workers. Livestock Research Report 919*. Wageningen, the Netherlands: Wageningen UR (University and Research Centre) Livestock Research. https://ccacoalition.org/en/resources/manure-management-training-manure-management-sub-tropics-manual-extension-workers.

The Economic Times (India) (2019). IFFCO launches nano-tech based fertilisers for on-field trials, 3 November. https://economictimes.indiatimes.com/news/economy/agriculture/iffco-launches-nano-tech-based-fertilisers-for-on-field-trials/articleshow/71878538.cms?from=mdr. Accessed 20 October 2020.

The Fertilizer Institute (2016). State of the industry. Fertilizer transportation. https://www.tfi.org/our-industry/state-of-industry-archive/2016/fertilizer-transportation. Accessed 21 October 2020.

The Fertilizer Institute (2017). Nutrient stewardship. https://nutrientstewardship.org/4rs/. Accessed 21 October 2020.

Thompson, N., Bir, C., Widmar, C. and Mintert, J. (2019). Farmer perceptions of precision agriculture technology benefits. *Journal of Agricultural and Applied Economics* 51(1), 142-163. https://doi.org/10.1017/aae.2018.27.

Ti , C., Xia, L., Chang, S. and Yan, X. (2019). Potential for mitigating global agricultural ammonia emission: A meta-analysis. *Environmental Pollution* 245, 141-148. https://doi.org/10.1016/j.envpol.2018.10.124.

Tiba, Z. (2011). Targeting the most vulnerable: implementing input subsidies. In *Safeguarding Food Security in Volatile Global Markets*. A. Prakash (ed.) Rome: FAO. 529-542. http://www.fao.org/3/i2107e/i2107e.pdf.

Tigabu, D. and Gebeyehu, M. (2018). *Agricultural Extension Service and Technology Adoption for Food and Nutrition Security: Evidence from Ethiopia*. FARA Research Report 3(4). Accra: Forum for Agricultural Research in Africa (FARA). https://library.faraafrica.org/wp-content/uploads/2019/10/FRR-Volume-3-No-4_2018-Agricultural-Extension-Service-and-Technology-Adoption-for-Food-and-Nutrition-Security-Evidence-from-Ethiopia.pdf.

Torre, L., Battaglia, V. and Caradonia, F. (2016). An overview of the current plant biostimulant legislations in different European Member States. *Journal of the Science of Food and Agriculture* 96, 727-734. https://doi.org/10.1002/jsfa.7358.

Trendov, N.M., Varas, S. and Zeng, M. (2019). *Digital Technologies and Agriculture in Rural Areas*. Status Report. Rome: FAO. http://www.fao.org/3/ca4985en/ca4985en.pdf.

Trenkel, M.E. (2010). *Slow- and Controlled-Release and Stabilized Fertilizers: An Option for Enhancing Nutrient Use Efficiency in Agriculture*. International Fertilizer Industry Association. https://www.fertilizer.org/images/Library_Downloads/2010_Trenkel_slow%20release%20book.pdf.

Tri-State Livestock News (2014). Rail board addresses fertilizer transportation concerns. https://www.tsln.com/news/rail-board-addresses-fertilizer-transportation-concerns/. Accessed 1 November 2020.

Trivedi, P., Schenk, P., Wallenstein, M. and Singh, B. (2017). Tiny microbes, big yields: Enhancing food crop production with biological solutions. *Microbial Biotechnology* 10, 999-1003. https://doi.org/10.1111/1751-7915.12804.

Tsujimoto, Y., Rakotoson, T., Tanaka, A. and Saito, K. (2019). Challenges and opportunities for improving N use efficiency for rice production in sub-Saharan Africa. *Plant Production Science* 22(4), 413-427. https://doi.org/10.1080/1343943X.2019.1617638.

United Nations Conference on Trade and Development (2018). *Achieving the Sustainable Develoment Goals in the Least Developed Coutries*. *A Compedium of Policy Options*.. https://unctad.org/system/files/official-document/aldc2018d4_en.pdf.

United Nations Economic Commission for Africa and African Development Bank (2018). *African Fertilizer Financing Mechanism AFMM Study (Sponsored by UNECA and AFMM) on Promotion of Fertilizer Production, Cross-Border Trade and Consumption in Africa*. https://www.afdb.org/fileadmin/uploads/afdb/Documents/Generic-Documents/Study__sponsored_by_UNECA___AFFM__on_promotion_of_fertilizer_production__cross-border_trade_and_consumption_in_Africa.pdf.

United Nations Environment Programme (2019). *Frontiers 2018/2019: Emerging Issues of Environmental Concern*. https://www.unenvironment.org/resources/frontiers-201819-emerging-issues-environmental-concern.

United Nations Environment Programme and Forum for Law, Environment, Development and Governance (2018). *Voluntary National Reviews: The Environmental Dimension*. *Policy Brief*. https://wedocs.unep.org/bitstream/handle/20.500.11822/25766/VNR-Brief.pdf?sequence=1&isAllowed=y.

United States Environmental Protection Agency (2015). *Chemical Advisory: Safe Storage, Handling and Management of Solid Ammonium Nitrate Prills*. https://www.epa.gov/sites/production/files/2015-06/documents/an_advisory_6-5-15.pdf.

United States Geological Survey (2020). *Mineral Commodities Summaries 2020*. https://pubs.usgs.gov/periodicals/mcs2020/mcs2020.pdf.

University of Tennessee (1999). *Fertilizers and Their Use*. Agricultural Extension Service. https://trace.tennessee.edu/cgi/viewcontent.cgi?article=1051&context=utk_agexcrop.

van Grinsven, H., Bouwman, L., Cassman, K., van Es, H., McCrackin, M. and Beusen, A. (2015). Losses of ammonia and nitrate from agriculture and their effect on nitrogen recovery in the European Union and the United States between 1900 and 2050. *Journal of Environmental Quality* 356-367. https://doi.org/10.2134/jeq2014.03.0102.

van Vliet, J., de Groot, H., Rietveld, P. and Verburg, P. (2015). Manifestations and underlying drivers of agricultural land use change in Europe. *Landscape and Urban Planning* 133, 24-36. https://doi.org/10.1016/j.landurbplan.2014.09.001.

Vanlauwe, B., Chianu, J., Giller, K., Merckx, R., Mokwunye, U., Pypers, P. *et al*. (2010). Integrated soil fertility management: Operational definition and consequences for implementation and dissemination. *Outlook on Agriculture* 39(1), 17-24. https://doi.org/10.5367/000000010791169998.

Voortman, R., Sonneveld, B. and Keyzer, M. (2003). African land ecology: Opportunities and constraints for agricultural development. *Ambio* 32, 376-373. https://doi.org/10.1579/0044-7447-32.5.367.

Wairegi, L., Bennett, M., Nziguheba, G., Mawanda, A., de los Rios, C., Ampaire, E. *et al*. (2018). Sustainably improving Kenya's coffee production needs more participation of younger farmers with diversified income. *Journal of Rural Studies* 63, 190-199. https://doi.org/10.1016/j.jrurstud.2018.07.009.

Wang, Y., Zhu, Y., Zhang, S. and Wang, Y. (2018). What could promote farmers to replace chemical fertilizers with organic fertilizers? *Journal of Cleaner Production* 199, 882-890. https://doi.org/10.1016/j.jclepro.2018.07.222.

Wanzala, M. and Groot, R. (2013). *Fertilizer Market Development in Sub-Saharan Africa*. Leek, UK: International Fertilizer Society. https://fertiliser-society.org/store/fertiliser-market-development-in-sub-saharan-africa/.

Weeks, J. and Hettiarachchi, G. (2019). A review of the latest in phosphorus fertilizer technology: Possibilities and pragmatism. *Journal of Environmental Quality* 48, 1300-1313. https://doi.org/10.2134/jeq2019.02.0067.

Weersink, A., Fraser, E., Pannell, D., Duncan , E. and Rotz, S. (2018). Opportunities and challenges for big data in agricultural and environmental analysis. *Annual Review of Resource Economics* 10, 19-37. https://doi.org/10.1146/annurev-resource-100516-053654.

Weithmann, N., Möller, J., Löder, M., Piehl, S., Laforsch, C. and Freitag, R. (2018). Organic fertilizer as a vehicle for the entry of microplastic into the environment. *Science Advances* 4(4), eaap8060. https://doi.org/10.1126/sciadv.aap8060.

Willer, H., Schlatter, B., Trávníček, J., Kemper, L. and Lernoud, J. (eds.) (20). *The World of Organic Agriculture*. *Statistics and Emerging Trends 2019*. Bonn: Research Institute of Organic Agriculture (FiBL), Frick, and IFOAM – Organics International. https://orgprints.org/id/eprint/37222/9/willer-et-al-2020-full-document-2020-02-28-4th-corrigenda.pdf.

Williams, J. (2019). IFA welcomes the FAO international code of conduct for the sustainable use and management of fertilizers, 1 July. *World Fertilizer Magazine*. https://www.worldfertilizer.com/environment/01072019/ifa-welcomes-the-fao-international-code-of-conduct-for-the-sustainable-use-and-management-of-fertilizers/. Accessed 24 October 2020.

Wohab, M., Gaihre, Y., Ziauddin, A. and Hoque, M. (2017). Design, development and field evaluation of manual-operated applicators for deep placement of fertilizer in puddled rice fields. *Agricultural Research* 6(3), 259-266. https://doi.org/10.1007/s40003-017-0267-5.

Wopereis, M., Haefele, S., Dingkuhn, M. and Sow, A. (2003). Decision support tools for irrigated rice based systems in the Sahel. In *Decision Support Tools for Irrigated Rice-Based Systems in the Sahel: A Practical Guide*. Bontkes, T. and Wopereis, M. (eds.). Muscle Shoals, AL: International Fertilizaer Development Center (IFDC) and Wageningen, the Netherlands: ACP-EU Technical Centre for Agricultural and Rural Cooperation (CTA). 114-126. https://www.researchgate.net/publication/292004995_Decision_support_tools_for_irrgated_rice-based_systems_in_the_Sahel.

World Bank (2014). *Improving Trade and Transport for Landlocked Developing Countries: A Ten-Year Review*. Washington, D.C.: World Bank Group. https://www.worldbank.org/content/dam/Worldbank/document/Trade/Landlocked_Countries.pdf.

World Bank (2019). Prevalence of undernourishment (% of population). https://data.worldbank.org/indicator/SN.ITK.DEFC.ZS?view=chart. Accessed 24 October 2020.

Wu, D., Xu, X., Chen, Y., Shao, H., Sokolowski, E. and Mi, G. (2019). Effect of different drip fertigation methods on maize yield, nutrient and water productivity in two-soils in Northeast China. Agricultural *Water Management* 213, 200-211. https://doi.org/10.1016/j.agwat.2018.10.018.

Wu, Y., Xi, x., Tang, X., Luo, D., Gu, B., Lam, S. *et al*. (2018). Policy distortions, farm size, and the overuse of agricultural chemicals in China. *Proceedings of the National Academy of Sciences of the United States of America* 115, 7010-7015. https://doi.org/10.1073/pnas.1806645115.

Xu, L. and Geelen, D. (2018). Developing biostimulants from agro-food and industrial by-products. *Frontiers in Plant Science* 9(1567), 1-13. https://doi.org/10.3389/fpls.2018.01567.

Yang, F., Xu, X., Ma, J., He, P., Pampolino, M. and Zhou, F. (2017). Experimental validation of a new approach for rice fertiliser recommendations across smallholder farms in China. *Soil Research* 55(6), 579-589. https://doi.org/10.1071/SR16328.

Yang, X. and Fang, S. (2015). Practices, perceptions, and implications of fertilizer use in East-Central China. *Ambio* 44, 647-652. https://doi.org/10.1007/s13280-015-0639-7.

Zamanian, K. and Kuzyakov, Y. (2019). Contribution of soil inorganic carbon to atmospheric CO2: More important than previously thought. *Global Change Biology* 25(1), e1-e3. https://doi.org/10.1111/gcb.14463.

Zingore, S., Mutegi, J., Agesa, B., Tamene, L. and Kihara, J. (2015). Soil degradation in sub-Saharan Africa and crop production options for soil rehabilitation. *Better Crops* 99(1). https://hdl.handle.net/10568/68702.

Zulfiqal, F., Navarro, M., Ashraf, M., Akram, N. and Munné-Bosch, S. (2019). Nanofertilizer use for sustainable agriculture: Advantages and limitations. *Plant Science* 289, 110270. https://doi.org/10.1016/j.plantsci.2019.110270.

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them

Envisioning A Chemical-Safe World

Chapter 8 of 12

The regulatory and policy environment of fertilizer management and use

Contents

1 Global drivers, actors and policies affecting pesticides and fertilizer use

2 Status and trends of pesticide use

3 The regulatory and policy environment for pesticide management

4 Environmental and health effects of pesticide use

5. The environmental, human health and economic impacts of pesticides

6 Current pesticide risk reduction and risk management

7 Status and trends of fertilizer use

About iv

8 The regulatory and policy environment of fertilizer management and use 1

8.1 Overview 1

8.2 International conventions and policy instruments that address fertilizer distribution and use 2

8.2.1 The United Nations Framework Convention on Climate Change 2
8.2.2 The United Nations Convention on Biological Diversity 4
8.2.3 The United Nations Convention on Wetlands 4
8.2.4 The United Nations Convention to Combat Desertification 4
8.2.5 The United Nations Economic Commission for Europe (UNECE) Convention on Long-range Transboundary Air Pollution 5
8.2.6 The UNECE Convention on the Protection and Use of Transboundary Watercourses and International Lakes 6
8.2.7 The UNECE Industrial Accidents Convention 6
8.2.8 The International Code of Conduct for the Sustainable Use and Management of Fertilizers 6
8.2.9 The Codex Alimentarius 7

8.3 International partnerships and initiatives addressing fertilizer distribution and use 8

8.3.1 The Global Soil Partnership 8
8.3.2 The Global Partnership on Nutrient Management 8

8.3.3 The International Nitrogen Initiative 9
8.3.4 The International Nitrogen Management System 9
8.3.5 The Global Phosphorus Research Initiative 10
8.3.6 The Sustainable Rice Platform 10
8.3.7 The Sustainable Agriculture Initiative Platform 10
8.3.8 The International Environmental Product Declaration System 10
8.3.9 The Sustainability Consortium 10
8.3.10 The World Trade Organization 11

8.4 Regional conventions, legislation and policy instruments linked to fertilizer distribution and use 11

8.4.1 Regional partnerships that are part of International initiatives 12
8.4.2 Initiatives specific to regions 12

8.5 National policies and legislation on fertilizers 19

8.5.1 Legislation on inorganic fertilizers 20
8.5.2 Legislation on organic fertilizers 21
8.5.3 Legislation to protect air and water quality 22
8.5.4 Fiscal policy instruments 23
8.5.5 Links between policies and sectors 24

8.6 Corporate responsibility and stewardship 25

8.6.1 4R Nutrient Stewardship 25
8.6.2 The Sustainability Consortium 26

References 27

9 Environmental and health effects of fertilizer use

10 The impact of fertilizer use

11 Current fertilizer risk reduction and risk management

12 Transformative actions to minimize the adverse impacts of pesticide and fertilizers

List of Boxes

Box 8.5-1	Regulating fertilizers	19

List of Figures

Figure 8.2-1	International Conventions with potential to influence fertilizer distribution and use	3
Figure 8.5-1	Number of countries addressing components of manure management in their policies reported in the study.	21
Figure 8.5-2	Amounts of fertilizer used in selected countries in Africa supplied through subsidy programmes and not from subsidized programs (thousands of tonnes).	23
Figure 8.6-1	The 4R Nutrient Stewardship concept.	25

List of Tables

Table 8.4-1	Maximum limits for heavy metals and pathogens in fertilizers and biostimulants in the new EU Fertilising Products Regulation.	14
Table 8.4-2	Progress made on implementing the EU Nitrates Directive.	15

About

In December 2017, Resolution 4 of the 3rd Session of the United Nations Environment Assembly (UNEA 3) requested "the Executive Director to present a report on the environmental and health impacts of pesticides and fertilizers and ways of minimizing them, given the lack of data in that regard, in collaboration with the World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO) and other relevant organizations by the fifth session of the United Nations Environment Assembly". In response to this request, UNEP published a *Synthesis Report on the Environmental and Health Impacts of Pesticides and Fertilizers and Ways to Minimize Them*[1] in February 2022 (United Nations Environment Programme [UNEP] 2022).

The overall goal of the synthesis report is to provide the information base to enable other advocacy actions to be taken by stakeholders to minimize the adverse impacts of pesticides and fertilizers. Specific objectives of the synthesis report are to:

- Update understanding of current pesticide and fertilizer use practices;
- Present major environmental and health effects of pesticides and fertilizers, during their life cycle, and identify key knowledge gaps;
- Review current management practices, legislation and policies aimed at reducing risks in the context of the global chemicals, environmental and health agenda;
- Identify opportunities to minimize environmental and health impacts, including proven and innovative approaches.

This chapter on "The regulatory and policy environment of fertilizer management and use" is the 8th in a series of 12 chapters that make up a comprehensive compilation of scientific information. The chapters were developed to both inform and further elaborate on the information provided in the synthesis report. Please note that the disclaimers and copyright from the synthesis report apply.

1 The Synthesis report is available at https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

The regulatory and policy environment of fertilizer management and use

8

8.1 Overview

The main harmful effects of fertilizer use addressed by international Conventions and policy instruments are greenhouse gas (GHG) emissions, air and groundwater pollution, and biodiversity loss. The 2019 International Code of Conduct for the Sustainable Use and Management of Fertilizers is intended to maximize the benefits of fertilizers and minimize the adverse effects (Food and Agriculture Organization of the United Nations [FAO] 2019a). [Chapter 8.2]

There is a regional imbalance in fertilizer use and in all regions, there are problems related to the misuse of fertilizers. Some regions have problems with overuse and others with underuse or inappropriate use.

There are more international partnerships and initiatives on nitrogen then on phosphorus. There may be a need to strengthen actions on phosphorus, so that they are closer to the level of the International Nitrogen Initiative, in order make phosphorus issues more visible. [Chapter 8.3]

Developed countries focus more on environmental issues than developing ones. There are more environmental initiatives in developed regions and countries; yet some developing regions and countries are major sources of environmental pollution (Alexandratos 1995).

The drive for policy change should distinguish between the needs of developed and developing countries, as their priorities in delivering on the 2030 Agenda for Sustainable Development can differ. Strategies for the implementation of policy instruments may also need to distinguish between developed and developing countries, and between different regions (Campbell *et al*. 2018). [Chapter 8.4]

There is also a need to harmonize regional policies and agreements with national ones (Joss *et al*. 2017; United Nations Economic Commission for Europe and African Development Bank [UNECA and AfDB] 2018; Wanzala-Mlobela *et al*. 2019). Lack of harmonization of standards across countries is a strong indication that national interventions, especially with regard to fertilizers, are still far from ideal (Joss *et al*. 2017; UNEP 2021). [Chapter 8.4.2]

There are gaps in information about the status of fertilizer policies at global, regional and national levels. In many cases, information on the progress made in implementing international and regional conventions at national level is not freely available to the public.

Shortcomings associated with fertilizer regulations exist in both developed and developing countries and national policies do not always cover all stages of the fertilizer life cycle (FAO 2019a). On fertilizer registration, low-income countries have the weakest performance and high-income countries performed best. The development and implementation of policies on fertilizer quality has progressed more in the case of inorganic fertilizers than in that of organic ones (World Bank 2017; World Bank 2019).

Countries have different priorities concerning fertilizer use and risks. They may use different combinations of instruments to achieve their goals. Fertilizers can be subject to legislation and regulations related to production, trade, distribution, marketing, safety and use. It can also be subject to legislation to protect air and water quality (FAO 2019a). [Chapter 8.5]

In some countries there are provisions in fertilizer legislation that aim to minimize environmental damage. In some there is also close collaboration between ministries addressing the food security agenda and the environmental agenda, which provides opportunities to exploit synergies and identify contradictions, overlaps and gaps in legislation. [Chapters 8.5.1, 8.5.2, 8.5.3]

More progress has been made on regulations for inorganic fertilizers than on those for organic fertilizers. Still, many countries have policies related to manure management, use and disposal of sewage sludge. [Chapters 8.5.1, 8.5.2]

The fertilizer industry is making contributions to sustainable nutrient management, for example by developing the 4R Nutrient Stewardship guidelines and helping to establish the Scientific Panel on Responsible Plant Nutrition (Johnson and Bruulsema 2014; International Fertilizer Association [IFA] 2020). [Chapter 8.6.1]

Implementing fertilizer regulations that take environmental concerns into consideration requires some of the tools and approaches discussed in Chapter 7, such as precision farming, decision support tools, and capacity development with regard to fertilizer use and management.

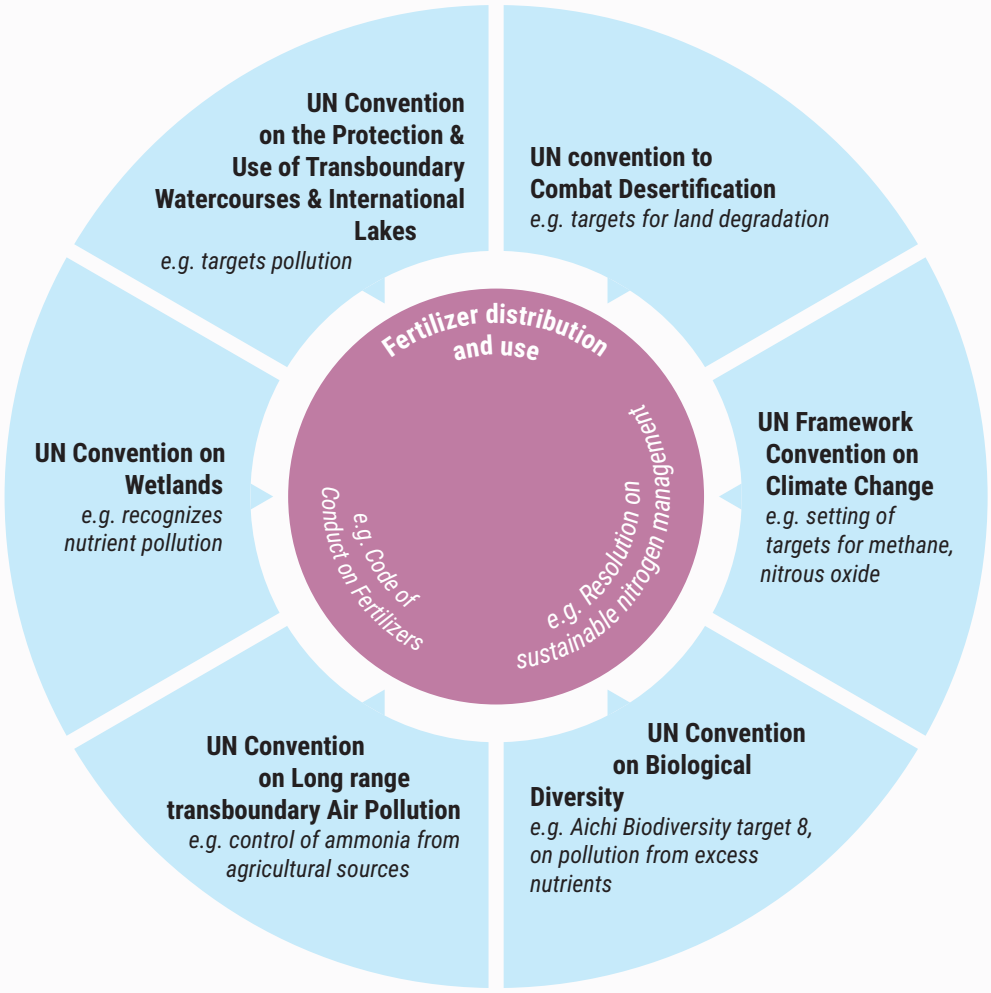
8.2 International conventions and policy instruments that address fertilizer distribution and use

The key Conventions addressing fertilizers (see Figure 8.2-1) are the three Rio Conventions established at the 1992 Earth Summit – the United Nations Framework Convention on Climate Change (UNFCCC), the Convention on Biological Diversity (CBD) and the Convention on Wetlands (the Ramsar Convention) – together with the United Nations Convention to Combat Desertification (UNCCD), the Convention on Long-range Transboundary Air Pollution (including the Gothenburg Protocol), and the Convention on the Protection and Use of Transboundary Watercourses and International Lakes.

There is guidance on the safe transport and storage of potentially explosive fertilizers in the United Nations Economic Commission for Europe (UNECE) Industrial Accidents Convention, which is also described.

The management and use of fertilizers are mentioned, or alluded to, in several other international and regional Conventions and policy instruments. The recently published International Code of Conduct for the Sustainable Use and Management of Fertilizers (the Fertilizer Code) promotes practices that encourage judicious use of fertilizers. The Fertilizer Code addresses the misuse, overuse and underuse of fertilizers.

Figure 8.2-1 International Conventions with potential to influence fertilizer distribution and use



The Codex Alimentarius standards and guidelines on food quality consider fertilizers as potential food and water contaminants.

8.2.1 The United Nations Framework Convention on Climate Change

The 1992 United Nations Framework Convention on Climate Change (UNFCCC) serves as the foundation for international climate agreements. There are 197 Parties to the Convention (United Nations Convention on Climate Change [UNFCCC] 2020a). The ultimate objective of all agreements under this Convention is stabilization of greenhouse gas (GHG) concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate (UNFCCC 2020b).

The Kyoto Protocol to the UNFCCC was adopted in December 1997 and entered into force in February 2005 (UNFCCC 2020c). It set targets for emissions of the main greenhouse gases (carbon dioxide [CO_2], methane (CH_4), nitrous oxide [N_2O], hydrofluorocarbons [HFCs], perfluorocarbons [PFCs], sulphur hexafluoride [SF_6]). Fertilizers, including animal manure, are among the sources of CO_2, CH_4 and N_2O emissions.

The Intergovernmental Panel on Climate Change (IPCC) is the United Nations body responsible for assessing the science related to climate change. It was created to provide policymakers with regular scientific assessments on climate change, its implications, and potential future risks, as well as to present adaptation and mitigation options (Intergovernmental Panel on Climate Change [IPCC] 2020). Some IPCC recommendations target fertilizers. For example, IPCC has suggested the

use of nitrification and urease inhibitors, precision farming, and improved manure management to reduce nutrient losses (Smith *et al*. 2014). A recent IPCC report on the current state of knowledge on climate change included improved fertilizer management and improved manure management among the practices that contribute to climate change adaptation and mitigation in cropland (IPCC 2019).

8.2.2 The United Nations Convention on Biological Diversity

The objectives of the United Nations Convention on Biological Diversity (CBD) are conservation of biodiversity, sustainable use of its components, and fair and equitable sharing of the benefits arising from the utilization of genetic resources (Secretariat of the Convention on Biological Diversity [CBD Secretariat] 2020a). The CBD covers ecosystems, species and genetic resources. Its Cartagena Protocol on Biosafety aims to ensure safe handling, transport and use of living modified organisms (LMOs) resulting from modern biotechnology that could have adverse effects on biological diversity, also taking into account risks to human health (CBD Secretariat 2000).

The CBD has the potential to influence fertilizer management. In the Strategic Plan for Biodiversity 2011-2020, Aichi Biodiversity Target 8 is: "By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity" (CBD Secretariat 2020b).

According to a scientific report on progress towards the Aichi targets (Leadley *et al*. 2014) which provided scientific evidence for the fourth edition of the *Global Biodiversity Outlook* (CBD Secretariat 2014), most countries had established national targets, or similar objectives related to Aichi Biodiversity Target 8 on pollution, in their national biodiversity strategies and action plans, with the majority of national targets referring to reduction of pollution generally and few referring specifically to reduction of excess nutrients.

A more recent report by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services [IPBES] 2019), intended to provide scientific evidence for the fifth edition of the *Global Biodiversity Outlook* (CBD Secretariat 2020c), concludes that some progress has been made on implementing policy responses and actions to conserve biodiversity. However, it emphasizes that the state of nature has continued to decline while progress on Aichi Target 8 and five other Aichi targets remains poor. According to this report, nutrient pollution continues to have adverse impacts on ecosystem functions and biodiversity despite increasing efforts to improve fertilizer use.

The Conference of the Parties to the CBD, at its 15th meeting in 2021, is expected to adopt a Post-2020 Global Biodiversity Framework which, among other things, will address all forms of pollution including excess nutrients.

The CBD is implemented in partnership with other biodiversity-related Conventions such as the UNCCD and the Convention on Wetlands (see below).

8.2.3 The United Nations Convention on Wetlands

The United Nations Convention on Wetlands (also known as the Ramsar Convention) provides the framework for national action and international cooperation with regard to the conservation and wise use of wetlands and their resources (Ramsar Convention on Wetlands 2018, 2020). This ensures that the benefits of wetlands contribute towards meeting the United Nations Sustainable Development Goals (SDGs), the Aichi Biodiversity Targets, the Paris Agreement on Climate Change (which entered into force in 2016), and other related international commitments. The Ramsar Convention recognizes that nutrient loads are among the major drivers of wetland degradation. As the only international treaty focused on wetlands, it provides a platform for collaboration and partnership in support of functional wetlands.

8.2.4 The United Nations Convention to Combat Desertification

The United Nations Convention to Combat Desertification (UNCCD), established in 1994, is a legally binding international agreement that links environment and development to sustainable land management. The Convention has a 2018-2030 Strategic Framework (United Nations Convention to Combat Desertification [UNCCD] 2017a). Implementation of the UNCCD is carried out at regional and national levels. Most countries have developed and submitted National Action Plans; alignment of sub-regional and regional action programmes has also been initiated (UNCCD 2020).

To disseminate sustainable land management (SLM) best practices, the UNCCD Secretariat works alongside the World Overview of Conservation Approaches and Technologies (WOCAT) to provide a network and database where specialists can share their knowledge about SLM practices and technologies (UNCCD 2016).

One of the key messages of the UNCCD *Global Land Outlook* (UNCCD 2017b) is that our food system focuses on short-term production and profit rather than on long-term environmental sustainability. The modern agricultural system has resulted in huge increases in productivity, holding off the risk of famine in many parts of the world. At the same time, it is based on monoculture, genetically modified crops, and intensive use of fertilizers and pesticides, all of which undermine long-term sustainability

The climate change mitigation and adaptation technologies considered to have the greatest impact, according to a report by UNCCD's Science-Policy Interface (SPI) (Sanz et al. 2017), include integrated soil fertility management practices (e.g., precision farming) and the use of both organic and inorganic fertilizers. This is a strong indication of the perceived role of fertilizers in sustainable land management.

8.2.5 The United Nations Economic Commission for Europe (UNECE) Convention on Long-range Transboundary Air Pollution

The Convention on Long-range Transboundary Air Pollution (LRTAP), which came into force in 1983, is a legally binding instrument on air pollution established within the framework of the United Nations Economic Commission for Europe (United Nations Economic Commission for Europe [UNECE] 1999). The Parties to the Convention are countries in North America, Europe and Central Asia.

There are eight protocols to this Convention concerning measures to reduce transboundary air pollution (UNECE 2019a). Those with the potential to influence fertilizer management and use include the Protocol concerning the Control of Emissions of Nitrogen Oxides (UNECE n.d.) and the Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (the Gothenburg Protocol) (UNECE 1999), which addresses four pollutants including ammonia and NOx.

The Task Force on Reactive Nitrogen (TFRN) has the long-term goal of developing technical and scientific information as well as options which can be used for strategy development across the UNECE to encourage coordination of air pollution policies on nitrogen in the context of the nitrogen cycle, and which may be used by other bodies outside the Convention, in consideration of other control measures (Convention on Long-range Transboundary Air Pollution Task Force on Reactive Nitrogen [CLRTAP-TFRN] 2020).

The Gothenburg Protocol to the LRTAP is intended to abate acidification, eutrophication and tropospheric (or ground level) ozone. It sets limits for sulphur, NOx, volatile organic compounds (VOCs) and ammonia and includes guidelines for measures to control ammonia emissions from agricultural sources (fertilizers, manures and animal housing) (UNECE 1999). A document prepared by TFRN to provide guidance to the Parties to the LRTAP Convention in identifying ammonia control measures for reducing emissions from agriculture addresses ammonia emission reduction measures in fertilizer and

manure application, along with manure storage techniques (Bittman et al. 2014).

Many Parties to the Convention are starting to put in place national legislation, especially to meet committed emissions ceilings (CLRTAP-TFRN 2020). Failure by countries to meet their ceilings for ammonia are likely due mainly to failure to reduce emissions related to fertilizers and manures. For example, in 2017 not all European Union (EU) Member States were in compliance with their ammonia ceilings and an overall increase in ammonia emissions of 2.5 per cent, between 2014 and 2017, was attributed to the agriculture sector (European Environment Agency 2019).

The revised Gothenburg Protocol, which is yet to enter into force, has been extended to include particulate matter ($PM_{2.5}$) among the pollutants covered (AirClim 2019). Since ammonia is one of the precursors of $PM_{2.5}$, reducing ammonia emissions from fertilizer and manure will contribute to reduced $PM_{2.5}$. (The relationship between ammonia, $PM_{2.5}$ and human health is discussed in Chapter 9.)

8.2.6 The UNECE Convention on the Protection and Use of Transboundary Watercourses and International Lakes

The UNECE Convention on the Protection and Use of Transboundary Watercourses and International Lakes (the Water Convention) entered into force in 1996 (UNECE 2020a). It strengthens transboundary water cooperation and measures for the ecologically sound management and protection of transboundary surface waters and groundwaters. In 2003 the Convention was amended to allow accession by countries outside the UNECE region. The amended Convention entered into force in 2013. Most countries in Europe and three countries in Africa have acceded to the Convention and many other countries have begun accession processes.

The Convention allows for the establishment of joint bodies. Among the tasks of such bodies is the development of concerted action programmes for the reduction of pollution loads from both point sources (e.g., municipal and industrial sources) and diffuse sources (particularly from agriculture) Fertilizers are among the main agricultural sources of diffuse pollution.

Under the Convention the Protocol on Water and Health and the Protocol on Civil Liability were adopted in 1999 and 2003, respectively (Transboundary Water Resources Management in Southeastern Europe [TWRM] 2012). The objective of the Protocol on Water and Health is to promote the protection of human health and well-being, within a framework of sustainable development, through improving water management including the protection of water ecosystems, and through preventing, controlling and reducing water-related disease (United Nations Economic Commission for Europe and World Health Organization [UNECE and WHO] 1999). Under this Protocol, Parties are to take measures to protect water resources from pollution from agriculture and other sources. Each Party has the obligation to establish and publish its national targets and the respective target dates for each target area within two years of becoming a Party. Parameters for which targets have been developed in some countries include nitrates, microbes, and pesticides in drinking water (UNECE 2019b).

8.2.7 The UNECE Industrial Accidents Convention

The UNECE's Convention on the Transboundary Effects of Industrial Accidents (the Industrial Accidents Convention) provides principles and guidance materials for all countries to help ensure industrial safety and prevent major industrial accidents (UNECE 2020b). It specifically covers the storage and handling of hazardous substances, including ammonium nitrate (NH_4NO_3). However, there is a lack of reports on implementation and on the number of countries with policies targeting transport and storage of fertilizers such as ammonium nitrate and perhaps calcium ammonium nitrate (CAN).

8.2.8 The International Code of Conduct for the Sustainable Use and Management of Fertilizers

The International Code of Conduct for the Sustainable Use and Management of Fertilizers (the Fertilizer Code) was developed by the

Global Soil Partnership (GSP) (see Chapter 8.4) in response to a request by the Committee on Agriculture of the Food and Agriculture Organization of the United Nations (FAO) to increase food safety and the safe use of fertilizers (FAO 2019a). It was also developed in response to the third United Nations Environment Assembly (UNEA-3) declaration on soil pollution, while ensuring enhanced support to the implementation of the Voluntary Guidelines for Sustainable Soil Management (VGSSM) (FAO 2017a).

The Fertilizer Code addresses issues of global importance, thereby contributing to the implementation of the United Nations Sustainable Development Goals (SDGs). It is intended to maximize the benefits of fertilizers and minimize their negative effects by providing a framework and a set of recommended practices for stakeholders involved with fertilizers (FAO 2019a).

Stakeholders addressed by the Fertilizer Code include governments, policymakers, the private sector, researchers, academia and farmers.

Practices promoted by the Fertilizer Code include nutrient recycling, and agronomic and land management to improve soil health; it recommends regulation related to the sale, distribution and labelling of fertilizer products wherever appropriate (FAO 2019a). The Fertilizer Code also promotes capacity development and education programmes for all stakeholders involved in the fertilizer value chain, and encourages developed countries to assist others in developing infrastructures and capacity to manage fertilizers throughout their life cycle. An expected outcome of the Fertilizer Code is the effective and efficient use of fertilizers to meet agricultural demands while minimizing nutrient losses to the environment.

The Fertilizer Code was endorsed by FAO member countries at the 41st session of the FAO Conference in June 2019. Its principles will first be widely disseminated at the global level. The second step in implementation will involve national capacity development to strengthen technical and local extension services in order to adopt sustainable use and management of fertilizers. This will be followed by assessing countries' capacities for soil fertility management and fertilizer use; organizing trainings on sustainable use and management of fertilizers for extension services and governmental staff, retailers and agro-dealers; implementation of the Global Soil Doctors Programme for building the capacity of farmers on the sustainable management of soil fertility (FAO 2020a); implementing the activities of the Global Soil Laboratory Network on the harmonization of methods and data on soil fertility assessment and fertilizer quality control (FAO 2020b); and establishing balanced/precision fertilization recommendations. The last steps will involve strengthening the national and regional fertilizer quality control, regulatory systems and supply/distribution chains.

In the field, case studies will be established in priority countries with different issues related to fertilizers (overuse, underuse and inappropriate use) in order to understand how to better advise on the sustainable use and management of fertilizers at the national level before upscaling in countries/regions with similar issues.

8.2.9 The Codex Alimentarius

The Codex Alimentarius Commission is an intergovernmental body established by FAO and the World Health Organization (WHO) to protect the health of consumers and help ensure fair practices in the food trade (Codex Alimentarius 2020). It promotes coordination of all food standards work undertaken by international governmental and non-governmental organizations (NGOs), as well as promoting sustainable agriculture.

There are Codex Alimentarius guidelines for levels of contaminants and toxins in foods, and the production, nutrition and labelling of produced foods. Fertilizer is a potential source of some of the contaminants covered within the scope of the Codex. For example, guidance is provided on maximum cadmium levels for selected vegetables, pulses, cereals and sea foods, as well as chocolate (Codex Alimentarius 2019). Fertilizer may contain cadmium as a contaminant.

Inclusion on food labels of information about the environmental damage that can result from food production would make consumers more aware of the "pollution footprint" of their food choices.

The Codex Alimentarius, together with other initiatives, can help drive changes in the labelling of food products.

8.3 International partnerships and initiatives addressing fertilizer distribution and use

In this section a number of international partnerships and initiatives that have the potential to influence fertilizer distribution and use are described.

8.3.1 The Global Soil Partnership

The Global Soil Partnership (GSP), whose Secretariat is hosted by FAO, was established in 2012 (FAO 2020c). Global soil issues are addressed by multiple stakeholders within the GSP. The GSP structure includes the Intergovernmental Technical Panel on Soils, which provides scientific and technical advice and guidance on global soil issues to the GSP; Regional Soil Partnerships, which build on existing regional networks or collaborative processes and provide guidance on regional goals/priorities and their required implementation mechanisms; and a Plenary Assembly in which member countries and other partners decide on the sustainable soil management agenda (FAO 2020d).

The main achievements of the GSP are the Revised World Soil Charter (FAO 2015a), a report on the *Status of the World's Soils* (FAO 2015b), the Voluntary Guidelines for Sustainable Soil Management (VGSSM) (FAO 2017a), the Global Soil Organic Carbon Map (FAO 2017b), the International Code of Conduct for the Sustainable Use and Management of Fertilizers (FAO 2019), the Global Soil Laboratory Network (2020b), the Global Soil Information System (FAO 2020e), World Soil Day (FAO 2021), the development of a database on soil-related legal instruments called SoiLEX (FAO 2020f), RECSOIL (recarbonization of global soils) (FAO 2020g), among others.

8.3.2 The Global Partnership on Nutrient Management

The Global Partnership on Nutrient Management (GPNM) was formed in response to the need to reduce the amount of excess nutrients in the global environment, consistent with global development. It reflects a need for global advocacy and provides a platform for stakeholders to forge a common agenda on nutrient management (Global Partnership on Nutrient Management [GPNM] 2014). The GPNM provides support for the development and sharing of knowledge and the expansion of global and regional partnerships, particularly through the existing regional-level Nutrient Management Platforms and through planned regional GPNM platforms (GPNM n.d.).

GPNM members include the International Nitrogen Initiative (INI), the South Asia Cooperative Environment Programme (SACEP), the UK-China Sustainable Agriculture Innovation Network (SAIN), and private sector or industry organizations such as the International Fertilizer Association (IFA).

GPNM is one of the three partnerships established under the *Global Programme of Action for the Protection of the Marine Environment from Land-based Activity (GPA)*. The purpose of the GPA is to provide guidance to regional and national authorities on the prevention, reduction, control and/or elimination of marine degradation from activities carried out on land. The contaminants considered include sewage, heavy metals and nutrients (United Nations Environment Programme [UNEP] 2020a). The GPA is coordinated by UNEP and implemented primarily by governments, in partnership with all stakeholders.

Global TraPs was launched in 2011 and ended in 2015 (Global TraPs 2015). Its aim was to identify technologies and policy options needed to ensure that future phosphorus use is sustainable, improve food security and environmental quality, and provide benefits to the poor. This project was designed to provide a stakeholder forum. Through its framework academia, NGOs, advocacy groups and the private sector engaged in a multi-stakeholder dialogue. At the end of the project, in order to properly utilize the project's achievements, a proposal was made to turn it into a more permanent structure under the UNEP Global Partnership for Nutrient Management (GPNM) in 2014.

8.3.3 The International Nitrogen Initiative

The International Nitrogen Initiative (INI) is a scientific body established to review the current understanding of the nitrogen cycle and to identify options for optimizing the management of reactive nitrogen globally while minimizing its negative effects on human health and the environment (International Nitrogen Initiative [INI] 2020). INI was created and is sponsored by the Scientific Committee of Problems of the Environment (SCOPE) and the International Geosphere-Biosphere Programme (IGBP). It has regional centres in North America, Latin America, Africa, Europe, East Asia and South Asia.

To address the challenges associated with nitrogen and phosphorus, the members of INI adopted the Kampala Statement for Action on Reactive Nitrogen in Africa and Globally during the sixth International Nitrogen Conference held in Uganda in 2013 (INI 2014). The issues highlighted in the statement of action for sub-Saharan countries from that meeting included: improving soil fertility status; improving nutrient use and supply; acting on nutrient and fertilizer policy; and reducing the contribution of nitrogen to the degradation of water bodies and to air pollution. Globally, the key messages included: improving nitrogen management; reducing nitrogen losses from agriculture; reducing nitrogen losses from the industry, transport and energy sectors; improving waste treatment; and informing individuals and institutions.

8.3.4 The International Nitrogen Management System

The International Nitrogen Management System (INMS) is emerging through an international project supported by the Global Environment Facility (GEF) (United Nations Environment Assembly [UNEA] 2019a), implemented by UNEP, and executed by INI and its partners through the Centre for Ecology in the United Kingdom (INMS 2020a). This project will bring together the science community, the private sector and civil society to gather and synthesize evidence that can support international policy development to improve global nitrogen management (INMS 2020b). It is a key opportunity to pull together a global and critical mass of science evidence on the nitrogen cycle, and to develop a sustained process that gets science, governments, businesses and civil society working together to build common understanding and deliver real change.

In March 2019, Resolution 4/14 on sustainable nitrogen management, which calls for coordination of nitrogen policies from national to global levels, and for sharing and making available existing information and knowledge in the development of evidence-based decision-making towards sustainable nitrogen management, was tabled by India and adopted by the fourth session of the United Nations Environment Assembly (UNEA 2019b).

Thereafter, the fourth meeting of the INMS (INMS-4) held in April 2019 allowed for follow-up on the resolution (INMS 2019). It recognized the many opportunities for better nitrogen management associated with water, air, climate, biodiversity, soils, stratospheric ozone, food and energy, including the need to develop improved coordination between relevant multilateral environmental agreements. The meeting supported the proposal in the UNEA-4 nitrogen resolution to establish an Inter-convention Nitrogen Coordination Mechanism and identified next steps (INMS 2019).

In October 2019, UN Member States endorsed a proposed roadmap for action on nitrogen challenges, the Colombo Declaration on

Sustainable Nitrogen Management (the Colombo Declaration) (UNEP 2019). A key goal of the Colombo Declaration is to halve nitrogen waste by 2030. As part of the Declaration, more than 30 countries endorsed UN plans for a campaign on sustainable nitrogen management called Nitrogen for Life, which stems from the Sustainable Nitrogen Management Resolution. The Declaration urges countries to conduct comprehensive assessments of nitrogen cycling, covering policy, implementation, regulation, and scientific aspects at national level and to sensitize citizens to understand the natural nitrogen cycle and how human impacts alter its balance.

8.3.5 The Global Phosphorus Research Initiative

The Global Phosphorus Research Initiative (GRPI) was founded in 2008 by researchers at the University of Technology in Sydney, Australia, and Linköping University in Sweden (Global Phosphorus Research Initiative [GPRI] 2020). The initiative is a collaboration between independent research institutes in Europe, Australia and North America. This initiative undertakes research on global phosphorus security for future food production. It also facilitates networking, dialogue and awareness-raising among policymakers, industry, scientists and communities on the implications of global phosphorus scarcity and possible sustainable solutions. In the past decade or two concerns have been expressed about the possible depletion of the world's non-renewable phosphate rock resources. However, thorough estimates of world phosphate rock supply indicate that a phosphorus crisis is not imminent (van Kauwenbergh, Stewart and Mikkelsen 2013; United States Geological Survey 2020) (see also Chapter 7.3).

8.3.6 The Sustainable Rice Platform

The Sustainable Rice Platform (SRP) is a multi-stakeholder platform established in 2011 (The Sustainable Rice Platform [SRP] n.d.). The platform is co-convened by UNEP and the International Rice Research Institute (IRRI) to promote resource efficiency and sustainability in trade flows, production and consumption operations and supply chains in the global rice sector.

8.3.7 The Sustainable Agriculture Initiative Platform

The Sustainable Agriculture Initiative (SAI) Platform was created in 2002 by Nestlé, Unilever and Danone (Sustainable Agriculture Initiative Platform [SAI Platform] 2020). The aim of the Platform is to support the development of sustainable agriculture worldwide. The main activities are building capacity for sustainable agriculture and communicating about sustainable agriculture to stakeholders (Sustainable Agriculture Initiative Platform [Sustainable Agriculture Initiative Platform 2020). The Platform has a membership of at least 106 companies and regional initiatives in Australia, Brazil and China. It recommends integrated crop management practices that include careful use of fertilizer. For example, for sustainable production of cereals in Europe, well-balanced fertilization that takes into account soil resources, crop nutrient needs, climatic conditions and surface, groundwater and contamination risks is recommended (Sustainable Agriculture Initiative Platform Platform 2006).

8.3.8 The International Environmental Product Declaration System

The International Environmental Product Declaration (EPD) System (EPD 2020) is a global programme for type III environmental declarations operating in accordance with International Organization for Standardization (ISO) standard ISO 14025. EPDs are based on scientific principles. They are independently verified and registered documents that communicate transparent and comparable information about the life-cycle environmental impact of products, including fertilizers. For example, EPD for arable crops contain information on estimates of direct emissions of ammonia, N2O and NO for different types of inorganic fertilizers and for animal manure (EPD 2013). The EPD System website has a library of EPDs and Product Category Rules (PCRs) that are available to the public. The programme is open for companies and organizations in any country.

8.3.9 The Sustainability Consortium

The Sustainability Consortium was created in 2009 by Walmart, the University of Arkansas and Arizona State University in the United States (The Sustainability Consortium [TSC] 2020). The Consortium is a global non-profit organization working to transform the consumer goods industry by partnering with leading companies to define, develop, and deliver more sustainable products. The consortium offers tools and services for use in addressing product sustainability, for both buyers and suppliers. Among the tools the Consortium has available on its website are key performance indicators for use in measuring the sustainability of coffee production e.g., on fertilizer use and other factors. In addition, the Consortium shares information, for example on best practices in agricultural data collection and field-level sustainability projects.

8.3.10 The World Trade Organization

Many countries are either signatories of WTO agreements or are under observer status (UNECA and AfDB 2018). The WTO discourages protectionist policies and countries are encouraged to open up their domestic markets to goods and services from other countries.

Enforcement of the WTO agreements could contribute to reduction in amounts of fertilizers used. Policies which support domestic prices, or subsidize production can encourage high production of crops. For example, the increase in production of rice, maize and wheat in China has been partially attributed to the market price support policy (Yu 2017). The WTO Agriculture Agreement encourages the use of national policies that cause minimal trade distortion. (WTO n.d). However, it is more lenient in the case of developing and least developed countries than in that of developed countries. For example, developing countries do not have to reduce their subsidies or tariffs as much as developed countries. Since farmers' decisions on crop management are to a large extent based on expected profits, removal of price support policies could result in reduced production and reduced use of inputs.

In 2016 the United States complained that the domestic support China offered to agricultural producers, particularly those producing rice, maize and wheat, appeared to be inconsistent with the WTO Agreement on Agriculture. In early 2019 the WTO ruled in favour of the complaint by the United States and gave China up to 31 March 2020 to implement the proposed recommendations (Hopewell 2019). Scrutiny by other countries of the economic support China provides to agricultural producers will likely contribute to lower producer prices in China and could contribute to a reduction in fertilizer use of. Currently, however, the key factor expected to contribute to reduced growth in the use of inorganic fertilizer in China is the zero growth environmental policy of 2015, which targets both pesticides and fertilizers (Jin and Zhou 2018).

8.4 Regional conventions, legislation and policy instruments linked to fertilizer distribution and use

Developed countries tend to give greater priority to environmental issues than do many developing countries, and most of the former have taken measures to reduce or prevent threats to the environment associated with agriculture. However, since developing countries are generally food insecure and have high poverty levels, environmental measures are commonly in direct competition for scarce resources (Alexandratos 1995).

There is an increasing need to achieve the Sustainable Development Goals (SDGs) associated with environmental issues and those associated with food security and livelihoods (Campbell *et al*. 2018). However, trade-offs may need to be

made between policies targeting environmental protection and those targeting food security. For example, in many developing countries there is need to increase use of nitrogen to meet Sustainable Development Goal (SDG) 1 (poverty reduction), SDG 2 (zero hunger) and SDG 3 (health), but too much nitrogen pollutes surface and groundwater and increases the need to meet SDGs 6 (clean water and sanitation), 13 (climate change), 14 (life below water) and 15 (life on land) (Campbell *et al*. 2018).

This section discusses regional partnerships that are part of international initiatives, followed by regional initiatives specific to Africa, Europe, North America, Latin America and Asia.

8.4.1 Regional partnerships that are part of International initiatives

Regional Soil Partnerships

Seven Regional Soil Partnerships (RSPs) covering Africa, Asia, Europe, Latin America and the Caribbean, the Pacific, the Near East and North Africa, and North America were formed between 2012 and 2019. The membership comprises stakeholders in these regions. The RSPs provide guidance on regional goals/priorities and their required implementation mechanisms. They also facilitate links with national and local soil management programmes. The Partnerships work in collaboration with FAO Regional Offices (FAO 2022).

International Nitrogen Initiative regional centres

The International Nitrogen Initiative is coordinated by a steering committee, led by a chair and six regional centre directors representing, Africa, Europe, Latin America, North America, South Asia and East Asia (INI n.d.). The regional centres are tasked with assessing, creating awareness of, and developing solutions to nitrogen issues in each region. They work in collaboration with relevant research institutions and universities in the regions. The triennial INI conferences held in 2004 (China), 2007 (Brazil), 2010 (India), 2013 (Uganda) and 2016 (Australia) produced nitrogen declarations that summarize and update the scientific evidence in order to advocate international action on sustainable nitrogen management. The first European Nitrogen Assessment was published by the European regional centre of INI and its partners in 2011. The Indian Nitrogen Group and the South Asian Nitrogen Centre of INI published the Indian Nitrogen Assessment in 2017. The International Nitrogen management System (INMS) is working towards publication of the first International Nitrogen Assessment by 2021.

8.4.2 Initiatives specific to regions

Africa

At the continental level, the African Union (AU) is the leading organization that influences national decisions. In the 2006 *Abuja Declaration on Fertilizer for the African Green Revolution*, AU Member States resolved to increase annual fertilizer use on cultivated land in Africa from 8 kg to 50 kg of nutrients per hectare by 2015 (New Partnership for Africa's Development [NEPAD] 2011). The African Development Bank (AfDB) was charged with the responsibility of establishing and managing the Africa Fertilizer Financing Mechanism (AFFM). The key function of AFFM was to provide financing required to debottleneck the use of fertilizers. The amount of NPK fertilizer applied on cultivated land in SSA has doubled from 8 kg to 17 kg of nutrients per hectare (Roy 2019).

Efforts to implement the Declaration by the Regional Economic Communities include developing and adopting resolutions on the harmonization of fertilizer policies and regulations and promoting national and regional fertilizer production and intraregional trade (NEPAD 2011). However, all sub-Saharan economic regions still have some way to go, according to a recent publication by Wanzala-Mlobela *et al*. (2019). The authors point out that in the *Economic Community of West African* States (ECOWAS) 13 countries have developed at least an implementing regulation and two have put in place a registration system. Other regional communities are even further behind in this process.

Again according to Wanzala-Mlobela *et al*. (2019), the *East African Community (EAC)* has guidelines

on how harmonization of fertilizer policies and regulatory frameworks should be undertaken in the region and is in the process of developing legislation to govern the fertilizer sector. The *Common Market for Eastern and Southern Africa (COMESA)* has developed a framework document delineating the key policy areas and recommendations with regard to national fertilizer policy and harmonization. The *Southern African Development Community (SADC)* region started to develop a harmonized system for labelling fertilizers, but progress has slowed.

Apart from economic communities, there are organizations that contribute to sustainable use of fertilizers. For example, the *African Fertilizer and Agribusiness Partnership (AFAP)* supports the agricultural inputs and agribusiness value chain by providing financial assistance to agro-dealers (Africa Fertilizer and Agribusiness Partnership 2019). It promotes the use of high quality and affordable balanced fertilizers, supports dialogue among stakeholders, and provides policy support to governments. Another example is the *African Organisation for Standardisation (ARSO)*. Established by the African Union and the United Nations Economic Commission for Africa (UNECA), it has a key mandate to harmonize standards in Africa (African Organisation for Standardisation 2020).

Europe

The new *EU Fertilising Products Regulation* entered into force in 2019 and will apply as of the summer of 2022 (European Union [EU] 2019; EU 2020). One of its main objectives is to encourage large-scale fertilizer production from domestic organic or secondary raw materials, in line with the circular economy model, by transforming waste into nutrients for crops (European Commission [EC] 2016; European Council 2019).

Unlike the previous Fertilizer Regulation, which succeeded in harmonizing the EU inorganic fertilizer market, the new regulation not only covers inorganic fertilizers, but also organic and organo-mineral fertilizers and microbial plant biostimulants. In addition, it addresses environmental concerns by setting harmonized limits for a range of contaminants (Table 8.4-1) (EU 2019).

The purpose of the EU *Sewage Sludge Directive* of 1986 is to regulate use of sewage sludge in agriculture in such a way as to prevent harmful effects on soil, vegetation, animals and man (EU 1986). The European Commission is currently assessing whether the Directive should be reviewed. It has launched a study to gather existing information on the environmental, economic, social and health impacts of practices of sewage sludge use on land that are currently in use (EC n.d.).

The *European Sustainable Phosphorus Platform (ESPP*) was formed in 2013 through a declaration signed by over 150 organizations (European Sustainable Phosphorus Platform 2019). The Platform ensures knowledge sharing, experience transfer and networking about opportunities in phosphorus management; facilitates discussions between the market, stakeholders and regulators; addresses regulatory obstacles; contributes to policy proposals; circulates information through newsletters, a website, conferences and publications; and promotes platform members' activities. The members represent a wide range of actors across the whole value chain of phosphorus stewardship: phosphorus mining and processing, water and waste treatment, food, feed and agriculture, phosphorus reuse and recycling, innovation and technology providers, knowledge institutions, NGOs, and governmental organizations.

The 1991 EU *Nitrates Directive* is intended to protect water quality in Europe by preventing nitrates from agricultural sources polluting ground and surface waters and by promoting the use of good farming practices (EC 2010). The Nitrates Directive restricts the use of fertilizers and manure in situations where there is a high risk of leaching and run-off, sets limits on the amount of nitrogen in manure applied per hectare of agricultural land (the limit is 170 kg nitrogen per hectare for livestock manure), and promotes balanced fertilization (i.e., balancing nitrogen and phosphorus inputs with plant demand) (van Grinsven et al. 2015).

Table 8.4-1 Maximum limits for heavy metals and pathogens in fertilizers and biostimulants in the new EU Fertilising Products Regulation.

	Organic fertilizers	Organo-mineral fertilizers	Inorganic macronutrient fertilizers	Biostimulants
Contaminants				
Cadmium	1.5 mg/kg dry matter	3 mg/kg dry matter if fertilizer has <5% P_2O_5; 60 mg/kg of dry matter if fertilizer has ≥5% P_2O_5	3 mg/kg dry matter if fertilizer has <5% P_2O_5; 60 mg/kg of dry matter if fertilizer has ≥5% P_2O_5	1.5 mg/kg dry matter
Hexavalent chromium (mg/kg dry matter)	2	2	2	2
Mercury (mg/kg dry matter)	1	1	1	1
Nickel (mg/kg dry matter)	50	50	100	50
Lead (mg/kg dry matter)	120	120	120	120
Inorganic arsenic (mg/kg dry matter)	40	40	40	40
Biuret (g/kg dry matter)	absent	12	12	
Perchlorate (mg/kg dry matter			50	
If these nutrients are not intentionally added				
Copper (mg/kg dry matter)	300	600	600	600
Zinc (mg/kg dry matter	800	1,500	1,500	1,500
Pathogens				
Salmonella spp.	Absence in 25 g or 25 ml	Absence in 25 g or 25ml	Absence in 25 g or 25 ml	Absence in 25 g or 25 ml
Escherichia coli or	1,000 in 1 g or 1 ml	1,000 in 1 g or 1 ml	1,000 in 1 g or 1 ml	Absence in 1 g or 1 ml
Enterococcacea	1,000 in 1 g or 1 ml	1,000 in 1 g or 1 ml	1,000 in 1 g or 1 ml	10 colony forming unit (CFU)/g
Listeria monocytogenes, Vibrio spp., Shingella spp. or Staphylococcus aureus				Absence in 25 g or 25 ml

EU Member States have made progress in implementing this Directive (Table 8.4-2). The existence of infringement procedures against countries (EC 2018) could be contributing to this progress.

Aquatic ecosystems are protected against eutrophication by the *Water Framework Directive* and the *Marine Strategy Framework Directive* (van Grinsven *et al*. 2015).

The Water Framework Directive (WFD), adopted in 2000, requires Member States to achieve good environmental status by 2015 for all water bodies, including marine waters up to one nautical mile from shore (Boesch 2019). The WFD also requires that nutrient concentrations do not exceed the levels established (van Grinsven, Tiktak and Rougoor 2016). Implementation of measures to meet the requirements of the WFD has included the design of river basin management plans (RBMPs) by Member States (van Grinsven, Tiktak and Rougoor 2016). The potential to meet the

Table 8.4-2 Progress made on implementing the EU Nitrates Directive (EC 2018; EC 2019b).

Step	Progress made
Water monitoring for nitrate concentrations and trophic status	Uneven efforts deployed in water monitoring by Member States; large number of new stations with no trends across the EU
Designation of nitrate vulnerable zones (NVZs)	Areas with potential water pollution that are not included in any NVZ exist; sometimes the designed territory is limited to the area around the monitoring stations
Establishment of codes of good agricultural practices (CGAPs) and voluntary implementation of the CGAPs	All members have established CGAPs
Establishment of action programmes	Several members have adopted action programmes at regional level
Review and possible revision of the designation of NVZs and of action programmes at least every four years	NVZs defined in 2003 and changed in 2007 and 2011; review of NVZs done in 2015, and new NVZs declared in 2016
Submission to the European Commission of a progress report every four years	Member States submit reports; the latest was finalized in 2017

Water Framework Directive goals successfully has been associated with mainstreaming water policy into other policy sectors, particularly agricultural policy (Carvalho *et al*. 2019).

The EU *Marine Strategy Framework Directive* (MSFD), adopted in 2008, aimed at achieving and maintaining good environmental status in European seas by 2020.

In 1974, the Baltic Sea countries joined the *Convention on the Protection of the Marine Environment of the Baltic Sea, also known as the Helsinki Convention*. The Baltic Marine Environment Protection Commission, or Helsinki Commission (HELCOM), implements the Convention's plans and directives (Baltic Marine Environment Protection Commission 2020). The contracting parties are Denmark, Sweden, Finland, Germany, Poland, Lithuania, Latvia, Estonia, the Russian Federation and the EU. A key objective of the Commission is to reduce emissions of nitrogen and phosphorus to the Baltic Sea. The Commission has set targets for the reduction of nitrogen and phosphorus loads (Boesch 2019).

The *Convention for the Protection of the Marine Environment of the North-East Atlantic, also known as the OSPAR Convention,* entered into force in 1998 (OSPAR Secretariat 2020a). It replaced the Oslo and Paris Conventions. The 15 countries in the OSPAR Commission are Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, the Netherlands, Norway, Portugal, Spain, Sweden, Switzerland and the United Kingdom. The OSPAR Eutrophication Strategy (based on the OSPAR Convention) aims to combat eutrophication in the OSPAR maritime area by achieving reduction at source of nitrogen and phosphorus inputs in areas where these inputs are likely to cause pollution (OSPAR 2020b).

The *Mediterranean Action Plan, also known as the Barcelona Convention*, was adopted by Mediterranean countries and the European Economic Community in 1976 (Boesch 2019). It was amended in 1995 as the *Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean*, which entered into force in 2004. Contracting Parties are committed to the adoption of measures against land-based pollution, protection of biological diversity, and monitoring of pollution.

The *Bucharest Convention on the Protection of the Black Sea against Pollution* was adopted in 1992 by the six countries which are Parties to the Convention (Bulgaria, Georgia, Romania, the Russian Federation, Turkey and Ukraine) (EC 2020a). The strategic plans for the Convention

include avoiding excessive nutrient loading. A large share of these nutrients are from diffuse sources.

Emissions of ammonia are regulated by the EU *National Emission Ceilings Directive* (van Grinsven *et al*. 2015). Recently introduced standards for pig and poultry production, based on best available techniques, are aimed at contributing to implementation of the National Emission Ceilings Directive, as well as the Nitrates Directive and the Water Framework Directive (EC 2017).

In 2019 the European Parliament announced the *European Green Deal* strategy (Climate Home News 2019). Plans for its implementation are now under development. The strategy "aims to transform the EU into a fair and prosperous society, with a modern, resource-efficient and competitive economy where there are no net emissions of greenhouse gases in 2050 and where economic growth is decoupled from resource use" (European Commission [EC] 2019a). The Green Deal will contribute to realizing the United Nations 2030 Agenda for Sustainable Development and the Sustainable Development Goals. It has the potential to influence nutrient management, for example through its Farm to Fork Strategy (whose purpose is to increase sustainability in the food chain), the circular economy strategy and the biodiversity strategy (European Council 2019).

The European Commission proposed the *Product Environmental Footprint (PEF)* and *Organization Environmental Footprint (OEF)* methods as common ways to measure environmental performance (EC 2015; EC 2020b). This approach was tested with more than 280 companies and organizations between 2013 and 2018 (EC 2020). *The Single Market for Green Products Initiative* contributes to meeting the Resource Efficiency Roadmap's goal of providing the right incentives for consumers to choose the most resource efficient products through appropriate price signals and clear environmental information (EC 2020b).

The EU *Common Agricultural Policy* was originally intended to increase agricultural production, ensure food security, protect farmers' quality of life and stabilize markets while maintaining reasonable prices for consumers. The scope has expanded with time to include such issues as environmental development and pollution, rural development, land management, animal welfare, and farmer training (FAO 2020i).

The *European Initiative for Sustainable Development in Agriculture* is an alliance of national organizations in six European countries (France, Luxembourg, Hungary, the Netherlands, Sweden and United Kingdom) founded in 2001 [European Initiative for Sustainable Development in Agriculture [EISA] 2011). It aims at developing and promoting sustainable farming systems and has a network of demonstration farms. It also has an Integrated Farming Framework for self-assessment and follow-up of internal developments by farmers. With regard to resource management, the guidelines in the EISA framework include information on the storage and use of inorganic and organic fertilizers.

The *European Innovation Partnership for Agricultural Productivity and Sustainability (EIP-AGRI)* is a policy instrument that promotes more stakeholder and demand-driven research and innovation in agriculture (Moeskops 2014). It was launched in 2012 to contribute to the EU's Europe 2020 strategy for smart, sustainable and inclusive growth (EIP-AGRI 2015). EIP-AGRI includes several elements that are supportive of organic farming and agroecological innovation (Moeskops 2014).

The *European Food Sustainable Consumption and Production Round Table* was set up in 2009 by the European Commission and business associations representing the food and beverage supply chains. Addressing the issue of uncoordinated application of methodologies and guidelines for assessing and labelling the environmental performance of food and drink products has been a major objective (EC 2014). A key output is the ENVIFOOD Protocol, a food and drink specific guidance document published in 2013 (Saouter *et al*. 2014). The Protocol is intended for use as complementary guidance to the Product Environmental Footprint (PEF) and Organizational Environmental Footprint (OEF) guides.

The *Roadmap to a Resource Efficient Europe* provides a framework explaining how policies interrelate and build on each other, in which future actions can be designed and implemented

coherently. It also outlines the inter-linkages between key sectors and resources and their associated EU policy initiatives. This Roadmap establishes the milestones for resource efficient and sustainable growth (EC 2011).

North America

The *Boundary Waters Treaty*, signed by Canada and the United States in 1909, is an agreement between the two countries that neither will pollute boundary waters, or waters that flow across the boundary, to an extent that would cause injury to health or property in the other country (International Joint Commission 2020). It provides general principles for preventing and resolving disputes over shared waters. Canada and the United States created the International Joint Commission, whose main responsibilities are to approve projects that affect water levels and flows across the boundary and to investigate transboundary issues and recommending solutions.

The *Great Lakes Water Quality Agreement* (amended in 1978, 1987 and 2012) is a commitment between the two countries that provides a framework for identifying binational priorities and implementing actions that improve water quality in the Great Lakes (Government of Canada 2013; United States Environmental Protection Agency [US EPA] 2020). The agreement includes nutrients, for example phosphorus loadings, among threats to the quality of waters of the Great Lakes.

The key regional policy targeting eutrophication is the *Gulf of Mexico Hypoxia Action Plan*, which aims to reduce nitrogen and phosphorus loads (van Grinsven *et al*. 2015). The Action Plan sets out a national strategy in the United States to reduce, mitigate, and control hypoxia in the Northern Gulf of Mexico and improve water quality in the Mississippi River Basin. The activities of the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force include coordinating and supporting nutrient management activities from all sources, restoring habitats to trap and assimilate nutrients (US EPA 2019). According to a study by van Meter *et al*. (2018), an important cause of failure to reduce nutrient loading to surface waters is the existence of nutrient legacies within intensively managed watersheds. The authors of this study concluded even if nitrogen use became 100 per cent effective, it would still take decades to meet the Action Plan's target nitrogen loads because of nutrient legacies..

The *North American Free Trade Agreement (NAFTA)* is an agreement between Canada, the United States and Mexico. Fertilizers and other selected commodities traded among the three countries are subject to zero duty (The Fertilizer Institute [TFI] 2018).

Latin America

Three of the largest trading blocks in Latin America are the Pacific Alliance, the Common Market of the South (the Mercado Común del Sur, or MERCOSUR) and the Andean Community (CAN).

The Pacific Alliance was formed in 2011 and is made up of Chile, Colombia, Mexico and Peru. Its mandate includes establishing alliances with the public and private sectors for the implementation of its Action Plan, providing support for emission reduction, and economic growth that takes into account environmental degradation (Pacific Alliance 2019). Among the objectives of the Alliance are promoting a common agenda on shared environmental priorities and contributing to the implementation of the SDGs within the framework of the 2030 Agenda (Pacific Alliance 2019).

The member countries of *MERCOSUR* are Argentina, Brazil, Paraguay, Uruguay and Venezuela. Bolivia is in the process of becoming a member. MERCOSUR has also signed trade agreements with other countries and with groups of countries. The Mercosur trading block was formed in 1991 to address free movement of goods, services, and factors of production between countries. The needs addressed in the environmental policy of MERCOSUR include sustainable management of natural resources; control of potentially adverse activities; adoption of minimization and treatment practices; cleaner technologies and recycling; and monitoring of shared ecosystems (Virasoro 1996).

The Andean Community (CAN) is a trading bloc of four countries: Bolivia, Colombia, Ecuador and Peru. Chile, Argentina, Brazil, Paraguay and Uruguay are associate members, while Panama, Mexico and Spain are Observers. In 2012 CAN approved the Andean Environmental Agenda for 2012-2016, which included actions on climate change, biodiversity and water resources (Ripley 2019).

Asia

The *South Asia Co-operative Environment Programme (SACEP)* is an intergovernmental organization established in 1982 by the governments of countries in South Asia to promote and support the protection, management and enhancement of the environment in the region (South Asia Co-operative Environment Programme [SACEP] 2019). The members are Afghanistan, Bangladesh, Bhutan, India, Maldives, Nepal, Pakistan and Sri Lanka. The target areas for projects carried out by the organization include environmental education, environmental legislation, biodiversity, air pollution, and protection and management of the coastal environment.

Intergovernmental efforts to combat transboundary air pollution are encouraged by the Malé Declaration on control and prevention of air pollution and its likely transboundary effects for South Asia (SACEP 2019). A desk study commissioned by SACEP on nutrient loading to the South Asian seas formed the basis for its active involvement in regional intergovernmental cooperation on nutrient losses from fertilizers and other sources contributing to pockets of coastal eutrophication (Bay of Bengal Large Marine Ecosystem 2014). This led to the involvement of SACEP as the regional partner of the INI and INMS project to facilitate the South Asian Nitrogen Assessment. The UNEA-4 sustainable nitrogen resolution was originally drafted and adopted during the INI-INMS South Asian Nitrogen Assessment meeting organized by SACEP in Maldives in September 2017. SACEP later mobilized the support of all governments in the region and requested them to table the resolution at the United Nations, an action which was eventually spearheaded by the Government of India.

The *South Asian Nitrogen Hub* was established in 2019 under the UK Global Challenge Research Fund (GCRF) to address the challenge that nitrogen pollution presents with regard to the environment, food security, human health and the economy in South Asia (UK Research and Innovation [UKRI] 2019). The Hub is a partnership of 32 research organizations, with project partners from the UK and South Asia; the eight SAREC countries are members. This partnership is led by the Centre for Ecology and Hydrology in the United Kingdom. Areas of research covered include improving nitrogen management in agriculture; improving the use of manure, urine and natural nitrogen fixation processes; and reducing pollution by recycling of nutrients.

The South Asian Nitrogen Hub was developed with input from the INI chair, the INI South Asian Nitrogen Centre, the Indian Nitrogen Group, the Sustainable India Trust and SACEP, and was refined at the United Nations Environment Assembly (UNEA-3) (UKRI 2019). The SACEP/INMS/UN Environment link will enable the Hub to reach global partners.

The Association of Southeast Asian Nations (ASEAN) was established in 1967 to promote economic cooperation and the welfare of people in the region. Issues targeted by ASEAN include food insecurity and pollution. For example, its Regional Soil and Nutrient Management Expert Group released Guidelines on Soil and Nutrient Management (SNM) in 2017 (Assocation of Southeast Asian Nations [ASEAN] 2017). In addition, the ASEAN Centre for Biodiversity (ACB) facilitates cooperation and coordination among the 10 ASEAN Member States, and with regional and international organizations, on the conservation and sustainable use of biological diversity. It is considered possible that reducing pollution from excess nutrients will contribute to improved biodiversity (ASEAN 2020).

The *Indian Nitrogen Group (ING)* is a network of nitrogen researchers and experts (UNEP n.d.). It has begun to identify research areas, and will work in collaboration with other bodies to harmonize national regional and international concerns and priorities with regard to reactive nitrogen.

8.5 National policies and legislation on fertilizers

This section presents information from global reviews and studies that are specific to fertilizers, followed by a regional fertilizer study on Africa. It also presents relevant information extracted from global studies on water regulations, air regulations and carbon tax schemes. Finally, it provides information on selected countries (e.g., those that use fertilizers intensively, or have environmental regulations with regard to fertilizers) in order to learn from their experiences.

In general, fertilizer legislation aims to ensure that farmers receive good quality fertilizers and that environmental damage is minimized. Fertilizers can be regulated through specific fertilizer legislation or other legislation (Box 8.5-1). Fiscal policies target the availability of fertilizers to farmers and can therefore encourage or discourage their use.

Box 8.5-1 Regulating fertilizers

The objectives of fertilizer regulations include provision of food and feed and the protection of air, soil, water bodies and consumers. Fertilizers are subject to various types of legislation and regulations related to production, trade, distribution, marketing, safety and use, which can vary among, or within, countries (FAO 2019a).

Countries differ (e.g. in their economic situations and agroecological conditions) and have different priorities concerning fertilizer use and risks. Consequently, they may use different combinations of instruments to achieve their goals. Regulation of the manufacturing, trade and use of fertilizers may be related to different sectors and policies. Depending on the interests to be protected, national fertilizer legislation may cover all fertilizers or just formulated products. It may regulate only the production and/or use of fertilizers, or provide rules applicable to all stages of their life cycle. Some countries develop legislation that establishes a registration mechanism for formulated fertilizers that mirrors the classical regulatory framework for pesticides, while others put more emphasis on risks arising from certain substances used in fertilizers such as ammonium nitrate.

Fertilizers can be directly regulated by specific fertilizer legislation (addressing, for example, the registration/authorization of fertilizers on the market and their quality, labelling and packaging) or can be addressed in general agrochemicals or agricultural inputs legislation. There are also other laws that may influence the regulatory framework for fertilizers and introduce specific rules or restrictions. They include, among others:

- *health legislation* or *occupational safety legislation* that addresses, for example, risks arising from the application and use of fertilizers;
- *food safety and quality legislation* including, for example, standards for potentially toxic trace elements, and *biofortification legislation* that includes, for example, guidance on agronomic biofortification;
- *food security legislation* that provides an enabling environment for fertilizer use;
- *environmental protection legislation* that restricts or regulates the use of certain substances including fertilizers (this legislation may cover, for example, air, soil, water or biodiversity);
- *water legislation*, for example on safe drinking water and the protection of surface and ground water
- *climate change related legislation*, for example on reducing agricultural emissions, and *legislation to combat desertification*;
- *soil legislation* that includes, for example, guidance on soil fertility management;
- *waste management legislation* that addresses the disposal of fertilizer waste, and *sewage legislation* covering treatment and disposal;
- *fisheries and aquaculture legislation* for the protection of aquatic species;
- *livestock management legislation* addressing animal production practices and manure management.
- *explosives legislation* that includes, for example, materials not intended for use as explosives but which have the potential to explode.

Whether dedicated fertilizer legislation is required in a country will largely depend on national policy priorities, as well as the comprehensiveness and effectiveness of existing legal instruments.

8.5.1 Legislation on inorganic fertilizers

Global overview

A recent source of data on national legislation on fertilizers is the 2017 report on *Enabling the Business of Agriculture* (EBA) project by the World Bank Group and its partners. The main focus of the EBA project is on measuring and monitoring regulations that affect the functioning of agriculture and agribusinesses (World Bank 2017).

The report covers 62 countries representing all regions (mainly Africa, Asia, Europe and Latin America) and income groups. The main topics covered are seed, fertilizer, machinery, finance, markets, transport, information and communication technology, and water. The topics covered under fertilizer are fertilizer registration, quality and control, and importing.

This study considered factors associated with fertilizer registration, importation and distribution, and quality control. In addition, data on time and cost to register fertilizer products were collected. Factors associated with fertilizer registration, importation and distribution, and quality control were scored. The total scores, or indices, for each of these three groups of factors ranged from 0 (the worst-case scenario) to 7 (the frontier, i.e., the best performance).

In general, the worst performing countries were in sub Saharan Africa. These countries had rudimentary regulatory frameworks for registering fertilizer. They also had the lowest scores for the quality control indicator, driven by absence of laws prohibiting mislabelled and open-bag fertilizers, lack of appropriate penalties, and absence of labelling requirements in at least one of the country's official languages. EU countries performed well across all fertilizer indicators, principally due to strong rules adopted and harmonized at the EU and European level. For example, Bosnia and Herzegovina, the Netherlands and Spain all performed well on fertilizer quality and importing and distribution indices (World Bank 2017). The 2019 report on *Enabling the Business of Agriculture* (EBA) is in agreement with the 2017 report. Low-income countries showed the weakest performance on registering fertilizer, followed by lower-middle income and higher-middle income countries, while high income countries performed best (World Bank 2019).

Reviews of regulations associated with occupational hazards, for example during manufacture, transport and storage, seem to be lacking. However, safety management practices in fertilizer factories have increasingly become more stringent to curb industrial accidents (International Fertilizer Association [IFA] 2014) and some countries have regulations restricting the use of potentially explosive fertilizers. According to a report by the National Academies of Sciences, Engineering and Medicine (2018), ammonium nitrate is listed among precursor chemicals in some countries (e.g., Australia, Canada, Singapore and EU Member States). The report further adds that some countries, including the United Kingdom (Northern Ireland) and Turkey, have banned ammonium nitrate from their markets, while the United Kingdom has banned possession of ammonium nitrate fertilizers that contain more than 27.5 per cent nitrogen by mass.

Regional studies

The present report provides information from a regional study carried out in Africa. Information on studies carried out in other regions is lacking or was not available.

On the recommendation of the African Fertilizer Financing Mechanism (AFFM) Governing Council, the United Nations Economic Commission for Europe (UNECA) took the lead in the exploration of existing and potential production, cross-border trade, trade flows, and consumption for inorganic fertilizer in Africa (UNECA and AfDB 2018). The study, carried out around 2016, found that in most African countries fertilizer policies encouraged private sector involvement in fertilizer manufacture and blending to ease access to and reduce the cost of fertilizers; encourage or call for the harmonization of policies with those at the regional levels; and encourage periodic soil analyses and fertilizer trials in order to update fertilizer recommendations

(UNECA and AfDB 2018). About one-third of African countries have formal fertilizer policy and regulatory frameworks to guide the fertilizer sector, but the rest do not have coherent fertilizer policies (Wanzala-Mlobela *et al*. 2019).

Although farmers use little fertilizer in most countries in Africa, there is a need to reduce or minimize the potential risks that fertilizers pose to the environment and human health. Addressing these risks through policies, however, faces many challenges. Many formal policies and regulations are outdated and are often not specific to fertilizers, but cover other agrochemicals as well (Wanzala-Mlobela *et al*. 2019). Enforcement is another challenge due to poor inspection capacity, inadequate laboratory equipment for testing, and the logistics involved (UNECA and AfDB 2018). Conducting soil testing and fertilizer trials can also be expensive. Registration at the regional level can reduce fertilizer costs (UNECA and AfDB 2018).

Fertilizer factories and blending facilities are subject to environmental impact assessment (UNECA and AfDB 2018).

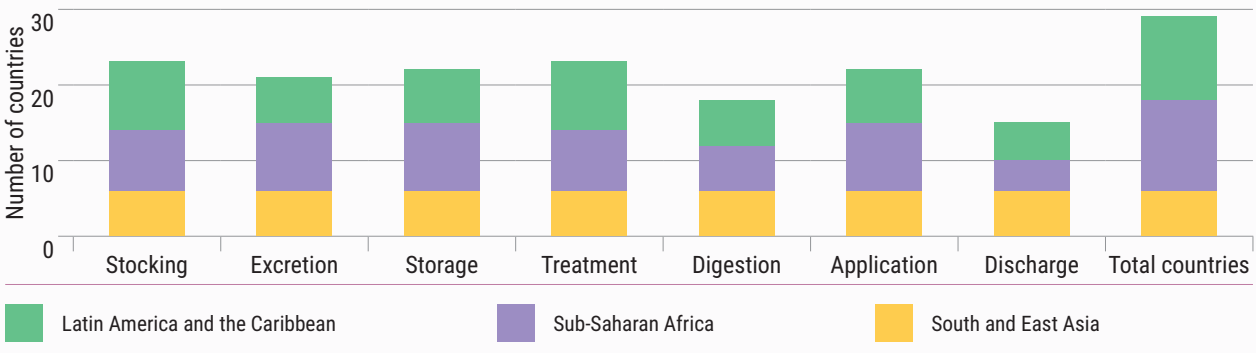
8.5.2 Legislation on organic fertilizers

Global or regional reviews of inorganic fertilizer policies appear to be lacking, despite the existence of regulations on processed organic fertilizers in some countries and regulations concerning farmers' management of organic fertilizers in many countries.

One report on the organic fertilizer industry was accessed: a study on the compliance of different organic phosphorus sources with the EU Sewage Sludge Directive by plants in Finland, Germany and Sweden (Sarvi, Ylivainio and Turtola 2017). According to that report, allowable limits for heavy metal concentrations differed among countries and among some products.

Regarding use of organic inputs, some regional studies have been carried out on manure. A study by Ndambi *et al*. (2019) on manure management practices and policies in 13 countries in sub-Saharan Africa concluded that policies do not always explicitly mention livestock manure management, but that manure is often considered a component of waste management. It concluded that animal waste is often regarded as a source of pollution and a potential human health risk rather than as a resource to be utilized and that, even when policies encouraging its use as a resource exist, enforcement can be a challenge since countries take limited action to promote good practices or to enforce legislation on manure management. In another study carried out in Latin America, SSA, and South and East Asia (Teenstra *et al*. 2014), 30 out of 34 countries surveyed had national policies related to manure management. However, enforcement of regulations was found to vary across the countries surveyed, with several policies targeting specific rather than all elements of manure management (Figure 8.5-1). In cases where different ministries were involved in

Figure 8.5-1 Number of countries addressing components of manure management in their policies reported in the study carried out by Teenstra *et al*. (2014).



Note: Data are for 29 countries (11 in Latin America, 12 in sub-Saharan Africa and six in South and East Asia).

designing manure policies, such policies were sometimes contradictory.

Christodoulou and Stamatelatou (2016) reviewed existing legislative frameworks and policies for sewage sludge management in several developed countries (Australia, Japan, New Zealand, the United Kingdom, the United States and the EU-27). All of these countries had laws on the use and disposal of sewage sludge, including its use as a fertilizer, but these laws had different restrictions. The most common parameter inspected was heavy metals. Some countries had regulations on phosphorus recovery from sewage sludge and other selected sources of phosphorus.

8.5.3 Legislation to protect air and water quality

In some countries fertilizer legislation has provisions that aim to minimize environmental damage (e.g., to biodiversity, air and water) (Food and Agriculture Organization of the United Nations and United Nations Environment Programme 2020). Data on the number of countries with such legislation in place are lacking. However, many countries have legislation to protect air and water quality, and some of this legislation includes provisions targeting pollution from fertilizers. The present report therefore considers legislation on the protection of air and water quality as part of wider "fertilizer legislation".

A 2021 report by UNEP found that in 2020 out of 195 countries, 64 (about one third) did not have any ambient air quality standards embedded within a legal instrument (UNEP 2021). Joss *et al*. (2017) reported that ambient air quality standards for PM2.5 and PM10 complied poorly with World Health Organization (WHO) guideline values. One of the study's conclusions was that these regulatory discrepancies amplify the differences in air quality and related health effects globally. Since particulate matter is associated with energy production and economic development, making progress on reducing its effects will require integrated strategies involving different disciplines (Davidson, Phalen and Solomon 2005). Lack of harmonization of standards across countries is a strong indication that national interventions, especially in regard to fertilizers, are still a long way off.

In the study by Joss *et al*. (2017), 117 out of 194 countries (60 per cent) had standards for at least one pollutant. The study also reported that for PM2.5 39 member states had short-term and 62 had long-term air quality standards, but compliance of these guidelines' with WHO guidelines was poor. Implementation of guidelines in countries is lagging. Adoption of WHO guidelines varies by pollutant, with WHO guidelines for NO2 being more likely to be adopted in law than that guidelines for ozone. Around 9 per cent of countries have ambient air quality standards for PM2.5 that meet WHO guidelines and 53 per cent of countries have standards for $PM_{2.5}$ that are less stringent that WHO AQGs according to UNEP (2021).

Information on the use of such measures to address air pollution from fertilizer production and fertilizer use is lacking. Nevertheless, it is likely that countries that manufacture fertilizers either have at least considered having standards for the fertilizer industry or apply such standards. For example, in the Republic of Indonesia a decree that provides for a standard for the quality of emissions from the fertilizer industry (Republic of Indonesia 2004). In the United States, the fertilizer industry is one source of some of the pollutants listed in the federal Clean Air Act (Hodge 1994). For example, phosphoric acid manufacturing and phosphate fertilizer production standards were promulgated in 1999 and at least 13 facilities are subject to at least one of the rules (US EPA n.d.).

Many countries have legislation on drinking water quality. WHO collected data on national and regional water quality regulations in 104 countries and territories and compared them with the WHO guidelines (World Health Organization [WHO] 2018). Among the contaminants studied were cadmium, nitrates, nitrites, Enterococci, *Escherichia coli*, Cryptosporidium, Giardia, Salmonella and helminths, which can be found in fertilizers and can affect human health (see Chapter 9 on health effects). Most of the countries studied (≥96 out of 104) had guidelines on cadmium, nitrates, nitrites and E. coli and ≤5 countries had standards for Cryptosporidium,

Giardia, Salmonella and helminths. National cadmium standards were above the WHO guidelines in slightly more than 50 per cent of countries, and standards for nitrates and nitrites were below the WHO guidelines in about for 30 per cent and 50 per cent of countries, respectively (WHO 2018).

A lack of guidelines in some countries, discrepancies between national and WHO guidelines, and poor compliance with national guidelines demonstrate that much still needs to be done with regard to legislation to protect air and water quality.

8.5.4 Fiscal policy instruments

Subsidies

In general, developed countries (unlike developing ones) do not provide significant direct fertilizer subsidies to farmers (Mamun, Martin and Tokgoz 2019). Many developing countries subsidize fertilizers to increase food production and improve farmers' incomes. Countries with fertilizer subsidies include Bangladesh, India, the Republic of Indonesia and Pakistan in Asia, and Malawi, Nigeria, Rwanda, Tanzania and several other countries in Africa (Africa Fertilizer and Agribusiness Partnership [AFAP and IFDC] 2017; UNECA and AfDB 2018). Fertilizer subsidies were estimated to amount to about United States dollars (USD) 1 billion in SSA by Jayne *et al.* (2016).

In some African countries in 2016, the proportion of subsidized fertilizers ranged from <25 per cent (e.g., in South Africa) to 100 per cent (e.g., in Ethiopia) (UNECA and AfDB 2018) (Figure 8.5-2.). In non-OECD countries coupled subsidies account for about a quarter of total support provided to the agriculture sector (Mamun, Martin and Tokgoz 2019).

Excessive and imbalanced use of nutrients contributes to soil degradation and water pollution (Gulati and Banerjee 2015). Subsidies, which have frequently been associated with imbalanced use of nutrients and fertilizer market distortions, can also be a burden on governments' budgets (Huang, Gulati and Gregory 2017). They may reduce funds that might have been available for other development priorities (UNEP 2020).

Figure 8.5-2 Amounts of fertilizer used in selected countries in Africa supplied through subsidy programmes and not from subsidized programs (thousands of tonnes). AFAP and IFDC (2017); UNECA and AfDB (2018).

Note: For Ethiopia, there is no direct subsidy program but a "hidden" subsidy exists.

Removing fertilizer subsidies – both those paid to fertilizer producers and those paid to farmers – increases fertilizer costs for farmers, especially smallholders. For example, removing all fertilizer support was predicted to lead to a decline in fertilizer use in China, India, the Republic of Indonesia and the Russian Federation (Bartelings *et al*. 2016). Removing subsidies can also influence farmers' decisions about the types of fertilizers they purchase. If fertilizer support policies were removed in India and the Republic of Indonesia, domestically produced nitrogen would become less cost attractive than imported phosphorus and potassium (Bartelings et al. 2016), contributing to more balanced and efficient fertilizer use. Experience shows that in order to transition to more sustainable practices, the process of phasing out or reforming the policy process needs to be well managed (UNEP 2020).

Some countries make direct payments to farmers who use practices known to contribute to environmental protection. For example, farmers in the EU receive payments for using farming practices that are beneficial with regard to soil quality, carbon sequestration and biodiversity (European Court of Auditors 2017).

Taxes

Environmental taxes can promote environmentally friendly behaviour and generate revenues that may be used to promote further environmental protection (Office for National Statistics of the United Kingdom 2018). However, very few countries have environmentally related fertilizer taxes (OECD n.d.). Still, in countries where carbon pricing is either being implemented or under consideration, applying a tax to fertilizers (e.g., to fertilizer manufacturing and transportation) will probably increase fertilizer prices. Currently, about 29 countries have implemented some form of carbon pricing, as carbon taxes and emission trading schemes/emission trading systems, covering about 8 per cent of global carbon emissions (World Bank 2020). However, carbon pricing does not currently appear to be applied to fertilizer in any country (OECD n.d.).

Government interventions that encourage high commodity prices (e.g., price support and high import tariffs) encourage production, while interventions that depress prices (e.g., commodity price ceilings and low import tariffs) can discourage investments in crop production (Lencucha *et al*. 2020). For example, in SSA the use of inorganic fertilizer has been shown to be strongly correlated with cereal prices (Sheahan and Barrett 2017).

Total support in agriculture

According to a recent Organisation for Economic Co-operation and Development (OECD) report on 54 countries (36 OECD member countries, the five non-OECD EU Member States, and 13 emerging economies), total support for agriculture in these countries averaged USD 708 billion/year from 2017 to 2019 (OECD 2020). About 75 per cent of total support (USD 536 billion) went to farmers and the remainder to the general services sector and consumer subsidies. Six countries implicitly imposed about USD 89 billion/year in taxes on farmers by artificially depressing prices. Noting that GHG emissions are increasing, the authors suggest that policy efforts should focus on environmental measures and on making food systems more resilient. They also emphasize that many agricultural policies are not well aligned with governments' sustainable food production and environmental health objectives.

8.5.5 Links between policies and sectors

At the national level, looking at the food security agenda alongside the environmental agenda will allow for regulations that support both agendas. The FAO and UNEP publication *Legislative Approaches to Sustainable Agriculture and Natural Resources Governance* (FAO and UNEP 2020) provides examples of countries with arrangements whereby the Ministry of Agriculture collaborates with the Water Ministry and other ministries. Such collaborations offer opportunities to enhance the sustainability of agricultural systems. They also present opportunities to exploit synergies among sectors. Analyses of existing regulatory frameworks at national level can indicate the existence of gaps, contradictions and overlap among legislations (FAO and UNEP 2020).

8.6 Corporate responsibility and stewardship

8.6.1 4R Nutrient Stewardship

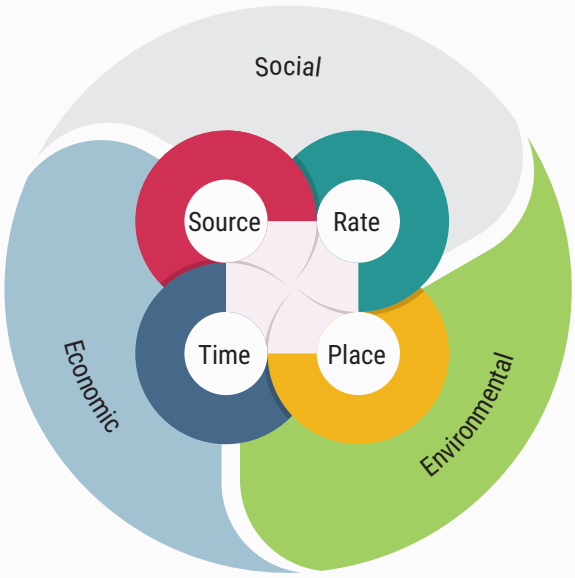
The 4R Nutrient Stewardship guidelines were developed by the fertilizer industry as a process to guide fertilizer Best Management Practices (BMPs) in all regions of the world (Johnson and Bruulsema 2014). The 4R philosophy, which is science-based, offers enhanced environmental protection, increased production, increased farmer profitability, and improved sustainability (Nutrient Stewardship 2017).

4R nutrient stewardship provides a framework for increasing crop yields sustainably and profitably while protecting the environment. This framework promotes application of the right fertilizer source (or product) (matching the fertilizer to crop needs and soil properties), at the right rate (matching the amount of fertilizer applied to crop needs), at the right time (making fertilizer nutrients available to the crop when they are needed) and in the right place (putting fertilizers where crops can use them) (Figure 8.6-1). (For the 4R approach, also see Chapter 7.) Choosing the right practices is supported by scientific principles (Johnson and Bruulsema 2014). For efficient fertilizer use, 4R nutrient stewardship recommends using these practices in combination with good agronomic practices.

Nutrient stewardship aims at establishing and increasing the adoption of the 4Rs as a strategy for economic, social, and environmental sustainability (Nutrient Stewardship 2017).

The International Fertilizer Association (IFA) product stewardship programme (IFA 2020) includes 12 Safety, Health and Environment (SHE) Principles, a certification programme (Protect and Sustain) for implementing product stewardship practices, and benchmarks that companies can use to assess their SHE performance relative to that of their peers. Recently IFA extended its sustainability portfolio to nutrient stewardship, starting with the adoption of IFA's Nutrient Stewardship Commitments in 2019. It is developing a code of practice for nutrient stewardship. In 2019 IFA also helped to establish the Scientific Panel on Responsible Plant Nutrition to meet the growing need for scientific research and expertise in order to further improve nutrient stewardship

Figure 8.6-1 The 4R Nutrient Stewardship concept. Johnson and Bruulsema (2014).



Fertilizers Europe is an association made up of 17 fertilizer manufacturers from countries across the European Union and eight national fertilizer associations. Its product stewardship programme "ensures the highest standard of fertilizer safety and environmental performance from production to application", including product development, sourcing of raw materials, additives and third party products, manufacturing, packaging, transportation, storage, marketing and sales, product application and farmer services (Fertilizers Europe 2022).

The fertilizer industry established the 4R Research Fund to help create sustainability indicators and environmental impact data for implementation of 4R nutrient stewardship across North America (Nutrient Stewardship 2017). The 4R Research Fund provides needed resource support with a focus on measuring and documenting the economic, social and environmental impacts of 4R nutrient stewardship. The Fund is within the Foundation for Agronomic Research (FAR) and is managed by The Fertilizer Institute (TFI). It includes representation from Fertilizer Canada and agricultural industry members.

8.6.2 The Sustainability Consortium

The Sustainability Consortium (TSC) is a global non-profit organization working to transform the consumer goods industry by partnering with leading companies to define, develop and deliver more sustainable products. It consists of manufacturers, retailers, suppliers, service providers, NGOs, civil society organizations, governmental agencies and academics. (The Sustainability Consortium 2020). It offers tools and services to help address product sustainability for agricultural and non-agricultural commodities. The results of the TSC 2016 Impact Report Analysis indicated that a large proportion of farmers do not collect data associated with sustainability (Slay, Reijs and Raster 2016). The fact that only 20 per cent of respondents collected data on fertilizer usage, GHG emissions and soil erosion was considered to represent a significant potential risk, but also an opportunity to improve both cost and sustainability.

References

African Fertilizer and Agribusiness Partnership (2019). AFAP website. https://www.afap-partnership.org/about-us-2/. Accessed 20 November 2020.

African Fertilizer and Agribusiness Partnership and International Fertilizer Development Center (2017). *2016 Eastern and Southern Africa Fertilizer Market Situation Statement.* https://africafertilizer.org/wp-content/uploads/2017/12/AFAP-IFDC-East-and-Southern-Africa-Fertilizer-situation-statement-2016.pdf.

African Organisation for Standardisation (2020). ARSO website. https://www.arso-oran.org/. Accessed 20 November 2020.

AirClim (2019). The convention on long-range transboundary air pollution. *Air Pollution and Climate Secretariat*. http://www.airclim.org/convention-long-range-transboundary-air-pollution-0. Accessed 20 November 2020.

Alexandratos, N. (1995). Minimizing the trade-offs between the environment and agricultural development. In *World Agriculture Towards 2010 - An FAO Study.* Rome: Food and Agriculture Organization of the United Nations and Chichester: John Wiley and Sons. Chapter 13. https://www.fao.org/3/V4200E/V4200E00.htm#Contets.

Alliance for a Green Revolution in Africa (2019). *Feeding Africa's Soils: Fertilizers to Support Africa's Agricultural Transformation*. https://agra.org/wp-content/uploads/2019/11/FeedingAfrica'sSoils.pdf.

Association of Southeast Asian Nations (2017). *ASEAN Guidelines on Soil and Nutrient Management (Final Draft).* https://www.researchgate.net/publication/319329318_ASEAN_Guidelines_on_Soil_and_Nutrient_Management/link/59a5366d45851570311b33ea/download.

Association of Southeast Asian Nations (2020). ASEAN centre for biodiversity. https://aseanbiodiversity.org/. Accessed 20 November 2020.

Backus, G. (2017). *Manure Management: An Overview and Assessment of Policy Instruments in the Netherlands.* Washington, D.C.: World Bank. https://openknowledge.worldbank.org/handle/10986/29250?locale-attribute=en.

Baltic Marine Environment Protection Commission (n.d.). HELCOM website. https://helcom.fi/about-us/. Accessed 23 November 2020.

Bartelings, H., Kavallari, A., van Meijl, H. and von Lampe, M. (2016). *Estimating the Impact of Fertilizer Support Policies: A CGE Approach*. Washington D.C.: Purdue University. https://www.gtap.agecon.purdue.edu/resources/res_display.asp?RecordID=4901.

Bay of Bengal Large Marine Ecosystem (2014). *Report of the Sub-regional Workshop to Validate the Scoping Study of Nutrient Pollution on the Coastal and Marine systems of South Asia, 21 to 22 May 2014, Colombo, Sri Lanka*. https://www.boblme.org/meetingDisplay.php?eventDisplay=2246.

Bittman, S., Dedina, M., Howard, C.M., Oenema, O. and Sutton, M.A. (2014). *Options for Ammonia Mitigation: Guidance from the UNECE Task Force on Reactive Nitrogen.* Edinburgh: Centre for Ecology and Hydrology. http://www.clrtap-tfrn.org/sites/clrtap-tfrn.org/files/documents/AGD_final_file.pdf.

Boesch, D.F. (2019). Barriers and bridges in abating coastal eutrophication. *Frontiers in Marine Science* 6 (123), 1-25. https://doi.org/10.3389/fmars.2019.00123.

Campbell, B.M., Hansen, J., Rioux, J., Stirling, C.M., Twomlow, S. and Wollenburg, E.L. (2018). Urgent action to combat climate change and its impacts (SDG 13): transforming agriculture and food systems. *Current Opinion in Environmental Sustainability* 34, 13-20. https://doi.org/10.1016/j.cosust.2018.06.005.

Carvalho, L., MacKay, E.B., Cardoso, A.C., Baattrup-Pedersen, A., Birk, S., Blackstock, K. *et al*. (2019). Protecting and restoring Europe's waters: An analysis of the future development needs of the water framework directive. *Science of The Total Environment* 658, 1228-1238. https://doi.org/10.1016/j.scitotenv.2018.12.255.

Christodoulou, A. and Stamatelatou, K. (2016). Overview of legislation on sewage sludge management in developed countries worldwide. *Water Science and Technology* 73(3), 453-462. https://doi.org/10.2166/wst.2015.521.

Climate Home News (2019). The EU releases its green deal. Here are the key points, 12 December. https://www.climatechangenews.com/2019/12/12/eu-releases-green-deal-key-points/. Accessed 20 November 2020.

Codex Alimentarius (2019). *General Standard for Contaminants and Toxins in Food and Feed. CXS 193-1995*. http://www.fao.org/fao-who-codexalimentarius/sh-proxy/en/?lnk=1&url=https%253A%252F%252Fworkspace.fao.org%252Fsites%252Fcodex%252FStandards%252FCXS%2B193-1995%252FCXS_193e.pdf.

Codex Alimentarius (2020). Codex Alimentarius International Food Standards. http://www.fao.org/fao-who-codexalimentarius/en/. Accessed 20 November 2020.

Convention on Long-range Transboundary Air Pollution Task Force on Reactive Nitrogen (2020). TFRN website. http://www.clrtap-tfrn.org. Accessed 20 November 2020.

Davidson, C.I., Phalen, R.F. and Solomon, P.A. (2005). Airborne particulate matter and human health: a review. *Aerosol Science and Technology* 39, 737-749. https://doi.org/10.1080/02786820500191348.

EIP-AGRI (2015). *EIP-AGRI Network*. https://ec.europa.eu/eip/agriculture/sites/agri-eip/files/eip-agri_brochure_network_2015_en_web.pdf.

EPD (2013). Arable crops: PCR information. https://www.environdec.com/PCR/Detail/?Pcr=8804. Accessed 20 November 2020.

EPD (2020). The international EPD system. https://www.environdec.com/home. Accessed 20 November 2020.

European Commission (2010). *The EU Nitrates Directive*. https://ec.europa.eu/environment/pubs/pdf/factsheets/nitrates.pdf.

European Commission (2011). *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions - Road Map to a Resource Efficient Europe*. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A52011DC0571.

European Commission (2014). *ENVIFOOD Protocol: Launch of the Collectively-agreed Sectorial Methodology for Assessing the Environmental Performance of Food and Drink Products in Europe*. https://publications.jrc.ec.europa.eu/repository/handle/JRC69154.

European Commission (2015). *The EC Organization Environmental Footprint and the Development of Sector-specific Guidance Documents*. https://publications.jrc.ec.europa.eu/repository/handle/JRC97586.

European Commission (2016). Circular Economy: new regulation to boost the use of organic and waste-based fertilisers, 17 March. https://ec.europa.eu/commission/presscorner/detail/de/MEMO_16_826. Accessed 20 November 2020.

European Commission (2017). New EU environmental standards for large poultry and pig farms, 22 February. https://phys.org/pdf406975798.pdf. Accessed 20 November 2020.

European Commission (2018). *Report from the Commission to the Council and the European Parliament on the Implementation of Council Directive 91/676/EEC Concerning Protection of Waters against Pollution by Nitrates from Agricultural Sources based on Member State Reports for the Period 2012-2015*. https://eur-lex.europa.eu/legal-content/en/TXT/?uri=CELEX%3A52018DC0257.

European Commission (2019a). *Annex to the Communication from the Commission to the European Parliament, the European Council, the. Council, the European Economic and Social Committee and the Committee of the Regions - The European Green Deal*. https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1596443911913&uri=CELEX:52019DC0640#document2.

European Commission (2019b). *Commission Staff Working Document Accompanying the Document Report from the Commission to the Council and the European Parliament on the Implementation of Council Directive 91/676/EEC Concerning the Protection of Waters against Pollution Caused from Agricultural Sources Based on Member State Reports for the Period 2012-2015*. https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1556887308650&uri=CELEX:52018SC0246.

European Commission (2020a). The Black Sea – The Bucharest Convention on the protection of the black sea against pollution adopted in 1992, 24 January. https://ec.europa.eu/environment/marine/international-cooperation/regional-sea-conventions/bucharest/index_en.htm. Accessed 20 November 2020.

European Commission (2020b). Single market for green products initiative: the environmental footprint pilot. https://ec.europa.eu/environment/eussd/smgp/index.htm. Accessed 20 November 2020.

European Commission (n.d.). Sewage sludge. https://ec.europa.eu/environment/topics/waste-and-recycling/sewage-sludge_en. Accessed 25 November 2020.

European Council (2019). EU adopts new rules on fertilisers, 21 May. https://www.consilium.europa.eu/en/press/press-releases/2019/05/21/eu-adopts-new-rules-on-fertilisers/. Accessed 20 November 2020.

European Court of Auditors (2017). *Greening: A More Complex Income Support Scheme, Not Yet Environmentally Effective*. https://www.eca.europa.eu/en/Pages/DocItem.aspx?did=44179.

European Environment Agency (2019). *NEC Directive Reporting Status 2019*. https://www.eea.europa.eu/themes/air/national-emission-ceilings/nec-directive-reporting-status-2019.

European Initiative for Sustainable Development in Agriculture (2011). *Gaining Excellence through Increased Awareness and Everyday Improvement: Guideline for Self-assessment of European Integrated Farming Businesses on the Basis of the European Integrated Farming Network*. http://www.sustainable-agriculture.org/wp-content/uploads/2012/07/Framework-Farmers-Version-110411.pdf.

European Sustainable Phosphorus Platform (2019). About the European Sustainable Phosphorus Platform (ESPP). https://phosphorusplatform.eu/platform/about-espp. Accessed 20 November 2020.

European Union (1986). *Council Directive of 12 June 1986 on the Protection of the Environment, and in particular of the Soil, when Sewage Sludge is Used in Agriculture (86/278/EEC)*. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:31986L0278&from=EN.

European Union (2019). *Regulation of the European Parliament and of the Council Laying down Rules on the Making Available on the Market of EU Fertilising Products and Amending Regulations (EC) No 1107/2009 and Repealing Regulation (EC) No 2003/2003 (Text with EEA Relevance)*. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32019R1009.

European Union (2020). *Safe and Effective Fertilisers on the EU Market (from 2022)*. https://eur-lex.europa.eu/legal-content/EN/LSU/?uri=CELEX%3A32019R1009.

Faraz, A., Lateef, M.A., Mustafa, M.I., Akhtar, P., Yaqoob, M. and Ur-Rehman, S. (2013). Detection of adulteration, chemical composition and hygienic status of milk supplied to various canteens of educational institutes and public places in Faisalabad. *The Journal of Animal and Plant Sciences* 23(1 Suppl.), 119-124. https://www.researchgate.net/publication/298226885_DETECTION_OF_ADULTERATION_CHEMICAL_COMPOSITION_AND_HYGIENIC_STATUS_OF_MILK_SUPPLIED_TO_VARIOUS_CANTEENS_OF_EDUCATIONAL_INSTITUTES_AND_PUBLIC_PLACES_IN_FAISALABAD.

Fertiliser Association (New Zealand) (2013). *Code of Practice for Nutrient Management (With Emphasis on Fertiliser Use)*. https://www.fertiliser.org.nz/site/code-of-practice/.

Fertilizer Australia (n.d.). Fertcare – What's it all about?. https://fertilizer.org.au/Fertcare/Fertcare-Whats-It-All-about. Accessed 20 November 2020

Fertilizers Europe (2020). The product stewardship program. http://www.productstewardship.eu/the-program/the-product-stewardship-program/. Accessed 20 November 2020.

Food and Agriculture Organization of the United Nations (2015a). *Revised World Soil Charter.* http://www.fao.org/3/I4965E/i4965e.pdf.

Food and Agriculture Organization of the United Nations (2015b). *Status of the World's Soils Resources - Main Report.* http://www.fao.org/3/i5199e/I5199E.pdf.

Food and Agriculture Organization of the United Nations (2017a). *Voluntary Guidelines for Sustainable Soil Management.* http://www.fao.org/3/a-bl813e.pdf.

Food and Agriculture Organization of the United Nations (2017b). Global S*oil Organic Carbon Map*. http://www.fao.org/documents/card/en/c/fb798a4c-ff06-4468-ad18-27787d1f3456.

Food and Agriculture Organization of the United Nations (2019). *The International Code of Conduct for Sustainable Use and Management of Fertilizers*. http://www.fao.org/documents/card/en/c/ca5253en/.

Food and Agriculture Organization of the United Nations (2020a). Global soil partnership: Global soil doctors programme. http://www.fao.org/global-soil-partnership/pillars-action/2-awareness-raising/soil-doctor/en/. Accessed 20 November 2020.

Food and Agriculture Organization of the United Nations (2020b). Global soil partnership: Global soil laboratory network. https://www.fao.org/global-soil-partnership/resources/highlights/detail/en/c/1476336/. Accessed 20 November 2020.

Food and Agriculture Organization of the United Nations (2020c). Why a global soil partnership?. http://www.fao.org/global-soil-partnership/about/why-the-partnership/en/. Accessed 20 November 2020.

Food and Agriculture Organization of the United Nations (2020d). *Global Soil Partnership: Highlights of the Work of GSP Pillars*. (VIII/2020/8). http://www.fao.org/fileadmin/user_upload/GSP/eighth_PA/GSPPA_VIII_2020_8.pdf.

Food and Agriculture Organization of the United Nations (2020e). Global Soil Partnership: soil information and data. http://www.fao.org/global-soil-partnership/areas-of-work/soil-information-and-data/en/. Accessed 22 November 2020.

Food and Agriculture Organization of the United Nations (2020f). SoiLEX – soil related legal instruments and soil governance. http://www.fao.org/soils-portal/soilex/en/. Accessed 22 November 2020.

Food and Agriculture Organization of the United Nations (2020g). *Global Soil Partnership - RECSOIL: Recarbonization of Global Soils (Update of 17.06.2020).* http://www.fao.org/global-soil-partnership/resources/highlights/detail/en/c/1201385/.

Food and Agriculture Organization of the United Nations (2020h). Global soil partnership: Regional soil partnerships. http://www.fao.org/global-soil-partnership/regional-partnerships/en/. Accessed 23 November 2020.

Food and Agriculture Organization of the United Nations (2020i). Family farming knowledge platform - 28 countries and one common agricultural policy: European family farmers and agricultural reform. http://www.fao.org/family-farming/detail/en/c/294004/. Accessed 20 November 2020.

Food and Agriculture Organization of the United Nations (2021). World soil day 2021 – halt soil salinization, boost soil productivity. http://www.fao.org/global-soil-partnership/resources/events/detail/fr/c/1372164/. Accessed 20 January 2021.

Food and Agriculture Organization of the United Nations and United Nations Environment Programme (2020). *Legislative Approaches to Sustainable Agriculture and Natural Resources Governance*. FAO Legislative Study No. 114. Rome: Food and Agriculture Organization of the United Nations. https://doi.org/10.4060/ca8728en.

Global Partnership on Nutrient Management (2014). *Global Partnership on Nutrient Management (GPNM) Progress Report – December 2014.* https://www.unenvironment.org/resources/report/global-partnership-nutrient-management-gpnm-progress-report-december-2014.

Global Partnership on Nutrient Management (n.d.). Global partnership on nutrient management (under GPA). https://sustainabledevelopment.un.org/partnership/?p=7426. Accessed 21 November 2020.

Global Phosphorus Research Initiative (2020). Phosphorus futures - an initiative of the global phosphorus research initiative. http://phosphorusfutures.net/global-research/. Accessed 20 November 2020.

Global TraPs (2015). *Global TraPs Newsletter* 13. http://www.globaltraps.ch/tl_files/pdf/newsletters/GT_Newsletter_13.pdf.

Government of Canada (2013). 2012 Great lakes water quality agreement, 25 November. https://www.canada.ca/en/environment-climate-change/services/great-lakes-protection/2012-water-quality-agreement.html. Accessed 26 October 2020.

Gulati, A. and Banerjee, P. (2015). Rationalising fertiliser subsidy in India: Key issues and policy options. *Indian Council for Research on International Economic Relations (ICRIER) Working Paper* 307, 1-48. https://www.econstor.eu/handle/10419/176138.

Hodge, C.A. (1994). Regulation and pollution control in the fertilizer industry. In *Pollution Control in Fertilizer Production.* Hodge, C.A. and Popovici, N.N. (eds.). Boca Raton, FL.: CRC Press. 23-34. https://www.taylorfrancis.com/books/9780429076787.

Hopewell, K. (2019). The WTO just ruled against China's agricultural subsidies. Will this translate to a big U.S. win?, 4 March. *Washington Post.* https://www.washingtonpost.com/politics/2019/03/04/wto-just-ruled-against-chinas-agricultural-subsidies-will-this-translate-big-us-win/. Accessed 20 November 2020.

Huang, J., Gulati, A. and Gregory, I. (eds.) (2017). *Fertilizer Subsidies - Which Way Forward? An IFDC/FAI Report*. Washington, DC: International Fertilizer Development Center and Delhi: Fertiliser Association of India. https://pdfslide.net/documents/fertilizer-subsidies-which-way-forward-subsidies-which-way-forward.html.

Intergovernmental Panel on Biodiversity and Ecosystem Services (2019). *Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. Brondizio, E.S., Settele, J., Díaz, S. and Ngo, H.T. (eds.). https://ipbes.net/global-assessment.

Intergovernmental Panel on Climate Change (2019). *Climate Change and Land. An IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse Gas Fluxes in Terrestrial Ecosystems - Summary for Policymakers - Approved Draft.* https://www.ipcc.ch/site/assets/uploads/2019/08/4.-SPM_Approved_Microsite_FINAL.pdf.

Intergovernmental Panel on Climate Change (2020). IPCC website. https://www.ipcc.ch/about/. Accessed 21 November 2020.

International Fertilizer Association (2014). *Safety Handbook: Establishing and Maintaining Positive Safety Management Practices in the Work Place. 3rd edition*. https://www.fertilizer.org/images/Library_Downloads/2014_ifa_safetyhandbook_3rdedition.pdf.

International Fertilizer Association (2020). What is product stewardship?. https://www.fertilizer.org/Public/Stewardship/Product_Stewardship/Public/Stewardship/Product_Stewardship/What_is_Product_Stewardship.aspx?hkey=1ce1d41f-236e-42cd-916a-6a1b2f8429c5. Accessed 20 November 2020.

International Joint Commission (2020). IJC website. https://ijc.org/en. Accessed 20 November 2020.

International Nitrogen Initiative (2014). *Kampala Statement-for-Action on Reactive Nitrogen in Africa and Globally.* http://nitrogen.ipni.net/ipniweb/region/nitrogen.nsf/0/D51BE25E41D286D185257E7E00567292/$FILE/2013%20Kampala%20Statement.pdf.

International Nitrogen Initiative (2020). INI webpage. https://initrogen.org/content/about-ini. Accessed 20 November 2020.

International Nitrogen Management System (2019). *Fourth Meeting of the International Nitrogen Management System (INMS-4). Summary of the High-Level Segment.* https://www.inms.international/sites/inms.international/files/INMS-4%20Summary%20Report%20of%20high-level%20segment%20(30%20July%202019)%20(002).pdf.

International Nitrogen Management System (2020a). Project hosts. https://www.inms.international/project-hosts/project-hosts. Accessed 20 November 2020.

International Nitrogen Management System (2020b). INMS website. https://www.inms.international/about_inms#targetText=An%20%E2%80%99International%20Nitrogen%20Management%20System,to%20improve%20global%20nitrogen%20management. Accessed 20 November 2020.

Jayne, T.S., Mason, N.M., Burke, W.J., and Ariga, J. (2016). Agricultural Input Subsidy Programs In Africa: An Assessment of Recent Evidence. *Feed the Future Innovation Lab for Food Security Policy Research Papers 29*. Michigan State University. https://www.canr.msu.edu/resources/agricultural-input-subsidy-programs-in-africa-an-assessment-of-recent-evidence.

Jin, S. and Zhou, F. (2018). Zero growth of chemical fertilizer and pesticide use: China's objectives, progress and challenges. *Journal of Resources and Ecology* 9(1), 50-58. https://doi.org/10.5814/j.issn.1674-764x.2018.01.006.

Johnston, A.M. and Bruulsema, T.W. (2014). 4R nutrient stewardship for improved nutrient use efficiency. *Procedia Engineering* 83, 365-370. https://doi.org/10.1016/j.proeng.2014.09.029.

Joss, M.K., Eeftens, M., Gintowt, E., Kappeler, R. and Künzli, N. (2017). Time to harmonize national ambient air quality standards. *International Journal of Public Health* 62, 453-462. https://doi.org/10.1007/s00038-017-0952-y.

Leadley, P.W., Krug, C.B., Alkemade, R., Pereira, H.M., Sumaila U.R., Walpole, M. *et al.* (2014). *Progress towards the Aichi Biodiversity Targets: An Assessment of Biodiversity Trends, Policy Scenarios and Key Actions.* Technical Series No. 78. Montreal: Secretariat of the Convention on Biological Diversity. https://www.researchgate.net/publication/269845681_Progress_towards_the_Aichi_Biodiversity_Targets_An_Assessment_of_Biodiversity_Trends_Policy_Scenarios_and_Key_Actions.

Lencucha, R., Pal, N.E., Appau, A., Thow, A.-M. and Drope, J. (2020). Government policy and agricultural production: A scoping review to inform research and policy on healthy agricultural commodities. *Globalization and Health* 16, 11. https://doi.org/10.1186/s12992-020-0542-2.

Mamun, A., Martin, W. and Tokgoz, S. (2019). *Reforming Agricultural Subsidies for Improved Environmental Outcomes*. Washington, DC: International Food Policy Research Institute. http://ebrary.ifpri.org/utils/getfile/collection/p15738coll2/id/133527/filename/133739.pdf.

Moeskops, B (ed.). (2014). *The European Innovation Partnership: Opportunities for Innovation in Organic Farming and Agroecology.* Brussels: European Innovation Partnership for Agricultural Productivity and Sustainability (EIP-AGRI). https://orgprints.org/29868/1/ifoameu_research_eip_dossier_en_201402.pdf.

National Academies of Sciences, Engineering and Medicine (2018). *Reducing the Threat of Improvised Explosive Device Attacks by Restricting Access to Explosive Precursor Chemicals.* Washington, D.C.: National Academies Press. https://www.nap.edu/catalog/24862/reducing-the-threat-of-improvised-explosive-device-attacks-by-restricting-access-to-explosive-precursor-chemicals.

Ndambi, O.A., Pelster, D.E., Owino, J.O., de Buisonjé, F. and Vellinga, T. (2019). Manure management practices and policies in Sub-Saharan Africa: Implications on manure quality as a fertilizer. *Frontiers in Sustainable Food Systems 3*, 29. https://doi.org/10.3389/fsufs.2019.00029.

New Partnership for Africa's Development (2011). *The Abuja Declaration on Fertilizers for an African Green Revolution - Status of Implementation at Regional and National Levels.* https://africafertilizer.org/wp-content/uploads/2017/04/Seventh-Progress-Report-on-the-Abuja-Declaration-on-Fertilizers-for-an-African-Green-Revolution.pdf.

Nutrient Stewardship (2017). 4R research fund. http://www.nutrientstewardship.org/4r-research-fund/. Accessed 21 November 2020.

Office for National Statistics of the United Kingdom (2018). Five facts about environmental taxes, 9 October. https://www.ons.gov.uk/economy/environmentalaccounts/articles/fivefactsaboutenvironmentaltaxes/2018-10-09. Accessed 21 November 2020.

Oil Spill Prevention Administration Response Secretariat (2020a). How OSPAR works. https://www.ospar.org/about/how. Accessed 21 November 2020.

Oil Spill Prevention Administration Response Secretariat (2020b). Eutrophication. https://www.ospar.org/work-areas/hasec/eutrophication#targetText=Eutrophication,the%20bottom%20of%20the%20sea. Accessed 21 November 2020.

Organisation for Economic Co-operation and Development (2020). *Measuring Distortions in International Markets: The Agriculture Sector.* Agriculture Policy Brief https://issuu.com/oecd.publishing/docs/measuring_distortions_in_internatio.

Organisation for Economic Co-operation and Development (n.d.). Database on policy instruments for the environment: Taxbases - tax rates of environmentally related taxes. https://pinedatabase.oecd.org/Query_2.aspx?QryCtx=3&isid=16a57da7-500f-4b65-b89e-69151386f6dc. Accessed 21 November 2020.

Pacific Alliance (2019). Environment. https://alianzapacifico.net/en/technical-group-environment/. Accessed 21 November 2020.

Ramsar Convention on Wetlands (2018). *Global Wetland Outlook: State of the World's Wetlands and their Services to People.* https://www.researchgate.net/publication/328093181_Global_Wetland_Outlook_State_of_the_World's_Wetlands_and_Their_Services_to_People/link/5bb70849299bf1049b6f83ff/download.

Ramsar Convention on Wetlands (2020). Ramsar website. https://www.ramsar.org. Accessed 20 November 2020.

Republic of Indonesia (2004). *Decree of the State Minister for Environment No. 133/2004 on the Standard Quality of Emission for Activities of the Fertilizer Industry*. https://www.informea.org/en/legislation/decree-state-minister-environment-no-1332004-standard-quality-emission-activities.

Ripley, K. (2019). Ministers adopt Andean environment agenda, water resources management action plan, 17 April. http://sdg.iisd.org/news/ministers-adopt-andean-environment-agenda-water-resources-management-action-plan/. Accessed 21 November 2020.

Roy, A. (2019). Foreword. In *Feeding Africa's Soils: Fertilizers to Support Africa's Agricultural Transformation*. Nairobi: Alliance for a Green Revolution in Africa. VI-VII. https://agra.org/wp-content/uploads/2019/11/FeedingAfrica'sSoils.pdf.

Rusike, J. and Camara, O. (2019). Fertilizer policy. In *Feeding Africa's Soils.* Nairobi: Alliance for a Green Revolution in Africa. Chapter 8. 88-109. https://agra.org/wp-content/uploads/2019/11/FeedingAfrica'sSoils.pdf.

Sanz, M.J., de Vente, J., Chotte, J.-L., Bernoux, M., Kust, G., Ruiz, I. *et al.* (2017). *Sustainable Land Management Contribution to Successful Land-based Climate Change Adaptation and Mitigation - A Report of the Science-Policy Interface.* Bonn: United Nations Convention to Combat Desertification. https://www.unccd.int/sites/default/files/documents/2017-09/UNCCD_Report_SLM_web_v2.pdf.

Saouter, E., Bauer, C., Blomsma, C., De Camillis, C., Lopez, P., Lundquist, L. *et al.* (2014). *Moving from the ENVIFOOD Protocol to Harmonized Product Category Rules and Reference Data: Current and Future Challenges of the European Food Sustainable Consumption and Production Round Table*. San Francisco: International Conference LCA of Food. https://www.researchgate.net/publication/271827090_Moving_from_the_ENVIFOOD_Protocol_to_harmonized_Product_Category_Rules_and_reference_data_current_and_future_challenges_of_the_European_Food_Sustainable_Consumption_and_Production_Round_Table.

Sarvi, M., Ylivainio, K. and Turtola, E. (2017). *Report on Compliance of Recycled Product with Present EU Fertilizer Regulations. BONUS: Science for a Future of the Baltic Sea Region*. https://www.researchgate.net/publication/319036840_Report_on_compliance_of_recycled_product_with_present_EU_fertilizer_regulations.

Secretariat of the Convention on Biological Diversity (2000). *Cartagena Protocol on Biosafety to the Convention on Biological Diversity: Text and Annexes.* https://www.cbd.int/doc/legal/cartagena-protocol-en.pdf.

Secretariat of the Convention on Biological Diversity (2014). *Global Biodiversity Outlook 4 – A Mid-term Assessment of Progress towards the implementation of the Strategic Plan for Biodiversity 2011-2020.* https://www.cbd.int/gbo/gbo4/publication/gbo4-en-hr.pdf.

Secretariat of the Convention on Biological Diversity (2020a). The Convention on biological diversity. https://www.cbd.int/convention/. Accessed 20 November 2020.

Secretariat of the Convention on Biological Diversity (2020b). Target 8 – Technical rationale extended (Provided in Document COP/10/INF/12/Rev.1). https://www.cbd.int/sp/targets/rationale/target-8/#footnote27. Accessed 20 November 2020.

Secretariat of the Convention on Biological Diversity (2020c). *Global Biodiversity Outlook 5 – Summary for Policymakers.* https://www.cbd.int/gbo5.

Sheahan, M. and Barrett, C.B. (2017). Ten striking facts about agricultural input use in Sub-Saharan Africa. *Food Policy* 67, 12-25. https://doi.org/10.1016/j.foodpol.2016.09.010.

Slay, C.M., Reijs, J. and Raster, A. (2016). *How to Get Sustainability Data Flowing in Agriculture Supply Chains. A Preliminary Report.* Scottsdale, Arizona: The Sustainability Consortium. https://www.sustainabilityconsortium.org/tsc-downloads/how-to-get-sustainability-data-flowing-in-agriculture-supply-chains/?ind=1502197187404&filename=TSC%202016_How%20to%20Get%20Sustainability%20Data%20Flowing%20in%20Agricultural%20Supply%20Chains.pdf&wpdmdl=18033&refresh=5fba8659db5841606059609.

Smith, P., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E.A. *et al*. (2014). Agriculture, Forestry and Other Land Use (AFOLU). In *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.* Edenhofer, O., Pichs-Madruga, R., Sokona, Y., Farahani, E., Kadner, S., Seyboth, K. *et al*. (eds.). Cambridge and New York: Cambridge University Press. 811-922. https://www.ipcc.ch/site/assets/uploads/2018/02/ipcc_wg3_ar5_chapter11.pdf.

South Asia Co-operative Environment Programme (2019). SACEP website. http://www.sacep.org/about-us. Accessed 21 November 2020.

Southern African Development Community (2011). *Regional Agricultural Policy - Country Summary Agricultural Policy Review Reports.* https://reliefweb.int/sites/reliefweb.int/files/resources/Regional%20Agricultural%20Policy%20-%20Country%20Summary%20Agricultural%20Policy%20Review%20Reports.pdf.

Studer, I. (2017). Renegotiation of NAFTA: An opportunity for sustainable development. *International Institute for Sustainable Development*. https://www.iisd.org/story/after-nafta/. Accessed 21 November 2020.

Sustainable Agriculture Initiative Platform (2006). *Principles and Practices for the Sustainable Production of Cereals.* https://saiplatform.org/wp-content/uploads/2006/06/sai_platform_principles_practices_cereals.pdf.

Sustainable Agriculture Initiative Platform (2020). SAI website. https://saiplatform.org/who-we-are/. Accessed 21 November 2020.

Teenstra, E., Vellinga, T., Aektasaeng, N., Amatayakul, W., Ndambi, A., Pelster, D. *et al*. (2014). *Global Assessment of Manure Management Policies and Practices.* Wageningen, the Netherlands: Wageningen UR Livestock Research. https://edepot.wur.nl/335445.

The Fertilizer Institute (2018). TFI outlines priorities for the NAFTA renegotiation, 24 October. https://www.tfi.org/newsroom/2017/NAFTA-priorities. Accessed 21 November 2020.

The Sustainability Consortium (2020). TSB website. https://www.sustainabilityconsortium.org/. Accessed 21 November 2020.

The Sustainable Rice Platform (n.d.). SRP website. http://www.sustainablerice.org/About-Us/. Accessed 21 November 2020.

Thompson, R.L., Lassaletta, L., Patra, P.K., Wilson, C., Wells, K.C., Gressent, A. *et al*. (2019). Acceleration of global N20 emissions seen from two decades of atmospheric inversion. *Nature Climate Change* 9, 993-998. https://doi.org/10.1038/s41558-019-0613-7.

Transboundary Water Resources Management in Southeastern Europe (2012). UNECE Water Convention. http://twrm-med.net/southeastern-europe/southeastern-europe/regional-dialogue/unece-water-convention. Accessed 21 November 2020.

UK Research and Innovation (2019). GCRF South Asian nitrogen hub. https://gtr.ukri.org/projects?ref=NE%2FS009019%2F1&_ga=2.42478437.1708564352.1568626098-87944384.1568626098. Accessed 20 November 2020.

United Nations Convention to Combat Desertification (2016). About the UNCCD- WOCAT partnership. https://knowledge.unccd.int/knowledge-products-and-pillars/best-practices-sustainable-land-management/about-unccd-wocat. Accessed 20 November 2020.

United Nations Convention to Combat Desertification (2017a). Countries agree on a landmark 2030 strategy to save fertile lands, 19 September. https://www.unccd.int/news-events/countries-agree-landmark-2030-strategy-save-fertile-lands. Accessed 20 November 2020.

United Nations Convention to Combat Desertification (2017b). *Global Land Outlook. First Edition*. https://knowledge.unccd.int/sites/default/files/2018-06/GLO%20English_Full_Report_rev1.pdf.

United Nations Convention to Combat Desertification (2020). Action programmes. https://www.unccd.int/convention/action-programmes. Accessed 20 November 2020.

United Nations Economic Commission for Africa and African Development Bank (2018). *African Fertilizer Financing Mechanism: AFMM. Study (Sponsored by UNECA and AFMM) on Promotion of Fertilizer Production, Cross-Border Trade and Consumption in Africa.* https://www.afdb.org/fileadmin/uploads/afdb/Documents/Generic-Documents/Study__sponsored_by_UNECA___AFFM__on_promotion_of_fertilizer_production__cross-border_trade_and_consumption_in_Africa.pdf.

United Nations Economic Commission for Europe (1999). Protocol to abate acidification, eutrophication and ground-level ozone. https://www.unece.org/env/lrtap/multi_h1.html. Accessed 21 November 2020.

United Nations Economic Commission for Europe (2019a). Convention on long-range transboundary air pollution. https://unece.org/convention-and-its-achievements. Accessed 21 November 2020.

United Nations Economic Commission for Europe (2019b). Protocol on water and health: Targets set by parties. https://www.unece.org/env/water/pwh_targets_set.html. Accessed 21 November 2020.

United Nations Economic Commission for Europe (2020). We must not wait for the next ammonium nitrate blast - solutions exist to improve safety, 6 August. https://unece.org/environment/press/we-must-not-wait-next-ammonium-nitrate-blast-solutions-exist-improve-safety. Accessed 21 November 2020.

United Nations Economic Commission for Europe (2020a). The Water Convention and the Protocol on Water and Health. https://unece.org/environment-policy/water. Accessed 21 November 2020.

United Nations Economic Commission for Europe (2020b). Industrial accidents. https://unece.org/environment-policy/industrial-accidents. Accessed 21 November 2020.

United Nations Economic Commission for Europe (n.d.). Protocol concerning the control of emissions of nitrogen oxides. https://www.unece.org/env/lrtap/nitr_h1.html. Accessed 23 November 2020.

United Nations Economic Commission for Europe and World Health Organization (1999). *Protocol on Water and Health to the 1992 Convention on the Protection and Use of Transboundary Watercourses and International Lakes.* https://www.unece.org/fileadmin/DAM/env/documents/2000/wat/mp.wat.2000.1.e.pdf.

United Nations Environment Assembly of the United Nations Environment Programme (2019a). *Implementation Plan "Towards a Pollution-Free Planet*. UNEP/EA.4/L.25. https://papersmart.unon.org/resolution/uploads/k1804190.pdf.

United Nations Environment Assembly of the United Nations Environment Programme (2019b). *Resolution adopted by the United Nations Environment Assembly on 15 March 2019: 4/14 - Sustainable Nitrogen Management*. UNEP/EA.4/Res.14. https://leap.unep.org/sites/default/files/unea-resolutions/Implementation%2520Pla.pdf.

United Nations Environment Programme (2019). Colombo Declaration calls for tackling global nitrogen challenge, 24 October. https://www.unenvironment.org/news-and-stories/press-release/colombo-declaration-calls-tackling-global-nitrogen-challenge. Accessed 21 November 2020.

United Nations Environment Programme (2020a). *Global Programme of Action for the Protection of the Marine Environment from Land-based Activity*. https://www.unep.org/resources/toolkits-manuals-and-guides/global-programme-action-protection-marine-environment-land.

United Nations Environment Programme (2020b). *Study on the Effects of Taxes and Subsidies on Pesticides and Fertilizers. Background Document to UNEA-5 Review Report on the Environmental and Health Effects of Pesticides and Fertilizers.* https://wedocs.unep.org/handle/20.500.11822/33582.

United Nations Environment Programme (2021). *Air Pollution Series Actions on Air Quality: A Global Summary of Policies and Programmes to Reduce Air Pollution*. https://wedocs.unep.org/bitstream/handle/20.500.11822/36694/AAQ_GSP_PRAP.pdf.

United Nations Environment Programme (2022). *Synthesis Report on Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them*. https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

United Nations Environment Programme (n.d.). Indian Nitrogen Group. http://www.nutrientchallenge.org/indian-nitrogen-group. Accessed 20 November 2020.

United Nations Framework Convention on Climate Change (2020a). Status of ratification of the Convention. https://unfccc.int/process-and-meetings/the-convention/status-of-ratification/status-of-ratification-of-the-convention. Accessed 21 November 2020.

United Nations Framework Convention on Climate Change (2020b). What is the United Nations Framework Convention on climate change?. https://unfccc.int/process-and-meetings/the-convention/what-is-the-united-nations-framework-convention-on-climate-change. Accessed 21 November 2020.

United Nations Framework Convention on Climate Change (2020c). What is the Kyoto Protocol?. https://unfccc.int/kyoto_protocol. Accessed 21 November 2020.

United States Environmental Protection Agency (2019). Mississippi River/Gulf of Mexico hypoxia task force: History of the hypoxia task force. https://www.epa.gov/ms-htf/history-hypoxia-task-force. Accessed 20 November 2020.

United States Environmental Protection Agency (2020). What is GLWQA?. https://www.epa.gov/glwqa/what-glwqa. Accessed 20 November 2020.

United States Environmental Protection Agency (n.d.). Phosphate fertilizer production plants and phosphoric acid manufacturing plants: National emission standards for hazardous air pollutants (NESHAP) - 40 CFR 63 subparts AA & BB. https://www.epa.gov/stationary-sources-air-pollution/phosphate-fertilizer-production-plants-and-phosphoric-acid. Accessed 20 November 2020.

United States Geological Survey (2020). *Mineral Commodities Summaries 2020.* https://pubs.usgs.gov/periodicals/mcs2020/mcs2020.pdf.

Van Grinsven, H.J.M., Bouwman, L., Cassman, K.G., van Es, H.M., McCrackin, M.L. and Beusen, A.H.W. (2015). Losses of ammonia and nitrate from agriculture and their effect on nitrogen recovery in the European Union and the United States between 1900 and 2050. *Journal of Environmental Quality* 44(2), 356-367. https://doi.org/10.2134/jeq2014.03.0102.

Van Grinsven, H.J.M., Tiktak, A., and Rougoor, C.W. (2016). Evaluation of the Dutch implementation of the Nitrates Directive, the water framework directive and the national emission ceilings directive. *NJAS: Wageningen Journal of Life Sciences* 78(1), 69-84. https://doi.org/10.1016/j.njas.2016.03.010.

Van Kauwenbergh, S.J., Stewart, M., and Mikkelsen, R. (2013). World reserves of phosphate rock… A dynamic and unfolding story. *Better Crops* 97(3), 18-20. http://www.ipni.net/publication/bettercrops.nsf/0/C3AB0523A890EBC685257BD50055E09A/$FILE/BC3%202013%20-%20p18.pdf.

Van Meter, K.J., Van Cappellen, P. and Basu, N.B. (2018). Legacy nitrogen may prevent achievement of water quality goals in the Gulf of Mexico. *Science* 360(6387), 427-430. http://doi.org/10.1126/science.aar4462.

Virasoro, C.L. (1996). Environmental aspects of regional integration in MERCOSUR. *Review of European, Comparative and International Environmental Law (RECIEL*) 5(1), 38-45. https://doi.org/10.1111/j.1467-9388.1996.tb00308.x.

Wanzala-Mlobela, M., Kalihangabo, M.C., Ba, M.N., Asseta, D. and Banda, K. (2019). The Abuja Declaration on fertilizers. In *Feeding Africa's Soils: Fertilizers to Support Africa's Agricultural Transformation.* Nairobi: Alliance for a Green Revolution in Africa. Chapter 3. 22-38. https://agra.org/wp-content/uploads/2019/11/FeedingAfrica'sSoils.pdf.

World Bank (2017). *Enabling the Business of Agriculture 2017.* https://www.google.co.uk/books/edition/Enabling_the_Business_of_Agriculture_201/vu8kDwAAQBAJ?hl=en&gbpv=1&printsec=frontcover.

World Bank (2019). *Enabling the Business of Agriculture 2019.* https://openknowledge.worldbank.org/bitstream/handle/10986/31804/9781464813870.pdf.

World Bank (2020). Carbon pricing dashboard - Map and data. https://carbonpricingdashboard.worldbank.org/map_data. Accessed 20 November 2020.

World Health Organization (2018). *A Global Overview of National Regulations and Standards for Drinking Water Quality.* https://www.who.int/publications/i/item/9789241513760.

World Trade Organization (n.d.). Agriculture: Fairer markets for farmers. https://www.wto.org/english/thewto_e/whatis_e/tif_e/agrm3_e.htm. Accessed 20 November 2020.

Yang, X., Dong, X., Jiang, Q. and Liu, G. (2019). Factors influencing public concern about environmental protection: An Analysis from China. *Discrete Dynamics in Nature and Society* 2019, 5983160. https://doi.org/10.1155/2019/5983160.

Yu, W. (2017). *How China's Farm Policy Reforms Could Affect Trade and Markets: A Focus on Grains and Cotton.* Geneva: International Centre for Trade and Sustainable Development. https://static-curis.ku.dk/portal/files/181223912/wusheng_yu_paper_final.pdf.

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them

Envisioning A Chemical-Safe World

Chapter 9 of 12

Environmental and health effects of fertilizer use

Contents

1 Global drivers, actors and policies affecting pesticides and fertilizer use

2 Status and trends of pesticide use

3 The regulatory and policy environment for pesticide management

4 Environmental and health effects of pesticide use

5. The environmental, human health and economic impacts of pesticides

6 Current pesticide risk reduction and risk management

7 Status and trends of fertilizer use

8 The regulatory and policy environment of fertilizer management and use

About ii

9 Environmental and health effects of fertilizer use 1

9.1 Overview 1

9.2 Adverse environmental effects of fertilizer use 3

9.2.1 Air 3
9.2.2 The terrestrial environment 7
9.2.3 The aquatic environment 11

9.3 Adverse health effects of fertilizer use 15

9.3.1 Adverse effects of occupational exposure 15
9.3.2 Adverse effects of dietary exposure 18
9.3.3 Adverse effects of residential exposure 27
9.3.4 Fertilizers and antimicrobial resistance 28

9.4 Links between the environmental and health effects of fertilizers 28

9.5 Key data gaps: environmental and health effects of fertilizers 29

References 30

10 The impact of fertilizer use

11 Current fertilizer risk reduction and risk management

12 Transformative actions to minimize the adverse impacts of pesticide and fertilizers

List of Boxes

Box 9.3-1 China: Dietary cadmium could be a threat to human health 20

Box 9.3-2 General findings of two epidemiological reviews on fertilizers ("fertilizers and human health", "cadmium case study") carried out for this report. 21

Box 9.3-3 Chronic kidney disease of unknown origin 23

List of Figures

Figure 9.2-1 Agriculture's shares of regional and global carbon dioxide equivalent (CO_2eq) emissions in 2017. 4

Figure 9.2-2 Agriculture's shares of methane (CH_4), nitrous oxide (N_2O) and carbon dioxide equivalent (CO_2eq) emissions, 1990-2017. 4

Figure 9.2-3 Contribution of different sources of agricultural emissions from crop and livestock production (per cent). 5

Figure 9.2-4 Amount of nitrogen lost as ammonia, as a percentage of nitrogen applied, for major inorganic fertilizers. 7

Figure 9.2-5 Global distribution of documented cases of hypoxia related to human activities. Each case is represented by a dot. The number of hypoxic areas is cumulative for the successive time periods. 13

Figure 9.3-1 Simplified generic schematic diagram of connections between fertilizers and human health issues. 16

List of Tables

Table 9.3-1 Respiratory effects in humans from inhalation exposure to ammonia in fertilizer factories. 17

Table 9.3-2 Evidence of high nitrate concentrations in water in selected countries. 22

About

In December 2017, Resolution 4 of the 3rd Session of the United Nations Environment Assembly (UNEA 3) requested "the Executive Director to present a report on the environmental and health impacts of pesticides and fertilizers and ways of minimizing them, given the lack of data in that regard, in collaboration with the World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO) and other relevant organizations by the fifth session of the United Nations Environment Assembly". In response to this request, UNEP published a *Synthesis Report on the Environmental and Health Impacts of Pesticides and Fertilizers and Ways to Minimize Them*[1] in February 2022 (United Nations Environment Programme [UNEP] 2022).

The overall goal of the synthesis report is to provide the information base to enable other advocacy actions to be taken by stakeholders to minimize the adverse impacts of pesticides and fertilizers. Specific objectives of the synthesis report are to:

- Update understanding of current pesticide and fertilizer use practices;
- Present major environmental and health effects of pesticides and fertilizers, during their life cycle, and identify key knowledge gaps;
- Review current management practices, legislation and policies aimed at reducing risks in the context of the global chemicals, environmental and health agenda;
- Identify opportunities to minimize environmental and health impacts, including proven and innovative approaches.

This chapter on "Environmental and health effects of fertilizer use" is the 9th in a series of 12 chapters that make up a comprehensive compilation of scientific information. The chapters were developed to both inform and further elaborate on the information provided in the synthesis report. Please note that the disclaimers and copyright from the synthesis report apply.

1 The Synthesis report is available at https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

9 Environmental and health effects of fertilizer use

9.1 Overview

Fertilizers are included among the sources of air, water and soil contamination and pollution.

There is an overall lack of adequate data on the environmental and health effects of fertilizers. Estimating these effects is hampered by, for example, the difficulty of quantifying the effects of diffuse sources of contaminants and pollutants.

Greenhouse gas (GHG) emissions comprise methane (CH_4) and nitrous oxide (N_2O) emissions from crop and livestock activities and carbon dioxide (CO_2) emissions from land use and land use change (Tubeillo *et al*. 2015). The crop and livestock sources of CH_4 are mainly enteric fermentation, rice cultivation and manure management, while those of N_2O are manure, inorganic fertilizers, and emissions from crop residues (Tubeillo *et al*. 2013). [Chapter 9.2.1]

GHG emissions occur during the manufacture of fertilizers (Zhang, W. *et al*. 2013). Fertilizer production is also reported to contaminate the environment with trace elements, for example cadmium and arsenic during the production of phosphate fertilizer (Mirlean and Roisenberg 2006). [Chapter 9.2.1] The application of nitrogen fertilizers has been associated with N_2O emissions (Albanito *et al*. 2017). These emissions are higher in the case of soils to which both inorganic and organic fertilizers have been applied (Charles *et al*. 2017). Efficient fertilizer use can help reduce such emissions (Thapa *et al*. 2016). [Chapter 9.2.1]

Ammonia is the main $PM_{2.5}$ precursor from agricultural activities, emitted from heavily fertilized fields and livestock waste, it combines with other precursors to form particulate matter (Li, X. *et al*. 2015; Bauer, Tsigaridis and Miller 2016; Wu, Y. et al. 2016). It can contribute to nutrient imbalances, and eutrophication (Organisation for Economic Co-operation and Development [OECD] 2013; Royal Society and Rand Europe 2018). Ammonia is also a key acidifying pollutant that contributes to soil acidification and acid rain (or acid deposition), which damages vegetation including crops and forests, causes further acidification of soils, lakes and rivers, and has adverse effects on biodiversity (FAO 2002; OECD 2013; Royal Society and Rand Europe 2018; United States Environmental Protection Agency [US EPA] 2020a). [Chapter 9.2.1]

Fertilizers can have a negative impact on soil's physical and chemical properties (e.g., soil acidification), as well as on soil organisms (e.g., microorganisms, invertebrates and other animals) and plants (FAO 2015; Wall, Nielsen and Six 2015; Humbert *et al*. 2016; Midolo *et al*. 2019). They may contain

pathogens and contaminants such as toxic concentrations of trace elements, and endocrine disrupters. Applying contaminated fertilizers to soil could introduce these contaminants into the environment (Gionfa 2018; Rodríguez Eugenio, McLaughlin and Pennock 2018). Nitrogen fertilization can affect soil biodiversity, contributing to the decomposition of soil organic matter and crop residues (Li, X.G. *et al*. 2017) which may lead to degradation of soil quality (FAO *et al*. 2020). [Chapter 9.2.2]

Microplastics, which are among the contaminants present in fertilizers, are reported to affect soil's biophysical properties. The sources of microplastics in soils include sewage sludge (biosolids), controlled-release fertilizers, compost, and wastewater used for irrigation (Gionfa 2018; He et al. 2018; Weithmann *et al*. 2018; Guo, J. et al. 2020). [Chapter 9.2.2]

Pollution of the aquatic environment by nitrogen and phosphorus contributes to eutrophication and affects the dynamics of the aquatic food web. This pollution has adverse effects on coral reefs, algae, aquatic invertebrates, amphibians and fish (Egertson and Downing 2004; Díaz *et al*. 2012; Rabalais *et al*. 2014; Jeppesen *et al*. 2016; Zieritz *et al*. 2016; Garriga, Montori and Llorente 2017; Khan *et al*. 2018). [Chapter 9.2.3]

Fertilizes are best known for their contribution to food security. The systematic reviews carried out for this report identified a data gap. There were no reports found directly linking responsible and efficient fertilizer use with human health effects or quantifying the contribution of fertilizers to these effects. Where the use of fertilizers was suspected to be the cause of certain illnesses, studies have been inconclusive, or data are lacking. Due to the lack of adequate data, studies that are not specific to fertilizers are considered in this chapter (e.g., those on contaminants that can potentially be from fertilizers). In some of these studies contaminants that could come from fertilizers were associated with adverse environmental and health effects. [Chapter 9.3]

Any negative health effects that can be attributed to inorganic or organic fertilizers are likely to be due mainly to nutrient losses before application (e.g., ammonia [nitrogen] losses during storage of manure) or after application (e.g., leaching of nitrates from the soil to groundwater).

Adverse health effects can also be attributed to the contaminants present in fertilizers (e.g., potentially toxic trace elements) and pathogens (Udeigwe *et al*. 2015; Rodríguez Eugenio, McLaughlin and Pennock 2018). [Chapter 9.3] Direct effects of fertilizers on human health can occur through contact, inhalation (e.g., exposure to ammonia in fertilizer factories or to manure in livestock production systems, exposure to particulate matter [$PM_{2.5}$], exposure to dusts from manure) (US EPA 2016) or explosions during transportation or storage (US EPA 2015; Laboureur *et al*. 2016; Guglielmi 2020). [Chapter 9.3.1] Indirect effects can occur through ingestion. For example, toxic trace elements in fertilizers can contaminate drinking water (Shukla and Saxena 2018; Jin *et al*. 2019) or enter the food web through their application to the soil, uptake by plants, and ingestion of contaminated food by humans or of contaminated feed by animals (Gall, Boyd and Rajakaruna 2015; Wiggenhause *et al*. 2019). [Chapter 9.3.2]

The links between fertilizer use and both environmental and health effects can be complex. Some of the effects could be gender or age specific. For example, dietary exposure to non-lethal toxic doses of nitrates has been associated with methemoglobinemia in infants, pregnancy loss, birth defects and infant mortality, among others (Gupta, Gupta and Gupta 2017). Manure can be contaminated by environmental oestrogen, and some studies have suggested a relationship between environmental oestrogen and breast cancer (Adeel *et al*. 2017). [Chapter 9.3.2]

Links between fertilizers and human health remain largely unknown. A systematic review of epidemiological studies showed that the outcomes with the largest number of studies were cancers for inorganic fertilizers and infections for organic fertilizers (Box 9.3-2). An appraisal of published systematic reviews and/or

meta-analyses of potential associations between exposure to cadmium (small amounts of which are found in phosphorus fertilizers) with human health-related outcomes included kidney disease, cancers, endocrine disorders and cardiovascular disease. Some reported findings were significant and others were not significant or inconclusive. Fertilizers were mentioned in some publications, but no study assessed fertilizer use as a source of cadmium exposure (Box 9.3-2). [Chapter 9.3.2]

Antimicrobial resistance (AMR) in humans has been associated with the presence of manure and contact with livestock (Huijbers *et al*. 2014; Huijbers *et al*. 2015; Muloi *et al*. 2019). Antimicrobial resistant bacteria, which have been detected in manure and sewage sludge (biosolids), end up in water and soil (Huijbers *et al*. 2015). [Chapter 9.3.4]

The environmental effects of fertilizer use can have an adverse impact on human health and nutrition. For example, degradation of the terrestrial and aquatic environments affects the sustainability of food production.

Studies have suggested that changes in human diet can affect the environment and health. For example, it has been proposed that increased adoption of plant-based diets would have positive effects on the environment and health. Greater intake of more plant-based foods would likely reduce the amounts of fertilizers used (Marlow *et al*. 2009; Sabaté and Soret 2014). [Chapter 9.4]

While there are indications that fertilizer use (especially when it is excessive, inefficient or irresponsible) may have adverse effects on the environment and health, some of the potential impacts of these effects remain unknown. [Chapter 9.5]

9.2 Adverse environmental effects of fertilizer use

Fertilizer use is related to air pollution, as well as to the contamination of terrestrial and aquatic environments. Air pollution related to fertilizer use includes the contribution of their use to anthropogenic greenhouse gas (GHG) emissions, ammonia emissions and particulate matter ($PM_{2.5}$) (Chapter 9.2.1). Contaminants from fertilizers in the terrestrial and aquatic environments include nutrients (e.g., from excessive application of fertilizers), metals (e.g., cadmium), endocrine disrupters and antibiotics (Chapters 9.2.2 and 9.2.3).

Excess nutrients from applied fertilizers may be transported to nearby water bodies (e.g., streams and lakes) by means of rainwater, erosion, tile drains and other drainage systems, irrigation channels and seepage, and on to underground water supplies through leaching. Contaminants in fertilizers (e.g., trace elements and pathogens in biosolids) can also pollute soils and water bodies.

9.2.1 Air

The contribution of fertilizers to greenhouse gas emissions

GHG emissions from crop and livestock activities, forestry and land use mainly comprise methane (CH_4) and nitrous oxide (N_2O) emissions from crop and livestock activities and carbon dioxide (CO_2) emissions from land use and land use change (Tubeillo *et al*. 2015).

In 2017 global emissions from crop and livestock activities and agricultural land use (largely deforestation and peatland degradation) were about 20 per cent of carbon dioxide equivalent (CO_2eq) emissions from all human activities, with crop and livestock activities contributing about 11 per cent (FAO 2020) (Figure 9.2-1). However, the contribution of crop and livestock activities and agricultural land use to global emissions has been declining, for example from 29 per cent in 1990

Figure 9.2-1 Agriculture's shares of regional and global carbon dioxide equivalent (CO_2eq) emissions in 2017. FAO (2020).

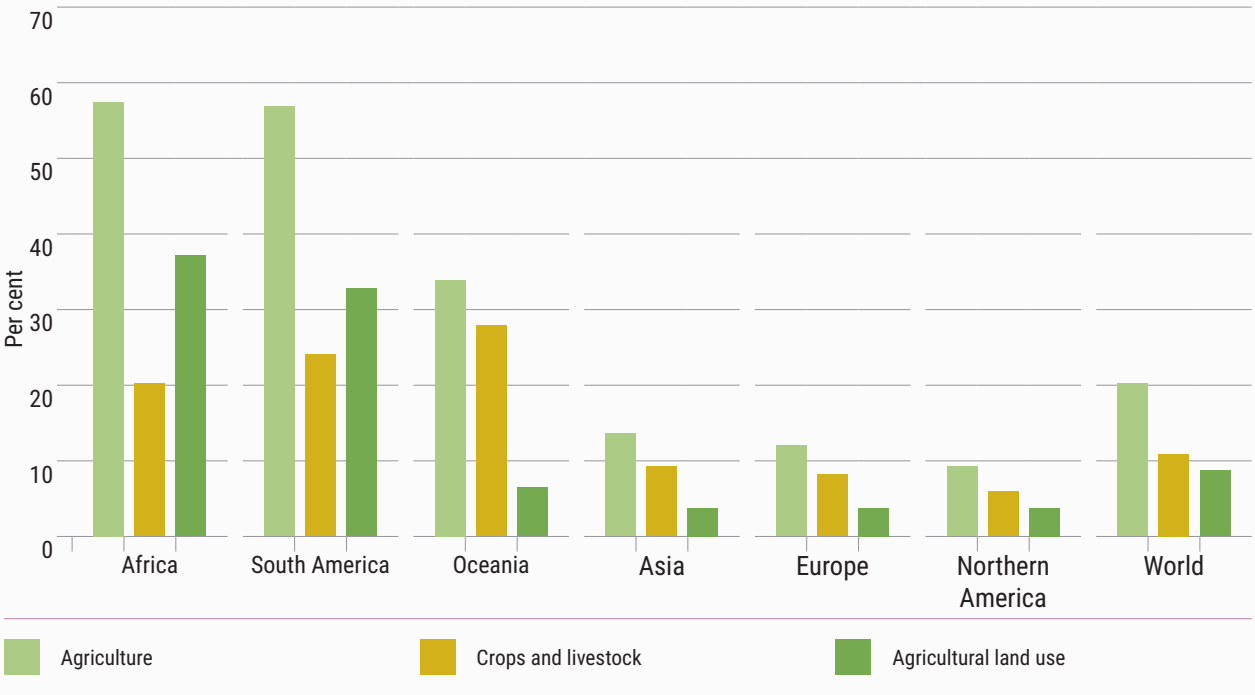
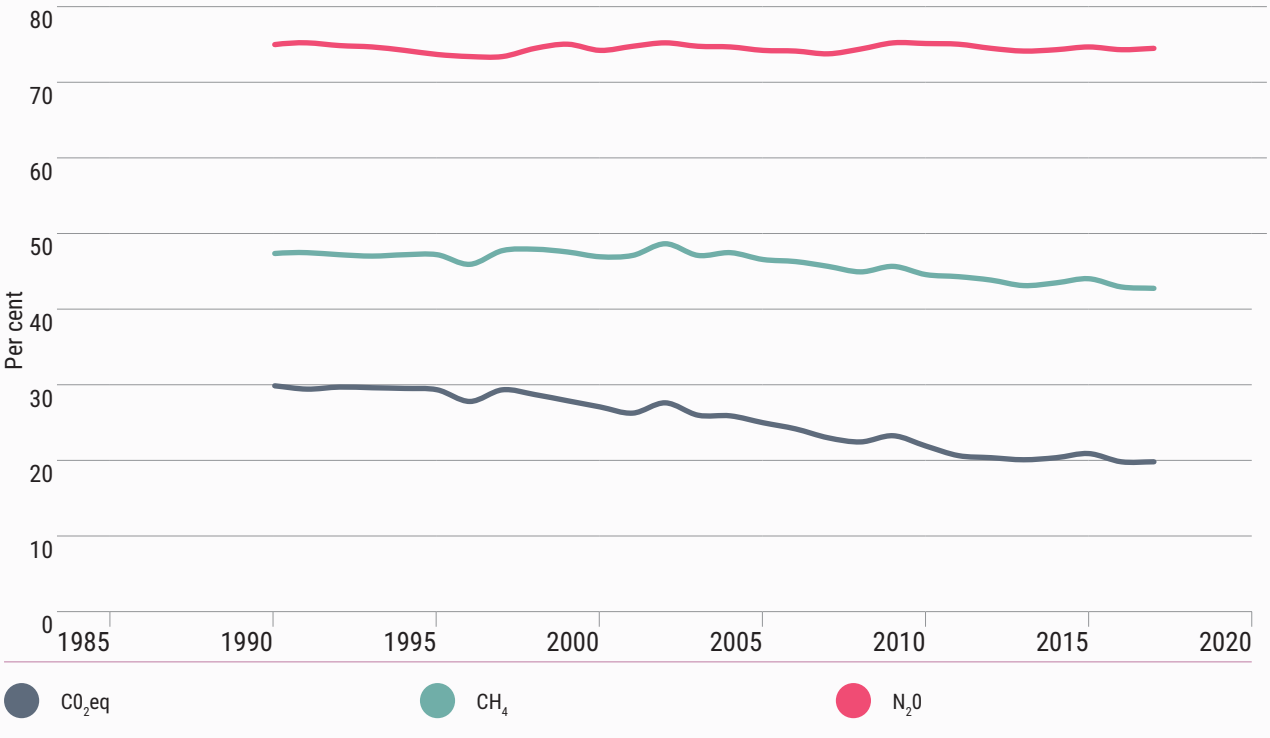
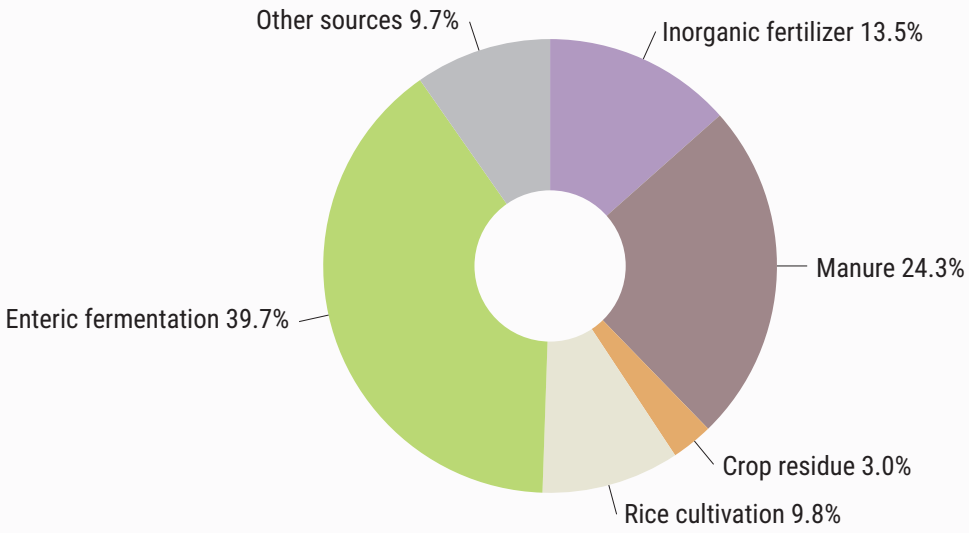


Figure 9.2-2 Agriculture's shares of methane (CH_4), nitrous oxide (N_2O) and carbon dioxide equivalent (CO_2eq) emissions, 1990-2017. FAO (2020).



to 20 per cent in 2017 (Figure 9.2-2). The decline is partially due to a decrease in agricultural land use emissions (from 6.9 to 5.0 gigatons [Gt] CO_2eq between 1990 and 2017). It is likely that the contribution of crop and livestock activities did not change much during this period.

Figure 9.2-3 Contribution of different sources of agricultural emissions from crop and livestock production (per cent). Tubiello *et al*. (2013).



Crop and livestock activities, forestry and land use accounted for 47 per cent and 74 per cent of total CH_4 and N_2O emissions, respectively, in 1990 and 42 per cent and 75 per cent of these emissions in 2017; CH_4 and N_2O emissions were largely from crop and livestock activities within the farm gate (FAO 2020). A 2021 study by UNEP also reported that approximately 40 per cent of CH_4 emission come from Crop and livestock activities (UNEP and Climate and Clean Air Coalition 2021)

Crop and livestock sources of CH_4 are mainly enteric fermentation, rice cultivation and manure management, while those of N_2O are manure, inorganic fertilizers, and emissions from crop residues (Tubeillo *et al*. 2013). Livestock production is a key contributor to agricultural emissions (Figure 9.2-3).

Studies have reported differences in estimates of GHG emissions among the various tools used to produce these estimates. For example, GHG emissions estimated for 2006 in the Republic of Indonesia were close to 500 million metric tons (Mt) and 1,400 Mt of carbon dioxide equivalent per year (CO_2eq yr^{-1}), according to the National Reducing Emissions from Deforestation and Degradation Agency (BP REDD+) 2015 and the Indonesian National Carbon Accounting System (INCAS) 2015 inventories, respectively (Austin *et al*. 2018).

The refinement of the 2006 Intergovernmental Panel on Climate Change (IPCC) Guidelines on National Greenhouse inventories, published in 2019, takes into consideration the climate and sources of nitrogen in estimating emissions, for example by including separate default values for organic nitrogen inputs and inorganic nitrogen fertilizers in wet climates instead of a single default value for both these nitrogen sources (Hergoualc'h et al. 2019). This suggests that GHG emissions could be over- or underestimated in some regions.

GHG emissions also occur during the manufacture of fertilizers. For every ton of nitrogen fertilizer produced and used in China, GHG emissions have been estimated to be 5.1, 0.9 and 5.2 tons of CO_2eq, respectively, during ammonia synthesis, conversion of ammonia to fertilizer products and application of fertilizer in fields (Zhang, W. *et al*. 2013). In addition, fertilizer production has been reported to contaminate the environment with trace elements, for example cadmium and arsenic during production of phosphate fertilizer in Brazil (Mirlean and Roisenberg 2006).

The application of nitrogen fertilizers has been associated with N_2O emissions (Albanito *et al*. 2017). These emissions are higher in the case of soils to which both inorganic and organic

fertilizers have been applied rather than organic fertilizer alone (Charles *et al*. 2017). Between 2007 and 2016, global N_2O emissions from natural and anthropogenic sources were estimated to be about 17.0 MT of nitrogen per year, of which anthropogenic sources contributed about 43 per cent (7.3 million tons of nitrogen per year), according to a study by Tian *et al*. (2020). The authors reported that anthropogenic emissions from nitrogen additions to land under crops (including manure left on pastures) contributed about 48 per cent (3.5 MT of nitrogen/year) of anthropogenic emissions while manure management contributed about 3 per cent (0.2 MT of nitrogen/year).

Efficient fertilizer use can help reduce these emissions. Based on a meta-analysis carried out to evaluate the effectiveness of enhanced efficiency fertilizers in reducing N_2O emissions in cereal production systems, Thapa *et al*. (2016) reported that nitrification inhibitors, double inhibitors (DIs: urease plus nitrification inhibitors) and controlled-release nitrogen fertilizers consistently reduced N_2O emissions compared with conventional nitrogen fertilizers across soil and management conditions. However, the effectiveness of nitrification and urease inhibitors was reported to depend on environmental factors and management practices (Abalos *et al*. 2014). In addition, N_2O fluxes were influenced more by the amount of nitrogen added than the nitrogen source (fertilizer, legume biomass or animal manure), but in coarse soils N_2O emissions from manure tended to be higher than those from inorganic nitrogen fertilizers (Han, Walter and Drinkwater 2017).

GHG emissions are the most important driver of human-induced climate change (Jantke *et al*. 2020). In January 2020, 11,000 scientists from throughout the world signed a letter (Ripple *et al*. 2019) in which they highlighted the urgent need to increase action to reduce GHG emissions, which are rising rapidly with increasingly damaging effects on the Earth's climate.

The Intergovernmental Panel on Climate Change (IPCC) (2018) has presented four illustrative model pathways for achieving the net emission reductions needed to limit global warming to 1.5°C above pre-industrial levels, with no or limited overshoot. One of these pathways (designated P2), which has a "broad focus on sustainability", includes global reductions of CH_4 emissions from agriculture in 2030 by 48 per cent relative to 2010 and of N_2O emissions from agriculture by 26 per cent in the same year, also relative to 2010. The other three pathways include smaller reductions of these agricultural emissions, or the actions they describe are projected to be able to achieve the overall net emission reductions required even if there are slight increases in these CH_4 and N_2O emissions (IPCC 2018).

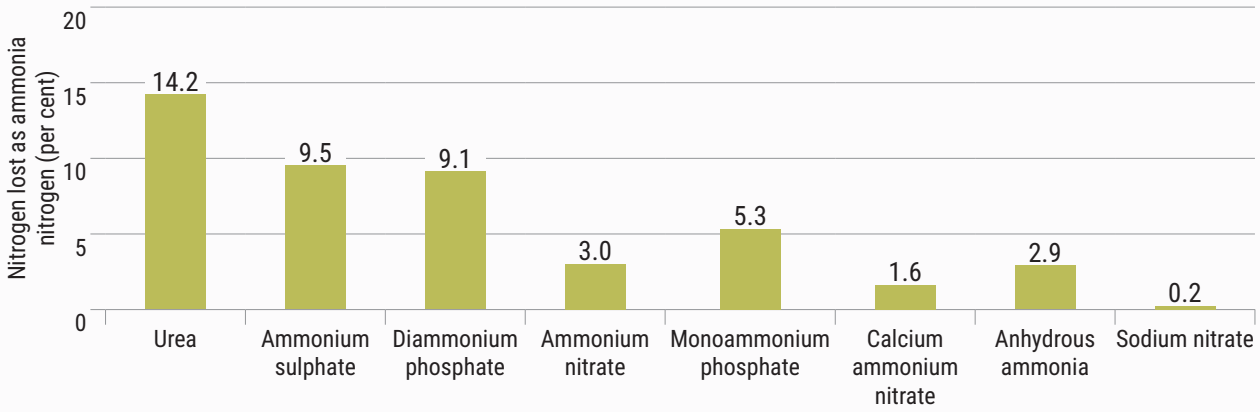
Ammonia emissions and $PM_{2.5}$

Ammonia is the main $PM_{2.5}$ (fine particles with aerodynamic diameters ≤2.5 µm) precursor from agricultural activities (Bauer, Tsigaridis and Miller 2016; Wu, Y. *et al*. 2016). The gas is emitted from heavily fertilized fields and livestock waste. It then combines with other precursors (mainly nitrogen oxides and sulphates) to form particulate matter. Other precursor gases include sulphur dioxide, nitrogen oxides and volatile organic components (VOCs) (Li, X. *et al*. 2015).

Particulate matter (PM) refers to solid particles and liquid droplets in the air. These particles can be from direct emissions (primary particles) or be formed from precursors (secondary particles). Primary particles are emitted directly from anthropogenic sources (e.g., industry, power plants, vehicles) or natural ones (e.g., outdoor biomass burning, dust storms). In the European Union (EU) agriculture was estimated to contribute 4 per cent of $PM_{2.5}$ (European Environment Agency [EEA] 2017). In Europe, parts of the United States and China about 50 per cent of precursors from anthropogenic sources have been estimated to be from agriculture (Bauer, Tsigaridis and Miller 2016). $PM_{2.5}$ associated with agriculture has largely been attributed to the use of nitrogen fertilizers and animal husbandry.

According to a EU report (Amann 2017), application of inorganic fertilizer contributed approximately 20 per cent of total ammonia emissions in the EU-28 region and manure management about 75 per cent. In the United

Figure 9.2-4 Amount of nitrogen lost as ammonia, as a percentage of nitrogen applied, for major inorganic fertilizers. Hergoualc'h *et al*. (2019).



States, about 80 per cent of ammonia emissions are from agriculture, of which 58 per cent from livestock manure, according to a US EPA report cited by Rotz (2020).

Based on data from two meta-analyses and other studies, the amount of nitrogen from fertilizers lost as ammonia, as a proportion of the total nitrogen applied, was estimated to be about 14 per cent for urea and ranged from 0.2 per cent to 9.5 per cent for other fertilizers (Figure 9.2-4) (Hergoualc'h *et al*. 2019). Increased use of nitrogen fertilizers has contributed to an increase in ammonia emissions, with South Asia accounting for more than 50 per cent of total global emissions (Xu, R. *et al*. 2019).

$PM_{2.5}$ among others causes reduced visibility in the air and when deposited on the ground or water. It can cause acidification, alter nutrient balance and affect ecosystem diversity (US EPA 2020b; US EPA 2020c; WHO 2018).

For the adverse health effects of $PM_{2.5}$, see Chapter 9.3.3.

Ammonia emissions and pollution

Apart from contributing to fine particulate matter in the atmosphere, ammonia is a key acidifying pollutant, for example contributing about 30 per cent of total emissions of acidifying pollutants in OECD countries (OECD 2013). Ammonia can contribute to nutrient imbalances, soil acidification and eutrophication when deposited in an ecosystem (OECD 2013; Royal Society and Rand Europe 2018). It also contributes to acid rain (or acid deposition), which damages vegetation (including crops and forests), causes acidification of soils, lakes and rivers, and has adverse effects on biodiversity (FAO 2002; US EPA 2020a).

9.2.2 The terrestrial environment

Fertilizers can have adverse environmental effects. For example, a negative impact on soil's physical and chemical properties, soil organisms (e.g., microorganisms, invertebrates and animals) and plants. For the positive environmental effects of fertilizers see Chapter 10.3.4.

Fertilizers may contain contaminants such as toxic concentrations of trace elements, pathogens and endocrine disrupters (this list is not exhaustive) (Chapter 9.3.2). For example, municipal biosolids have been found to be a sink for many pharmaceuticals and the chemicals in personal care products (Rodríguez Eugenio, McLaughlin and Pennock 2018). Applying contaminated fertilizers to soil may introduce these contaminants into the environment, possibly causing harm to beneficial organisms (Rodríguez Eugenio, McLaughlin and Pennock 2018).

Sources of microplastics in soils include sewage sludge (He et al. 2018; Weithmann *et al*. 2018; Guo, J. et al. 2020), controlled-release fertilizers,

compost, and wastewater used for irrigation (Gionfa 2018). In addition, plastics used in intensive agriculture (e.g., in greenhouses or as mulching material) can fragment to microplastics (Steinmetz *et al*. 2016).

Fertilizers and soil quality/properties

Excessive or inefficient use of fertilizers can contribute to soil acidification. Human-induced acidification of agricultural soils is primarily associated with product removal (e.g., the removal of all harvestable materials) or increases in nitrogen and sulphur inputs (e.g., through excessive use of fertilizers) (FAO 2015). For example, in China the acidification of agricultural soils is largely associated more with nitrogen fertilization than with acid deposition (Guo, J. *et al*. 2010). A meta-analysis of 106 studies estimated that nitrogen addition reduced soil pH by 0.26 on average globally (Tain and Niu 2015). Moreover, acidification may contribute to further soil pollution through the mobilization of toxic trace elements (Rodríguez Eugenio, McLaughlin and Pennock 2018).

Nitrogen fertilization can affect soil biodiversity, contributing to the decomposition of soil organic matter and crop residues (Li, X.G. *et al*. 2017) and may lead to degradation of soil quality (FAO *et al*. 2020).

Sewage sludge may contain toxic trace elements (Mortvedt 1996). Deposition and accumulation of these elements in the soil can influence the relative availability of nutrients to plants. For example, increased concentrations of zinc in the soil can lower the accumulation of lead in wheat grain (Nan *et al*. 2002).

Other contaminants which can be found in fertilizer, and which are increasingly considered important, include endocrine disruptors, organic contaminants and microplastics (also see Chapter 9.3.2). Not much is known about the effects of these contaminants on soil properties. However, microplastics have been reported to affect soil's biophysical properties (de Souza Machado *et al*. 2019).

Soil microorganisms

Beneficial microorganisms in agriculture include microorganisms that break down dead organic matter, mycorrhizal fungi that contribute to improved retention of nutrients and water in the soil and facilitate root penetration leading to improved nutrient use by plants, and nitrogen-fixing rhizobia (FAO *et al*. 2020).

On the other hand, some soil-borne microorganisms can cause diseases in humans (Rodríguez Eugenio, McLaughlin and Pennock 2018). See Chapter 9.2 on the adverse human health effects of fertilizer use for additional information.

Excess fertilizer applications typically reduce the abundance of mutualistic soil biota, which enables an increase in pathogenic microbiota and pests (Wall, Nielsen and Six 2015).

The application of nitrogen fertilizer has been associated with reduced microbial biomass, and excessive amounts of nitrogen applied over a long period have been associated with declines in the abundance of microbes and fungi (Treseder 2008). The relative abundance of different types of microorganisms within groups can also be affected. For example, in a study by Dai *et al*. (2018) nitrogen fertilizer reduced the abundance of Acidobacteria and increased the relative abundance of Proteobacteria and Actinobacteria. In addition, Liu, P. *et al*. (2017) demonstrated that changes in bacterial communities can differ with type of fertilizer. A study by de Graaff et al. (2019) reported that the effects of nitrogen fertilizer on different groups varied. For example, the authors found that inorganic nitrogen fertilizer can have negative and positive impacts on the diversity of arbuscular mycorrhizal fungi and bacteria, respectively.

Tillage, sometimes used to incorporate fertilizers into the soil, can have adverse effects on soil microorganisms. This practice can contribute to increased mineralization of soil organic matter due to greater exposure to microbial decomposers and optimal moisture and temperature (Dhaliwal et al. 2019).

A number of studies have demonstrated or reported that contaminants, that is, potentially toxic trace elements (Chu 2018; Mishra, Singh and Arora 2017), endocrine disrupters and pharmaceuticals (Frková *et al*. 2020) potentially present in fertilizers can interfere with microbial biomass and diversity (Rodríguez Eugenio, McLaughlin and Pennock 2018). However, although they may be present in fertilizers, fertilizers are not the only source of these contaminants in soil (Shi *et al*. 2019). For example, toxic concentrations of trace elements in soil can result in adverse effects on soil microbial diversity and activity (Xie *et al*. 2016; Srivastava et al. 2017); however, there are indications that organic inputs, particularly those from waste streams, probably pose greater risks with respect to soil pollution than inorganic fertilizers (Lugon-Moulin *et al*. 2006; Pizzol, Smart and Thomsen 2014). For examples of contaminants in fertilizers, see Chapter 8, Table 8.4.1.

In experiments by Kuzikova *et al*. (2019) nonylphenol, an endocrine disrupter which has been reported in sewage sludge, for example by Mao *et al*. (2012), was associated with reduced diversity of microbial species when added to the soil. Pharmaceuticals, which can affect the diversity of the soil community (Pino-Otín *et al*. 2017), have been detected in manure (Cycoń, Mrozik and Piotrowska-Seget 2019). Despite an absence of studies on the adverse effects of fertilizers on soil microorganisms, the studies presented here suggest that fertilizers, especially untreated organic fertilizers, have the potential to adversely affect soil microorganisms.

Soil invertebrates

Beneficial soil invertebrates in agriculture include invertebrates that accelerate litter decomposition and nutrient cycling, improve soil aggregation and porosity, and aid in the control of plant pathogens (FAO *et al*. 2020). These invertebrates can also regulate the activity or composition of soil microbial communities.

The global decline in insect populations has been partially attributed to the use of pesticides and inorganic fertilizers (Sánchez-Bayo and Wyckhuys 2019).

Fertilizers have also been reported to have positive and negative effects on invertebrate biodiversity and populations for some types of soil invertebrates. For example, increased soil fertility, or application of nitrogen, has been associated with increased the populations of soil mites (Cole, Buckland and Bardgett 2005), increased growth of nematodes (Liu, T. *et al*. 2015), increased earthworm populations (Edwards and Lofty 1982) and increased abundance of dipterans and beetles (Lemansk and Scheu 2015). However, Minor and Norton (2004) reported that predatory mites (Mesostigmatid) benefited from the application of biosolids, while saprophagous and mycophagous mites (Oribatida) were adversely affected. Adverse effects of the use of excessive fertilizers have also been reported in the case of other invertebrates. Nitrogen application resulted in reduced species richness for nematodes (Liu, T. *et al*. 2016) and the use of excessive amounts of sludge produced adverse effects in earthworms (Edwards and Lofty 1982).

The negative effects of nitrogen inputs on soil organisms could be partly due to acidification. In a study on nitrogen inputs by Chen, Lan and Bai (2015), reduced bacterial, fungal and nematode biomass was mainly associated with soil acidification. Earthworms have also been reported to be affected negatively by soil acidity (Solomou *et al*. 2013).

High concentrations of toxic trace elements in soil can contribute to reduced earthworm density and diversity (Tovar-Sánchez *et al*. 2018; Wang, K. *et al*. 2018), although it is likely that these effects rarely occur as a result of normal fertilization practices.

There are indications that endocrine disrupters can have adverse effects on soil invertebrates. For example, in a study on the effects of bisphenol A (BPA) on earthworms, the authors concluded that observed changes in gene expression could have such effects on earthworm populations (Novo *et al*. 2018).

Studies have associated microplastics in soil with the health of soil organisms (Helmberger, Tiemann and Grieshop 2019). For example, an increase in concentrations of microplastics in litter was associated with reduced growth of earthworms

(Lwanga *et al*. 2016). The review by Helmberger, Tiemann and Grieshop (2019) demonstrates that this a new and growing area of research and that much is still unknown.

Some studies have compared the effects of inorganic and organic fertilizers on soil invertebrates. Organic fertilizers increased the total abundance of soil fauna, whereas low level inorganic fertilizers caused increases in four- and nine-year-old poplar stands but not in 20 year-old stands (Wang, S. 2016).

Vertebral Animals

Vertebral animal poisoning from drinking water contaminated with cyanobacteria has been reported (Svirčev *et al*. 2019). Effects of oral exposure of animals to cyanotoxins include lesions in the small intestine and hepatic system and damage to the DNA in intestines (Kubickova *et al*. 2019).

Contaminants in fertilizer can enter the food chain and can have adverse effects due to , the intake of contaminated feed.

Perchlorate, an endocrine disruptor of concern to human health (Chapter 9.3.2), has been shown to interfere with iodine uptake by the thyroid gland in amphibians, leading to retardation of metamorphosis and reduced growth rates (Miyata and Ose 2012). The thyroid effect of the chemical has also been reported in fish (Matthiessen, Wheeler and Weltje 2018)

Fluoride, an impurity in phosphatic fertilizer, was generally found to accumulate in the upper soil profile following long-term application of superphosphate fertilizer in pastures (McLaughlin *et al*. 2001). Although the risk of fluorosis in grazing animals was reported to be low, continued applications could increase this risk. In New Zealand a model estimated that ingestion of soil during grazing could cause chronic fluorosis in sheep and cattle during the winter (Cronin *et al*. 2000). Actual cases of fluorosis have been reported. Chronic fluoride toxicosis (with up to 15 per cent of a herd going lame) was reported in an extensive beef cattle herd in northern Australia, where large quantities of fertilizer-grade phosphate were fed to animals as part of a mineral supplement (Jubb et al. 1993).

Plants

Studies on the effects of fertilizer use on plant biodiversity have mostly focused on grasslands and natural plant communities. Several studies indicate that the use of inorganic nitrogen fertilizer can influence plant biodiversity. A systematic review concluded that plant biodiversity in mountain grasslands is reduced by nitrogen addition (Humbert *et al*. 2016). A meta-analysis of studies assessing the effects of nitrogen addition on terrestrial natural and semi-natural plant communities by Midolo *et al*. (2019) concluded that applying nitrogen fertilizers reduced plant species richness and abundance. A study on the impacts of 120 years of inorganic fertilizer and manure use in grassland by Kidd *et al*. (2017) showed that there could be differences in plant species richness and abundance depending on nutrient sources. For example, in the study yearly application of manure and inorganic nitrogen fertilizer were both associated with fewer plant species than the control (no inorganic fertilizer or manure).

Plants exposed to toxic trace elements in the soil show reduced growth and reduced yields (Srivastava *et al*. 2017). Exposure to toxic trace elements can also contribute to reduced produce quality. For example, in a wheat study it was associated with decreased protein content in plant parts and increased concentrations of these elements in grain (Athar and Ahmad 2002). Exposure to excessive amounts of trace elements can influence plant populations. For example, plant diversity and population sizes were affected negatively in grassland polluted by trace elements (Hernández and Pastor 2008).

The application of compost made from plant material can result in increased crop disease, especially if fertilizers are not well composted (University of Nebaska-Lincoln 2020). Crop residues are potential sources of disease spores (Pizolotto *et al*. 2019) and insects for the next crop. Hence, if such residues are partially composted and applied to the soil as fertilizer,

disease infections and insect pest infestations can affect the next crops.

Endocrine disrupters have been shown to affect biological nitrogen fixation in legume plants by interfering with communication between plants and soil bacteria (Fox *et al*. 2001) and to reduce leaf chlorophyll content and plant growth (Kim, Kwak and An 2019).

Plants have been reported to take up antibiotics present in manure (Kumar 2005) and to take up AMR genes from antimicrobial-spiked manure (i.e., dairy manure fortified with individual antimicrobial compounds belonging to the sulfonamide and tetracycline classes; Mullen *et al*. 2019), indicating that such contaminants can pose a risk to humans and animals through the food chain.. Manure application could enhance the prevalence and stimulate the propagation of antibiotic resistance in agricultural soils (Pérez-Valera *et al*. 2019; Wu, N. *et al*. 2020). However, the risk may not increase with increased use of manure (Peng *et al*. 2018). For example, in a study by Liu, P. *et al*. (2017) the abundance of soil antimicrobial resistance genes increased after manure fertilization but gradually returned to normal.

The terrestrial ecosystem and food webs

An increase in mineral nitrogen and phosphorus from polluting sources (e.g., atmospheric deposition and run-off water), and in agroecosystems due to excessive use of mineral fertilizers, have consequences for plant growth and microbial decomposition that can cascade to soil food webs. This causes nutrient imbalances (typically carbon:nitrogen [C:N], carbon:phosphorus [C:P] or nitrogen:phosphorus [N:P]) between consumers and their resources, strongly constraining nutrient cycling and limiting consumer reproduction and growth (Maaroufi and De Long 2020). The effects of nutrient imbalance on soil organisms and food webs can be direct (in the case of organisms such as bacteria and fungi that take up nutrients directly from the soil solution) or indirect, via alterations of the nutritional content of primary producers and their litter inputs, with cascading consequences for both green and brown food webs.

Interactions exist between soil invertebrates and soil microorganisms Soil invertebrates prey on soil microorganisms. They also compete with them for resources (Scheu, Ruess and Bonkowski 2005). In addition, they are involved in the mixing and breakdown of plant residue inputs to soil (Chamberlain *et al*. 2006) and therefore make organic matter accessible to microorganisms (Bray and Wickings 2019). It is likely that some practices affect both soil invertebrates and soil microorganisms, but not necessarily in the same manner. It is also likely that, as with soil microorganisms, the effects of inorganic and organic fertilizers on soil invertebrates can be positive.

9.2.3 The aquatic environment

Nutrient pollution in the aquatic environment contributes to eutrophication and affects the dynamics of the aquatic food web. For example, it has adverse effects on coral reefs, algae, aquatic invertebrates, amphibians and fish. It also has an impact on water quality and the value of water bodies and waterfront properties (Chapter 10.2.2) and can affect human health (Chapter 9.3.2).

Protecting, restoring and monitoring water quality can have positive effects on both terrestrial and marine ecosystem health, as well as, for example, reducing the costs of drinking water treatment. According to the United Nations summary progress update for Sustainable Development Goal (SDG) 6 ("Ensure availability and sustainable management of water and sanitation for all"), there are currently insufficient country data to estimate global status or trends with regard to meeting this SDG. Of the 48 countries that reported in both 2017 and 2020, 21 are on track to improve water quality; urgent action is therefore needed globally to improve monitoring systems for surface water and groundwater and to define water quality standards (UN-Water 2021).

Some data exist on regional and national clean-up initiatives. In North America, harmful algal blooms and hypoxia in Lake Erie (Barbiero *et al*. 2018; Sgro and Revie 2018; US EPA 2020d) and Chesapeake Bay (Chesapeake Bay Foundation 2019; US EPA 2020e) have threatened ecosystem and human health for a number of years.

This problem is being addressed through cooperation between national, state and other bodies, and progress is reported (Governments of Canada and Ontario 2018; US EPA 2020f; Chesapeake Bay Foundation 2021). For farmers, proposed measures to reduce agricultural inputs of nutrients to both Lake Erie and Chesapeake Bay include following the 4Rs of nutrient stewardship (right time, right place, right rate, and right source), conservation tillage, and growing cover crops (US EPA 2018; Chesapeake Bay Program 2019).

Eutrophication

This section discusses eutrophication at the global level, with its causes and the process involved and elaborates on the effects of aquatic pollution on coral reefs, algae, aquatic invertebrates, amphibians and fish. It should be noted that no relevant reports on ecotoxicological field studies were found.

Since the 1960s the number of water bodies in which hypoxia associated with eutrophication has been reported has been increasing (Díaz *et al*. 2012; Rabalais *et al*. 2014; Breitburg, Gregoire and Isensee 2018; Breitburg *et al*. 2018). There were about 10 documented sites in 1960 (Figure 9.2-5); the number increased to 762 by 2013, with 479 experiencing hypoxia, 55 having once experienced hypoxia but improving, and 228 experiencing other symptoms of eutrophication including algal blooms, species loss, and impacts to coral reef assemblages (World Resources Institute 2013).

Eutrophication is triggered by a change in the quantity, relative proportions or chemical forms of nitrogen and phosphorus entering aquatic systems. It is influenced by environmental factors such as temperature and light. Sources of nutrients include agriculture, human sewage, urban run-off, industrial effluent, and fossil fuel combustion (Selman *et al*. 2008; Díaz *et al*. 2012; Le Moal *et al*. 2019).

Attempts have been made to estimate contributions from various sources. Between 1900 and 2000 global nutrient delivery to streams increased from 34 to 64 Mt nitrogen and from 5 to 9 Mt phosphorus per year (Beusen *et al*. 2016). The contribution of agriculture to nutrients in surface water increased by about five times for nitrogen (6 to 33 Mt N/year, which is 19 per cent to 51 per cent of the total) and slightly more than doubled for phosphorus (2 to 5 Mt P/year, which is 35 per cent to 56 per cent of the total). Apart from food production, another key contributor could be non-food items (clothing, goods for shelter, services and other manufactured products), which were estimated to account for 35 per cent of global marine eutrophication and 38 per cent of the global freshwater eutrophication in 2011 (Hamilton *et al*. 2018).

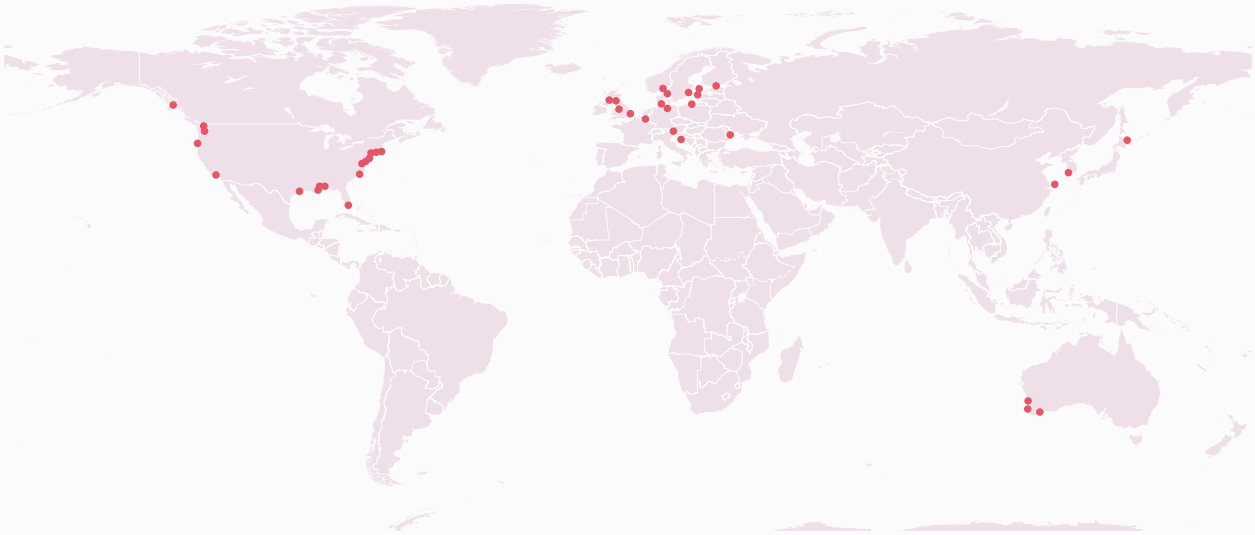
The increase in nutrients in aquatic ecosystems contributes to an increase in plant biomass and surface blooms of cyanobacteria (blue-green algae) (Khan *et al*. 2018). The death and decomposition of these undesirable aquatic plants reduces penetration of light in the water and causes depletion of oxygen levels in the water body, with extreme depletion of oxygen leading to the death of aquatic organisms and the formation of hypoxic zones (Khan *et al*. 2018). The formation of hypoxic zones has also been linked to climate change, which is driving losses of nutrients. Rising global temperatures are contributing to decreased oxygen in the water, and hence to the formation of hypoxic zones (Brietburg *et al*. 2018; Delgado 2021).

Coral reefs

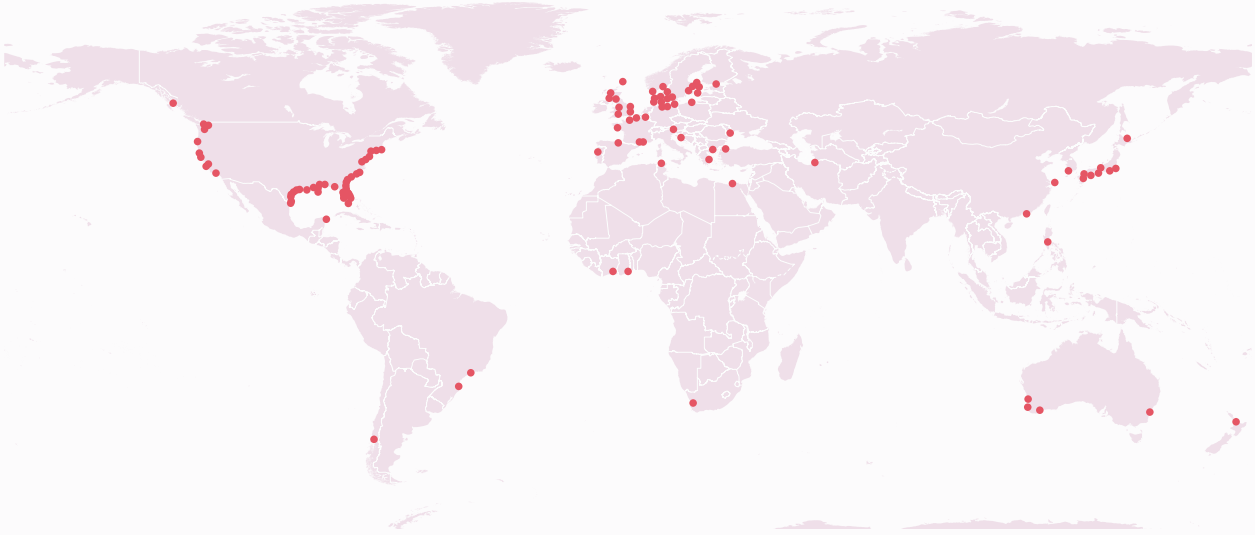
The world's coral reefs are under threat from local activities such as overfishing and destructive fishing, coastal development, climate change, ocean acidification, watershed-based pollution, and marine-based pollution (Wear 2016; Hoegh-Guldberg *et al*. 2017). About 50 per cent of coral reefs have been lost in the past 20 years, and it is feared that 70-90 per cent will disappear during the next 20 years as a result of climate change and pollution; rising sea surface temperatures and acidic waters could eliminate nearly all existing coral reef habitats by 2100 (American Geophysical Union 2020). Anthropogenic land uses (e.g., agriculture and urban development) can contribute to declines in coral cover, diversity, colony size and structural complexity (Carlson, Foo and Asner 2019). Pollution from agriculture has been identified as a threat in at least 25 per cent of coral reefs globally (Burke *et al*. 2011).

Figure 9.2-5 **Global distribution of documented cases of hypoxia related to human activities. Each case is represented by a dot. The number of hypoxic areas is cumulative for the successive time periods.**
Rabalais *et al*. (2010) as reproduced in Díaz *et al*. (2012).

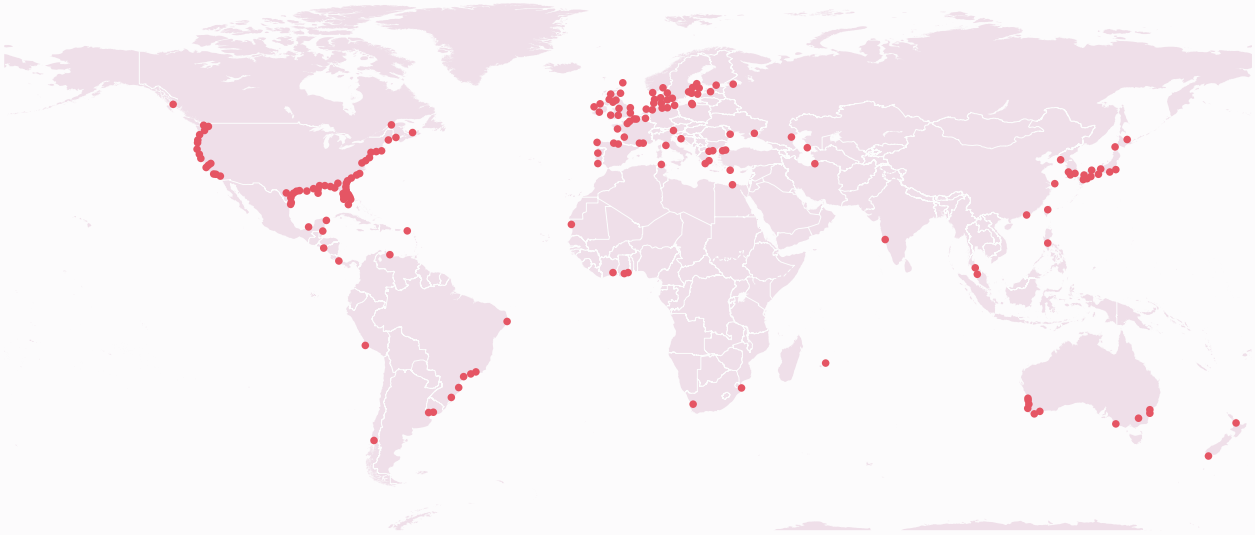
Up to 1969



Up to 1989



Up to 2009



A well-known example of a coral reef adversely affected by human activities is the Great Barrier Reef (GBR) in Australia. Studies have reported reductions in coral cover and seagrasses and animal species (Kroon *et al*. 2016). The decline has been partially attributed to nitrogen and phosphorus nutrient loads in rivers in the catchments draining into the GBR (Thorburn, Wilkinson and Silburn 2013). As pointed out by the authors of that study, a key source of nitrogen and phosphorus is the application of substantial amounts to crops. Several measures have been taken to manage the GBR. They include establishing targets to reduce nutrient loads from key catchments, enacting regulations, and setting up a reef trust (Department of Agriculture, Water and Environment of Australia 2020). Scientists have warned that saving coral reefs globally will require more aggressive actions from countries (Gibbens 2020).

Algae

Anthropogenic sources have been related to abundance of algae (Berthold *et al*. 2018). Nutrient supplementation with fertilizers increases algae productivity (Boyd 2018). With excessive nutrient loading, the initial impacts are excessive growth of phytoplankton, microalgae (e.g., epiphytes and microphytes) and macroalgae (i.e., seaweed), which leads to reduction of light; inhibition of coral growth because the algae outcompetes coral larvae for available surfaces to grow; a shift in phytoplankton species and composition; and creation of favourable conditions for the development of nuisance, toxic, and otherwise harmful algal blooms (Selman *et al*. 2008). Harmful algal blooms are mainly the result of cyanobacteria, and some types of harmful cyanobacterial algal blooms produce toxins called cyanotoxins which can affect aquatic ecosystems (Graham, Dubrovsky and Eberts 2017).

Algae are eaten by microscopic animals (zooplankton) and insects, both of which are then eaten by small fish which may in turn eaten by larger predatory fish (Watson and Cichra 2013). However, water quality tends to deteriorate with an increase in nutrient input, subsequently affecting production negatively (Boyd 2018). The onset of oxygen-deficient conditions can greatly alter the community structure of water-column phytoplankton (Gauns *et al*. 2020). According to Ger *et al*. (2016), cyanobacteria compete with other phytoplankton for resources and these other phytoplankton decline during blooms. This study also reports that cyanobacteria are a poorer food source for zooplankton than some other phytoplankton. Hence, feeding on cyanobacteria affects zooplankton growth.

Fish

Reports on fertilizers' effects on fish are largely based on data collected in eutrophic water bodies. Common carp increased and sport fish decreased with an increase in lake trophic status in Canada (Egertson and Downing 2004). Hypoxia had a negative impact on fish survival (Jeppesen *et al*. 2016) and reduced the growth rate of sea bass larvae (Vanderplancke *et al*. 2014). "Taste-and-odour compounds" produced by cyanobacteria may accumulate in fish and make them unpalatable to humans (Graham, Dubrovsky and Eberts 2017). However, a study on algal bloom in 2011 in Lake Erie in North America (the largest bloom in history) did not identify any decline in the value of commercial fishing as a result of this bloom while the weight and value of the fish harvest were above typical values (International Joint Commission 2014).

Inorganic and organic fertilizers are sometimes used to promote algal growth in aquaculture (e.g., in fish and shrimp farming) (Boyd 2018). A World Bank report has suggested that effluents could be used as a source of nutrients in fisheries (White 2017).

Amphibians

Increased nutrients in water can reduced or increased tadpole survival and growth (Mann *et al*. 2009). In experiments the growth of tadpoles was reduced when they were exposed to ammonia and nitrate fertilizers (Garriga, Montori and Llorente 2017) and metamorphosis was delayed upon exposure to nitrates (Wang, M. *et al*. 2015).

Aquatic invertebrates

Acidification and nutrient pollution are among the most important anthropogenic factors threatening

freshwater mussel diversity in Peninsular Malaysia (Zieritz *et al*. 2016). Hypoxia and toxins, from eutrophication, affect reproduction and growth of invertebrates. According to a meta-analysis of impacts of hypoxia in invertebrates, hypoxia reduces reproduction and growth is not as sensitive (Galic, Hawkins and Forbes 2019). Hypoxia was also reported to reduce the growth of oysters (Jeppesen *et al*. 2016). In addition, toxins released upon the death of algal blooms may accumulate in aquatic invertebrates (Bownik 2016)

Elevated nitrogen concentrations in aquatic environments can increase vector populations. For example, the number of mosquitos in rice fields increased with the use of inorganic fertilizer rather than organic manures (Victor and Reuben 2000). Moreover, mosquito survival rates were higher in water with than without ammonium nitrate fertilizer (Muturi *et al*. 2016). In another study Culex mosquitoes laid more egg clutches in nutrient-rich containers than in containers without nutrients (Reiskind and Wilson 2004). Still, Johnson *et al*. (2010) have argued that the effects of environmental nutrient enrichment on disease are complex and multifaceted, varying with the type of pathogen, host species and condition, attributes of the ecosystem, and the degree of enrichment; some pathogens increase in abundance, whereas others decline or disappear.

9.3 Adverse health effects of fertilizer use

Fertilizes are best known for their contribution to food security. Few studies demonstrate that fertilizers have adverse effects on human health. However, they have been identified among the sources of contamination and pollution of air, water and soil and hence can potentially affect human health. For example, nitrates in drinking water have been associated with negative health effects although fertilizers are not their only source. Nitrate sources for humans can be exogenous (e.g., from food and water) and endogenous (the body can make nitrates).

There are reports that inorganic and organic fertilizers are among the sources of pollution (such as increased nitrate concentrations in groundwater and drinking water) that can affect human health (de Paz *et al*. 2009; Ward *et al*. 2018; Yu, G. *et al*. 2020). However, during the systematic review carried out for this report, there were no reports found of fertilizers directly causing human health effects when applied in a responsible and efficient manner or quantifying the contribution of fertilizers to such health effects. For some illnesses, where fertilizer use was suspected to be the cause, studies have been inconclusive or data are lacking.

Based on the existing literature, it seems likely that any negative health effects which can be attributed to inorganic or organic fertilizers are mostly due to nutrient losses before application (e.g., ammonia [nitrogen] losses during storage of manure) or after application (e.g., leaching of nitrates from the soil to groundwater). When fertilizers are handled and used responsibly, these losses are minimized and benefits are maximized (Chapter 10.3).

The perceived, actual and potential links between fertilizers and human health effects are presented in Figure 9.3-1. However, the relationships between these links are simplified in the figure. In real life the pathways connecting fertilizer contaminants/emissions to human health effects are not so distinct and can be rather complex.

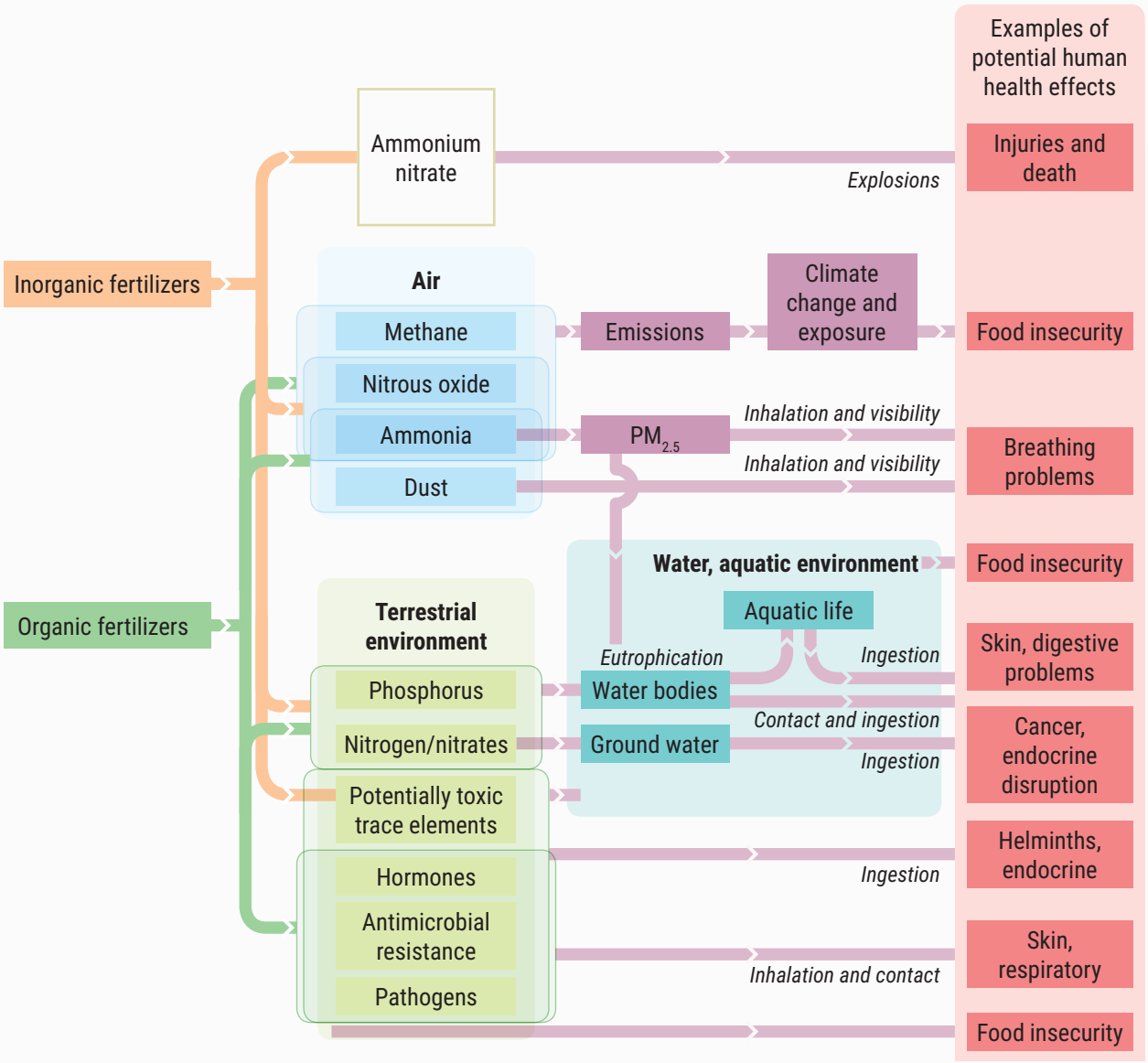
9.3.1 Adverse effects of occupational exposure

The reported adverse effects of occupational exposure to fertilizers mainly concern contact or inhalation of ammonia from fertilizers and manure (US EPA 2016). Another hazard associated with fertilizers is explosions and fires in storage settings involving (reactive) nitrogen fertilizers (US EPA 2015; Laboureur *et al*. 2016; Guglielmi 2020).

Ammonium nitrate and explosions/fires

Incidents involving ammonium nitrate fertilizers, including disasters in which many lives were lost,

Figure 9.3-1 Simplified generic schematic diagram of connections between fertilizers and human health issues.



have been documented (US EPA 2015; Laboureur *et al*. 2016; Guglielmi 2020; European Commission [EC] n.d.). Additional information about these accidents is provided in Chapter 10.2.1 and Figure 10.2-1.

Guidelines and advice on the transport, storage and handling of ammonium nitrate fertilizer (Chapter 7.2) have been developed based on reviews of these incidents (Pittman *et al*. 2014; US EPA 2015). Accidents can be prevented by using fertilizer formulations which reduce the potential for uncontrollable fires and detonation, and safety measures which provide assurance against uncontrollable fires (Gyenes and Wood 2015; Babrauskas 2016; Babrauskas 2017).

Exposure to ammonia

Major sources of ammonia gas include leaks and spills during commercial synthesis, production, storage, processing or transporting of ammonia; refrigeration equipment failure; decaying manure from livestock; application of fertilizers; sewage or wastewater effluent; burning of coal, wood or other natural products; volcanic eruptions; forest fires; and the decomposition of nitrogenous compounds (US EPA 2016). Ammonia is also found naturally

in the environment. It is a component of the global nitrogen cycle and is one of the main precursors of particulate matter ($PM_{2.5}$) from agricultural activities.

Acute toxic effects

Short-term inhalation exposure to high levels of ammonia in humans can cause irritation and serious burns in the mouth, lungs and eyes (US EPA 2016).

In case reports of injury following acute exposure, changes in lung function have been observed. Respiratory effects have also been reported in some, but not all, controlled human exposure studies conducted on volunteers, according to the literature reviewed in US EPA (2016).

Chronic toxic effects

Chronic exposure to airborne ammonia can increase the risk of respiratory irritation, cough, wheezing, tightness in the chest, and impaired lung function in humans (US EPA 2016).

In fertilizer factories respiratory effects following inhalation exposure to ammonia have been reported, according to a toxicological review

Table 9.3-1 Respiratory effects in humans from inhalation exposure to ammonia in fertilizer factories.
Based on US EPA (2016).

Place/country	Description	Effects	References
Urea plant and ammonia plant in a urea fertilizer factory, Bangladesh	Exposures to ammonia and acute respiratory effects were determined for workers in both plants Workers at the ammonia and urea plants were classified as low and high exposure groups, respectively The two groups were compared to controls (from the administration building)	Urea plant workers had higher mean exposure to ammonia and prevalence of acute respiratory symptoms, than ammonia plant workers The symptoms with highest prevalence in the urea plant were chest tightness and cough	Rahman *et al.* (2007) as cited in US EPA (2016)
Two urea fertilizer factories, Saudi Arabia	Study to determine the prevalence of respiratory symptoms and disease among workers chronically exposed to ammonia gas The factories were classified as high and low exposure factories; both factories were compared to controls (administration staff)	Increased prevalence of respiratory symptoms (i.e. bronchial asthma, chronic brochities in high exposure factory than in low exposure factory	Ballal *et al.* (1998)
Ammonia factory, Saudi Arabia	Workers exposed to ammonia divided into two groups (high and low cumulative exposure); both groups were compared with controls	Decreased lung function with greater cumulative ammonia concentration	Ali *et al*. (2001)
Fertilizer chemical plant, India	Workers exposed to irritants in diammonium phosphate (DAP), urea and ammonia plants vs. controls	Decreased lung function in workers from the three plants; DAP workers most affected, followed by ammonia, and urea workers were least affected	Bhat and Ramaswamy (1993)

by US EPA (Table 9.3-1). In a study on healthy humans during ammonia exposure in an exposure chamber, symptoms related to irritation and central nervous effects were reported (Sundblad *et al*. 2004). Fertilizer was among the irritants reported to induce dermatitis in farmers in Poland (Kieć-Swierczyńska, Krecisz and Swierczyńska-Machura 2003).

In a study on rats, ammonia was reported to damage the respiratory system (Perkins *et al*. 2017).

Other contaminants

Dusts in pig housing were found to contain antibiotics (Hamscher *et al*. 2003) and endotoxins (Shin et al. 2019). In a systematic review of the literature (Farokhi, Heederik and Smit 2018) several studies showed a significant relationship between airborne endotoxins and human respiratory health.

Lam *et al*. (2015) reported that occupational health risks associated with wastewater and excreta management in Southeast Asia included diarrhoea, skin infection, parasitic infection, bacterial infection and epilepsy.

The use of phosphate rock, which contains small amounts of naturally occurring radionuclides, in the production of fertilizers generates a by-product called phosphogypsum. Phosphogypsum maintains nearly 80 per cent of its original radioactivity due to ^{238}U decay products such as radon, ^{226}Ra, and polonium, ^{210}Po (Rodríguez Eugenio, McLaughlin and Pennock 2018). World production of this waste is currently around 200 million tons per year (Chernysh *et al*. 2021). Phosphogypsum waste disposed of in water or on land can be a threat to the environment and human health (Saadaoui *et al*. 2017; US EPA 2020g; Chernysh *et al*. 2021). Nevertheless, the literature associating this waste with environmental and human health effects does not appear to be extensive.

9.3.2 Adverse effects of dietary exposure

This section discusses the adverse effects of trace elements, particularly cadmium; adverse effects of nitrates; and adverse effects of urea and pathogens in organic fertilizers. It also covers endocrine disrupters in fertilizers and toxins in eutrophic water bodies, as well as the effects of fertilizers on food quality.

Potentially toxic trace elements

Potentially toxic trace elements can be ingested through food and water intake (Gall, Boyd and Rajakaruna 2015). In the literature both leafy and non-leafy vegetables are reported to be good accumulators of trace elements when grown in contaminated soil (Khan *et al*. 2015). A review of the literature on the abiotic inorganic contaminants in foods by McLaughlin, Parker and Clarke (1999) considered arsenic, cadmium, mercury, lead and selenium to be the most important elements in terms of food chain contamination. According to FAO's report on soil pollution by Rodríguez Eugenio, McLaughlin and Pennock (2018), the fertilizer industry is considered a source of potentially toxic trace elements such as mercury, cadmium, arsenic, lead, copper and nickel and natural radionuclides like ^{238}U, ^{232}Th and ^{210}Po. Livestock manure and sewage sludge are among the sources of trace elements such as chromium, copper, nickel, lead and zinc.

Cadmium

Cadmium is released by natural and anthropogenic sources to the atmosphere, aquatic environments and terrestrial environments (UNEP 2010). According to the literature, over 90 per cent of cadmium released to the environment is from anthropogenic sources (Khan *et al*. 2017). Industrial processes (e.g., smelting of non-ferrous metal ores, fossil fuel combustion, ferrous metal production) are the main sources of atmospheric cadmium; atmospheric deposition contributes to the build-up of soil cadmium (WHO 2019a). Production of rechargeable nickel-cadmium batteries accounts for most global cadmium consumption. Other uses include alloys, anticorrosive coatings, pigments, polyvinyl chloride stabilizers, and semiconductors for solar cells (United States Geological Survey 2020).

Major contributors to cadmium levels in agricultural soils appear to be atmospheric deposition, phosphate fertilizers, sewage sludge

and animal manure (UNEP 2010). Recent studies on the actual contributions of different sources globally were not found. However, according to a recent EU report on the total annual input of cadmium to agricultural soils in Europe (Smolders 2017), phosphate fertilizers contributed about 58 per cent, atmospheric deposition about 25 per cent and manure, sludge and lime about 17 per cent; the annual input from these sources was slightly less than 0.2 per cent of the current stock of cadmium in the soil. Cadmium emissions in the EU fell by about 65 per cent between 1990 and 2016 (EEA 2018). High cadmium concentrations in soil have been associated with industrial activities in several world regions. For example, high concentrations have been reported in rice grown in soils contaminated by industrial discharges in Asia (Chaney 2012) (Box 9.3-1).

Phosphorus fertilizers contain small amounts of cadmium, as this metal occurs naturally in phosphate rock (Roberts 2014). In a study in New Zealand cadmium concentrations in the topsoil were some seven to eight times greater than in deeper horizons; cadmium, phosphorus and uranium were highly correlated, consistent with a common source which was most likely phosphate fertilizer (Salmanzadeh *et al*. 2016). Another study reported that in many New Zealand territorial authorities there were no farms with soil cadmium concentrations beyond the range that naturally occurs in that country (Abraham *et al*. 2016).

A wheat study in which phosphorus fertilizer was labelled with a radioisotope revealed that 6.5-15 per cent of the cadmium in the shoot was from the fertilizer and that most of fertilizer cadmium (>97per cent) remained in the roots and soils (Wiggenhause *et al*. 2019). According to the authors, a limitation of this study was that source tracing with stable isotopes overestimated the proportion of cadmium in plants derived from the fertilizer because the isotope ratios of the sources were not sufficiently distinct from those of the soils. The fact that use of phosphate fertilizers can contribute to the uptake of larger quantities of cadmium in plants has been known for at least half a century (Schroeder and Balassa 1963).

To minimize soil pollution from fertilizers, many countries (e.g., in the EU) have threshold values for cadmium in fertilizers.

Cadmium uptake in plants has been associated with soil properties such as soil pH, per cent carbon, soil zinc (Smolders 2001), soil salinity (Li, Chaney and Schneiter 1994; McLaughlin *et al*. 1994), soil cadmium concentrations, precipitation during the growing period (Eriksson *et al*. 1996) and plant species (Hatch, Jones and Burau 1988). High cadmium levels have been reported in some crops, for example cocoa in the Latin America and Caribbean region (Meter, Atkinson and Laliberte 2019) and rice and vegetables in China (Chen *et al*. 2018). A study by John (1973) compared cadmium concentrations in edible plant parts. It reported that higher cadmium levels were found in leafy vegetables (i.e., lettuce and spinach leaves) compared with non-leafy plant parts (brassica tops, radish and carrot tubers, pea seeds and oat grains). Tobacco is among the non-food crops known to hyperaccumulate cadmium (Suman *et al*. 2018). Food is the main source of exposure to cadmium in general, but for smokers tobacco is another important source as tobacco plants (like other plants) take up cadmium from the soil (WHO 2019a). Absorption via the lungs is higher than via the gastrointestinal tract. Cadmium exposure from drinking water is relatively unimportant compared with exposure from diet (UNEP 2010).

The FAO and WHO Joint Expert Committee on Food Additives has established a provisional tolerable monthly intake (PTMI) for cadmium, based on meta-analysis of epidemiological studies on the relationship between urinary cadmium and beta-2-microglobulin (a marker of renal tubular effects; WHO 2019a). According to a report by the European Food Safety Authority, health-based guidance values for oral intake could be exceeded by children and adults at the 95th percentile of exposure (European Food Safety Authority [EFSA] 2012). It should be noted, however, that the tolerance level proposed by EFSA is lower than that proposed by the FAO/WHO Joint Expert Committee (2.5 vs. 5.8 µg/kg of body weight per week) (European Union [EU] 2020).

Adverse health effects of cadmium

Acute toxic effects

According to UNEP's Final Review of Scientific Information on Cadmium (UNEP 2010), acute cadmium poisoning in humans as a result of occupational exposure to airborne cadmium is characterized by irritation of the respiratory tract and lung damage.

Chronic toxic effects

Chronic toxic effects in humans include damage to the kidneys (proteinuria) and skeletal damage, with the former an early effects indicator. Lung cancer has also been associated with occupational inhalation of cadmium (UNEP 2010; Vacchi-Suzzi *et al*. 2016).

As noted in the UNEP report, cases of cadmium poisoning have been reported in farming households and by industry workers. In Japan after the Second World War, bone and kidney effects were linked to consumption of rice irrigated with water polluted with cadmium combined with a low-calcium diet. The major source of pollution was a mine located upstream (UNEP 2010). Among workers, concentrations of cadmium in blood and urine have been associated with ambient cadmium concentrations. Hence, for occupational effects the main route of exposure reported in workers is via inhalation (UNEP 2010; Vacchi-Suzzi *et al*. 2016). Chronic exposure matters more than acute (one-time) exposure, as cadmium has a long biological half-life of 15-20 years in humans and the effects manifest themselves mainly in older people (>50 years) (Smolders 2017). While dietary exposure to cadmium is recognized as a potential threat to human health (Box 9.3-1), no studies have assessed fertilizer use as a source of exposure, according to an epidemiological review carried out for this report (Box 9.3-2).

Effects in mammals are similar to those seen in humans. In addition, in animal experiments, cadmium has been reported to have reproductive effects such as interference with production of hormones and skeletal effects (e.g., disturbance of the accumulation of bone mass) (UNEP 2010; Vacchi-Suzzi *et al*. 2016). Cadmium mainly

Box 9.3-1 China: Dietary cadmium could be a threat to human health

Cadmium intake by the population is increasing in China. In a study on dietary exposure to cadmium by Song *et al.* (2017), cadmium uptake was estimated to be above the general provisional tolerable monthly intake established by the Joint FAO/WHO Expert Committee on Food Additives in at least 5 per cent of the study group. The estimated dietary cadmium intake for some subsistence rice farmers in southern China is comparable to that of people in the region of Japan where Itai-Itai disease (UNEP 2010; WHO 2019a) was first reported (Wang, P. *et al.* 2019).

Studies have reported that mining is the main pollution source for cadmium contamination of soils (e.g. Liu. X. *et al*. 2016), for example when rice is irrigated with water contaminated by mining and smelting (Wang, P. *et al*. 2019). In addition, increased application of fertilizer nitrogen was associated with an increase in cadmium concentration in the soil solution (Mitchell *et al.* 2020) and excessive use of nitrogen fertilizers has been associated with increased cadmium in food crops (Wang, P. *et al*. 2019). Soil pH has declined in the major Chinese crop production areas, which has been partially attributed to high rates of soil application of nitrogen fertilizers (Guo, J. *et al*. 2010). Lower soil pH can contribute to increased cadmium concentrations in plants (e.g. see Dai, H., Wei and Skuza 2020). High intake of dietary cadmium has also been associated with food choice. Rice has a comparatively high cadmium concentration compared to other cereals, and per capita consumption of rice is high in China as in some other Asian countries (Wang, P. *et al.* 2019).

According to the literature on cadmium reviewed in the FAO report on soil pollution (Rodríguez Eugenio, McLaughlin and Pennock 2018), measures that can reduce crop uptake of cadmium include addition of lime (to raise soil pH) and zinc (to reduce cadmium absorption and accumulation). In addition, the method of application and type of fertilizer used can influence uptake of cadmium by plants. For example, Fan *et al.* (2017) reported that split applications of ammonium or urea resulted in significantly lower cadmium concentrations in Chinese cabbage grown in soil contaminated by cadmium compared with single applications. However, no such difference was observed when nitrate was used as the source of nitrogen.

Box 9.3-2 General findings of two epidemiological reviews on fertilizers ("fertilizers and human health", "cadmium case study") carried out for this report.[1]

The first of two reviews carried out for this report was a systematic review of epidemiological studies linking fertilizer use and human health effects. Few studies were identified for final inclusion (47 for inorganic fertilizers, 17 for organic ones). The outcomes with the largest number of studies were cancers for inorganic fertilizers and infections for organic ones. The studies reported significant, not significant or inconclusive findings for the outcomes studied. Although this epidemiological review highlighted several limitations of the studies (e.g. study design, poor exposure assessment) it concluded that this is an emerging evidence base.

The second was a review and appraisal of published systematic reviews and/or meta-analyses on potential associations of cadmium exposure with human health-related outcomes. Among the assessed publications, the outcomes studied included kidney disease, cancers, endocrine disorders and cardiovascular disease. Significant, not significant or inconclusive findings were reported. Fertilizers were mentioned in some publications, but no study assessed fertilizer use as a source of cadmium exposure.

The findings of the two reviews are a clear indication of the existing data gap with regard to the health effects of fertilizers.

1 The full reviews are presented as an annex to this report. "Epidemiological Evidence Report: Exposure to pesticides and fertilizers and human health".

bioaccumulates in the kidneys and liver of vertebrates; it can also accumulate in aquatic invertebrates and algae.

Several studies (Cheng *et al*. 2019; Guo, J. *et al*. 2021) have demonstrated that there could be a link between cadmium and the spread of AMR. Relationships have been observed between tolerance to potentially toxic trace elements like cadmium and AMR among bacteria (Wales and Davies 2015; Singer *et al*. 2016; UNEP 2017; Cheng *et al*. 2019). Furthermore, the addition of cadmium to soil has been reported to increase the abundance and translocation of antibiotic resistance encoded genes in the soil and in lettuce (Guo, H. *et al*. 2021). Cheng *et al*. (2019) argue that although much remains unknown about direct transmission of AMR through the food chain to humans, the risk should not be neglected.

Uranium

An important use of uranium is as nuclear fuel to generate electricity in nuclear power stations. Uranium occurs naturally in mines (uranium ores), but is also found in industrial wastes and as a by-product of the copper and phosphoric acid industries, in areas affected by nuclear power plant accidents, and in phosphate rock (Kiegiel, Gajda and Zakrzewska-Kołtuniewicz 2017). Currently, obtaining uranium from rich uranium ores is considered cheaper than recovering it from secondary sources (Kiegiel, Gajda and Zakrzewska-Kołtuniewicz 2017). However, since uranium is a finite resource there may be a need in the distant future to recover it from secondary sources.

Phosphate rock contains some uranium, and some of this uranium remains in the final fertilizer product during mineral processing (Haneklaus *et al*. 2017). In the EU-28 it has been estimated that the amount of uranium extracted from imported phosphate rock could have been up to 334 tons in 2017 (Tulsidas *et al*. 2019). However, according to that study, recovery of uranium from phosphate fertilizers is constrained by low uranium prices. Liesch, Hinrichsen and Goldscheider (2015) reported that the range and spatial distribution of uranium and occasional peak values in drinking water appeared to be related to geogenic sources, and that agricultural land use generated a measurable but low background of uranium in groundwater. According to the literature, ingestion of uranium can affect human and animal health (Schnug and Lottermoser 2013).

Nitrates in drinking water

Sources of nitrates in drinking water

Nitrates in drinking water are from anthropogenic and geogenic sources. Inorganic and organic fertilizers are reported to be among the largest anthropogenic source of nitrates in groundwater (Shukla and Saxena 2018; Jin *et al*. 2019). Studies in India have identified increased nitrate concentrations in surface and groundwater with sewage percolation, and use of excessive amounts of inorganic fertilizers, among the sources of nitrates in drinking water (Gupta, Gupta and Gupta 2017). In the United States nitrogen inputs (inorganic fertilizer, manure) have been among the factors associated with increased violations of the allowable limit for nitrates in drinking water (Pennino, Compton and Leibowitz 2017).

The results of a bibliometric analysis carried out to provide insights into research activities and tendencies with regard to global drinking water from 1992 to 2011 (Fu, Wang and Ho 2013) show that arsenic, nitrates, fluoride, lead and cadmium were the most researched water contaminants. This study reported that in 1992 the number of articles about arsenic, nitrates, fluoride, lead and cadmium was below 50 for each contaminant, but by 2012 there were more than 250 articles about arsenic and nitrates, close to 200 about fluoride, and fewer than 200 about lead and cadmium. Although no such analysis was found for the period after 2012, nitrates in drinking water have continued to be an area of interest for researchers.

Evidence of high nitrate concentrations in drinking water

The guideline values for nitrates in drinking water for human consumption are set at 11 mg/litre for nitrate nitrogen and 50 mg/litre for nitrate ion in the WHO guidelines for drinking water quality (WHO 2016; WHO 2017a).

Nitrate contamination of groundwater is a global problem (Zhou 2015; Ward *et al*. 2018). However, global and national documentation on the extent and seriousness of this problem is scarce. The literature suggests that nitrates in drinking water could be affecting the health of a substantial proportion of the global population.

According to Mateo-Sagasta, Zadeh and Turral (2018), nitrates in groundwater have been reported to be a major problem in Europe, the United States, and South and East Asia. In Europe, nitrate drinking water limit values had been exceeded in around one-third of the groundwater bodies for which information was currently available (Mateo-Sagastra and Burke 2010). In India studies have reported a continuous increase in nitrate concentrations and nitrate concentrations above permissible levels in many states (Gupta, Gupta and Gupta 2017). Other examples are shown in Table 9.3-2.

Sources of nitrates in humans

Approximately 80 per cent of dietary intake of nitrates is from vegetables, with the remainder

Table 9.3-2 Evidence of high nitrate concentrations in water in selected countries.

	Study description	Conclusions	Reference
Australia	Quality of water in 1,054 wells	Data identified fertilizer as the main nitrate source in ≈50 per cent of wells with high nitrate concentrations	Thorburn *et al.* 2003)
China	Quality of water in monitored rivers	Standards for drinking or bathing not met in 70 per cent and 28 per cent of rivers in 2001 and 2015, respectively	Cassou, Jaffee and Ru (2017)
Switzerland	Quality of underground water	Concentration > the allowable limit in > 10 per cent of the sites and in ≈ 40 per cent of samples from a highly populated, intensively farmed plateau	Federal Office for the Environment of Switzerland (2019)
United States	Violations of the allowable limit of nitrate in drinking water	Among the factors positively related to violations were fertlizer purchases and nitrogen inputs (animal manure, inorganic fertilizer), soil organic matter and population density	Pennino, Compton and Leibowitz (2017)

coming from other foods and drinking water (Blekkenhorst *et al*. 2018). Nitrates are also produced in the body. This endogenous synthesis of nitrate complicates the risk assessment of nitrate (WHO 2003).

Adverse health effects of nitrates

Acute toxic effects

According to the literature cited by the World Health Organization (WHO 2003), fatalities have occurred following a single intake of 4-50 grams of nitrate (equivalent to 67-833 mg of nitrate per kilogram of body weight) and toxic doses have ranged from two to nine grams.

Chronic toxic effects

Exposure to non-lethal toxic doses has been associated with methemoglobinemia, also known as blue baby syndrome, in infants up to six months of age; it has also been associated with pregnancy loss, birth defects and infant mortality, cardiorespiratory issues (e.g., asthmatic attacks), gastrointestinal issues (e.g., diarrhoea in children), cancer, thyroid disruption and other health disorders (Gupta, Gupta and Gupta 2017).

Some health effects have been reported in populations. For example, according to literature cited by Gupta, Gupta and Gupta (2017), in India ingestion of water with high concentrations has been associated with diarrhoea, acute respiratory infection, early onset of hypertension, and increased blood pressure in school children.

An additional effect reported in animal experiments is kidney damage. Increasing kidney damage in rabbits was associated with intake of water with increasing concentrations of nitrates (Gupta *et al*. 2010).

There is, however, a lack of consensus among studies on the health effects of nitrates. A meta-analysis by Bahadoran *et al*. (2015) reported no significant association between nitrate exposure and the risk of thyroid cancer, hyper- and hypothyroidism, and a significant association between higher exposure to nitrite and the risk of thyroid cancer. In addition, Powlson *et al*. (2008) concluded that there was no consensus on links between nitrates and either blue baby syndrome or cancer of the digestive tract. Furthermore, a meta-analysis by Song, Wu and Guan (2015) associated high nitrate intake with a reduced risk of gastric cancer. Studies quantifying the contribution of fertilizers to nitrates in drinking water or groundwater are lacking. Some studies have suggested a possible relationship between fertilizer use and kidney disease, but other possible causes of that disease have been proposed (Box 9.3-3).

Box 9.3-3 Chronic kidney disease of unknown origin

Reports of chronic kidney disease of unknown origin (CKDu) are increasing globally, including in rural agricultural communities in Central America and other regions (Lunyeara *et al*. 2016; Singh *et al*. 2020).

A review of epidemiological studies from Central America concluded that, among other risk factors, there is a positive relationship between high (versus low) drinking water intake and CKDu (Gonzáles-Quiroz *et al.* 2018). In remote Australian Aboriginal communities where there is a high incidence of CKDu, levels of nitrates and uranium greatly exceed officially recommended levels. Most of these communities rely on raw groundwater to supply their domestic needs. They may be ingesting high levels of both nitrates and uranium, probably including uranyl nitrates (Rajapakse *et al.* 2019).

Possible causes of CKDu in different parts of the world continue to be examined. For example, a recent report based on the existing literature concluded that in Central America epidemiological patterns could not be explained by non-occupational risk factors, and that occupational heat stress was likely the main driver of the CKDu epidemic in that region (Wesseling *et al*. 2020). However, the review of epidemiological studies by Gonzáles-Quiroz *et al.* (2018) found no significant associations between heat stress and reported cases of CKDu in Central America. Their meta-analysis showed positive associations for males (versus females), family history of CKD and lowland altitude, as well as water intake. In addition to heat stress, there were no significant associations with pesticide exposure, non-steroidal anti-inflammatory drugs intake or alcohol consumption.

Urea

Very few incidents involving urea appear to be mentioned in the literature. The Toxicological Review of Urea by the United States Environmental Protection Agency (US EPA 2011) mentions a report by Steyn (1961) on accidental poisoning of 80 farm workers presumed to be exposed to a fertilizer containing 98 per cent urea. Symptoms reported were nausea, vomiting, excitement and convulsions accompanied by urination. According to the review, symptoms reported in other studies include decreased platelets adhesiveness in humans and reduced growth and negative effects on liver and kidney in animals.

Pathogens in organic inputs

Several studies have reported negative human effects of pathogens found in manure and sewage sludge/biosolids (Udeigwe *et al*. 2015). Livestock (e.g., cattle, pigs and chickens) can be carriers of pathogenic bacteria, protozoan parasites, helminthic worms and viruses (Goss, Tubeileh and Goorahoo 2013). A review by Delahoy *et al*. (2018) identified Salmonella, Lassa virus, Cryptosporidium and Toxoplasma gondii as important pathogens that could potentially be transmitted in animal faeces. Other pathogens which can potentially be transmitted through animal manure include Escherichia coli and Enterococci (Stocker *et al*. 2015). Use of raw sewage was associated with the presence of Giardia (a protozoa) and Ascaris (a helminth) in vegetables (Amahmid, Asmama and Bouhoum 1999). Helminths can be transmitted by eggs present in human faeces, which can contaminate the soil (WHO 2020a). The pathogens infect humans through oral intake, for example consumption of contaminated food and water or inhalation (Udeigwe *et al*. 2015) and dermal contact (Ercumen *et al*. 2017).

Livestock ownership

In a systematic review by Penakalapati *et al*. (2017) livestock ownership was associated with childhood diarrhoea, but was also protective against diarrhoea or was unrelated to it. The relationship between livestock ownership and child growth was also inconsistent with regard to reported positive, negative and no associations. Children who slept in the same quarters (or rooms) as animals were more likely to suffer from environmental enteric dysfunction than those sleeping away from animals.

Unrefined vs. refined organic fertilizers

The risk of contamination differs among organic fertilizers. Organic inputs processed before use will most likely have less pathogen contamination than those used before processing. Industrial organic inputs can present fewer risks, in relation to pathogens and contamination, than organic inputs produced and recycled within the farm. There are regulations targeting the quality of organic inputs. Chapters 8.4.2 and 8.5.2)

Pathogen populations in manure can be reduced by improving food and housing hygiene for livestock, storing manure at high temperatures, composting, avoiding the use of fresh manure, and applying manure early in the season to allow maximum time between application and harvesting (Hoagland *et al*. 2018). Some pathogens in manure can survive up to 120 days; persistence is longer in cold than in warm soil (Goss, Tubeileh and Goorahoo 2013).

In a study aimed at reducing pathogen survival by composting pig manure (McCarthy *et al*. (2011) the derived compost complied with EU regulations for processed manure products, as E. coli and Enterococcus were below limits and it was Salmonella-free. However, even when regulations are in place some products may be below standards. For example, in a study carried out in the United States by Brinton, Storms and Blewett (2009) some of the composts sold had pathogen concentrations above US EPA standards at the point of sale,

Endocrine disruptors

The International Programme on Chemical Safety (IPCS) has defined an endocrine disruptor as "an exogenous substance or mixture that alters function(s) of the endocrine system and consequently causes adverse health effects in an intact organism, or its progeny, or (sub) populations" (IPCS 2002; Bergman *et al*. 2013). The same report defines a potential endocrine

disruptor as "an exogenous substance or mixture that possesses properties that might be expected to lead to endocrine disruption in an intact organism, or its progeny, or (sub)populations".

Cadmium and nitrates (discussed earlier in this chapter) and perchlorate and steroid hormones are included among potential endocrine disrupters.

Potential endocrine disrupters have been reported in animal manure. Manure from pigs was found to contain endocrine disrupting compounds (Combalbert *et al*. 2012). These compounds can reach humans through contaminated water used for drinking, recreation or fishing (Goss, Tubeileh and Goorahoo 2013).

Perchlorate

According to a statement by the European Commission, perchlorate occurs naturally in the environment although other sources include nitrate and potash deposits and the manufacture, use and disposal of rocket propellants, explosives, fireworks, flares and air bag inflators (EC 2015). Water, soil and inorganic fertilizers were suspected to be relevant contributors of perchlorate found in fruits and vegetables in the EU, and Member States were requested to investigate sources of perchlorate in food.

Perchlorate is a competitive inhibitor of iodine uptake by the thyroid. It can interfere with thyroid function due to iodine shortage (Kabir, Rahman and Rahman 2015). Inhibition is reversible, as perchlorate does not bind to the thyroid and is rapidly excreted (FAO and WHO 2011). This concern is addressed in the 2019 EU Regulation on Fertilising Products, where the limit value for perchlorate in an inorganic macronutrient fertilizer was set at 50 mg/kg dry matter (EU 2019).

EU Member States have a harmonized enforcement approach to perchlorate, including harmonized guidance on perchlorate levels as a reference for trade within the EU aimed at protecting the health of consumers (EC 2015). In 2019 definitive limits were proposed and were favourably reviewed on the technical level by Member States in 2019 (EC 2019). In a 2020 amendment to Regulation (EC) No. 1881/2006, a section on maximum perchlorate levels in selected foodstuffs was added (EC 2020, p. L 160/5).

Steroid hormones

Groundwater is susceptible to contamination by steroid hormones from livestock facilities (Kolok et al. 2018). Application of manure to agricultural land is a route for oestrogens into terrestrial environment, and oestrogens have been detected in groundwater (Adeel *et al*. 2017). Steroid hormones have been detected in the roots of Chinese cabbage grown in soil treated with compost (Zhang, J. et al. 2019). Some studies have suggested a relationship between environmental oestrogens and breast cancer, but more data are required to confirm the presence or absence of such a relationship (Adeel *et al*. 2017). Overall, evidence on the contribution of fertilizers to endocrine disruptors and the impact of this contribution on human health is lacking.

Organic contaminants

Organic contaminants are persistent and include organochlorines (OCs), polychlorinated dibenzo-p-dioxin (PCDD), polychlorinated dibenzofurans (PCDF) and poly- and perfluoroalkyl substances (PFAS) (Rodríguez Eugenio, McLaughlin and Pennock 2018). These contaminants have been detected in sludge (Venkatesan and Halden 2013) and reported in soils with a history of sludge application (Weber *et al*. 2018). They can be taken up by plants, leading to dietary exposure (Brambilla *et al*. 2018). Their occurrence in soil is localized around industrial and urban areas and is not widespread (Rodríguez Eugenio, McLaughlin and Pennock 2018).

Evidence concerning the contribution of fertilizers to organic contaminants in soil and the effects of this contribution on human health is lacking.

Microplastics

Global plastic production continues to increase (PlasticsEurope 2019; Research and Markets 2021). At the same time, plastic pollution is growing rapidly (Meijer *et al*. 2021;

Silva *et al*. 2021). Microplastics are solid plastic particles, generally defined as 5 millimetres (mm) or less in size, which come from a variety of sources including the fragmentation and degradation of larger pieces of plastic litter (Anderson *et al*. 2016). Nanoplastics can be defined as solid plastic particles less than 1 micrometre (µm) in size which originate from engineered material or are produced unintentionally through the degradation and fragmentation of larger plastic objects, including microplastics (Jakubowicz, Enebro and Yarahmad 2021). Despite widespread concerns, little is yet known about the human health and environmental risks of microplastics and nanoplastics (WHO 2019b). Moreover, "what is known is surrounded by considerable uncertainty" (Science Advice for Policy by European Academies [SAPEA] 2019, p. 108).

"Primary microplastics" are produced intentionally (UNEP 2019). Very small plastic pellets used in manufacturing, and the microbeads added to cosmetics and personal care products (an application banned in some countries) are well known examples of primary microplastics. A European Chemicals Agency report (European Chemicals Agency [ECHA] 2019) describes the intentional use of microplastics in controlled-release fertilizers and fertilizer additives as follows: "Polymers in fertilizing products are primarily used to ensure the following functions: controlled release of nutrients over a period of up to 18 months through micro-encapsulation; anti-caking, prilling and other preservative functions; as fertilizer additives; reduced dust formation during application of fertilizers; [and] reduced run-off of fertilizers" (ECHA 2019, p. 104).

The ECHA report estimates that the total amount of microplastics intentionally used in the EU was estimated at 51,500 tons per year, of which the controlled-release fertilizers and fertilizer additives contributed about 29 per cent. Coated seeds and controlled plant protection products were found to have contributed around 25 per cent and household cleaning products and cosmetics about 31 per cent of total microplastics intentionally used in the European Economic Area per year (ECHA 2019).

Routes of human exposure to microplastics, whatever their sources, include ingestion, inhalation and dermal contact (Prata *et al*. 2020). Microplastics have been detected in foodstuffs, including eatable fruits and vegetables (Conti *et al*. 2020), as well as in human stool (Schwab *et al*. 2019) and in human placentas (Ragusa *et al*. 2021). Estimates by Cox *et al*. (2019) of annual dietary consumption of microplastics in the United States increased when inhalation was considered; individuals who drank bottled water ingested more microplastics than those who drank tap water. Microplastics have, however, been detected in both tap and bottled water, as well as in marine water, freshwater and wastewater. As emphasized by WHO (2019b), the quality and quantity of data vary across these water types.

According to the WHO report on microplastics in drinking water (WHO 2019b), absorption and distribution of very small microplastic particles, including nanoplastics, in the human body may be higher than in the case of larger particles. The WHO report adds that chemicals may arise from the production and degradation of microplastics, and that microplastics have the potential to accumulate hydrophobic persistent organic pollutants (POPs). A limited number of studies in freshwater indicate that microplastics could possibly enable long-distance transport of pathogens and increase the transfer of antimicrobial resistant genes between microorganisms.

The WHO report found that, despite substantial knowledge gaps, "the potential risks from microplastic-associated microorganisms are far lower than the well-established risk posed by the high concentrations and diversity of pathogens present in human and livestock wastes in drinking-water sources. In addition, in terms of providing surfaces for attachment and transport of microorganisms including pathogens, the concentrations of microplastics reported in drinking-water sources are far lower than concentrations of non-plastic particles that contribute to normal turbidity in water" (WHO 2019b, p. 48). However, it emphasized the need to better understand the sources and occurrence of microplastics in fresh water and

drinking water; the efficacy of different treatment processes and combinations of processes; and the significance of the potential return of microplastics to the environment from treatment waste streams, including the application of sludge biosolids to agricultural land (WHO 2019b).

Among the potential human health and environmental risks of microplastics being assessed by scientists, it has been suggested that they could contribute to the enrichment of antibiotic resistant bacteria (ARB) and the spread of AMR (WHO 2019b; Zhang, Y. *et al*. 2020; UNEP 2021).

Eutrophication

The extent and causes of eutrophication are discussed in Chapter 9.2.3.

Toxins

Toxins from harmful cyanotoxins have been reported in eutrophic waterbodies on all continents (Laybourn-Parry and Pearce 2007; Graham, Dubrovsky and Eberts 2017; Kubickova *et al*. 2019; Svirčev *et al*. 2019). Toxins produced by harmful cyanobacterial blooms can affect not only humans, but also fish and other animals. These toxins can enter the food chain through contaminated food sources. They have been found in fish muscles (Poste, Hecky and Guildford 2011). Humans may be exposed to them through oral, respiratory and dermal routes, as well as through haemodialysis if the dialysate is contaminated with cyanotoxins (Chorus and Welker 2021). They can affect the nervous system, lungs, liver, kidney, immune system and digestive system (Kubickova *et al*. 2019). Symptoms reported in humans include abdominal pain, vomiting, headache and diarrhoea (US EPA 2020h).

Adverse health effects

Acute toxic effects

Acute toxic effects resulting from eutrophication have been reported in humans. Some case studies provide evidence that exposure to cyanobacterial toxins in drinking water and recreational exposure to freshwater cyanobacteria can lead to illness (Chorus and Walker 2021).

At a haemodialysis clinic in Brazil, 116 (89 per cent) of 131 patients experienced visual disturbance, nausea, vomiting and muscle weakness following routine haemodialysis treatment; thereafter 100 patients developed acute liver failure (Azevedo *et al*. 2002). These effects were associated with the presence of cyanobacteria in the water supply reservoir, as cyanotoxins were found in patients' serum and liver tissue.

Chronic toxic effects

According to a literature review by Kubickova *et al*. (2019), chronic exposures to drinking water contaminated with cyanobacterial compounds, in some instances in combination with other factors, have been associated with increased incidence of liver cancer or chronic liver damage and increased incidence of colorectal or small intestinal cancer. However, according to Chorus and Welker (2021), it is impossible to show causation or to derive concentration-response data from the epidemiological studies available to date.

Nutrient dilution

Yield increases due to intensification have been related to the "dilution" of some nutrients (Cakmak and Kutman 2018). In wheat, nitrogen application reduced the essential amino acid index (EAAI) and protein digestibility-corrected amino acid score (PDCAAS) in grain (Zhang, P. *et al*. 2017). It is therefore possible that unbalanced fertilizers could affect the quality of food derived from plants. Balanced nutrition may reduce protein dilution as yield increases (Duncan *et al*. 2018).

9.3.3 Adverse effects of residential exposure

The section covers the adverse health effects of atmospheric $PM_{2.5}$ on human health.

For the adverse environmental effects of $PM_{2.5}$, see Chapter 9.2.1.

Epidemiology

Adverse health effects

For health effects this report relies on the recent Policy Assessment for the Review of the National Ambient Air Quality Standards for Particulate Matter by the United States Environmental Protection Agency (US EPA 2020b). That document reviews currently available evidence on health effects and exposure/risk information, among other topics. Some of its conclusions on health are presented below:

Acute and chronic toxic effects (short-term and long-term, respectively)

From agricultural activities ammonia is the main precursor for $PM_{2.5}$ (Bauer, Tsigaridis and Miller 2016; Wu, Y. et al. 2016). The Policy Assessment (US EPA 2020b) concludes that:

a causal relationship exists between long-term exposure to $PM_{2.5}$ and both mortality and cardiovascular effects.

there is likely to be a causal relationship between $PM_{2.5}$ and respiratory effects for both short and long-term exposure duration.

long-term exposure to $PM_{2.5}$ is likely to have a causal relationship with cancer and nervous system effects.

evidence suggests, but is not sufficient to infer, that a causal relationship is likely to exist between $PM_{2.5}$ and metabolic effects (e.g., diabetes, alterations in glucose and insulin homeostasis), reduced fertility, reduced chances of pregnancy, reduced foetal growth, low birth weights and preterm babies for both short- and long-term exposure.

9.3.4 Fertilizers and antimicrobial resistance

There is a risk that AMR will be transmitted from livestock systems to humans through direct contact or via the food chain (Huijbers 2016; WHO 2017b; Bandyopadhyay and Samanta 2020; Bennani *et al*. 2020).

AMR bacteria, which have been detected in manure, end up in water and soil (Huijbers *et al*. 2015). AMR in humans has been associated with the presence of manure (Muloi *et al*. 2019) and contact with livestock (Huijbers *et al*. 2014).

Since similar antimicrobials are provided to humans and animals (Tang *et al*. 2017), it is difficult to quantify the contributions of the livestock sector to AMR resistance in human beings. Still, the large amounts of antimicrobials used in livestock production could pose a large AMR risk if it is not already having a significant impact on human health. Interventions that restrict antibiotic use in food producing animals have been associated with a reduced antimicrobial resistance in animals, but studies on the effects of such interventions on AMR in humans remains scarce (Tang *et al*. 2017; Scott *et al*. 2018). However, it is likely that such interventions can result in reduced antimicrobial resistance in humans (Scott *et al*. 2018)

There are ongoing AMR surveillance programmes in Europe, but the results are difficult to compare due to differences in methodologies used in data collection (Schrijver *et al*. 2018).

For pesticides and AMR, see Chapter 4.4.7.

9.4 Links between the environmental and health effects of fertilizers

The environmental effects of fertilizer use can have negative effects on human health and nutrition. For example, degradation of the terrestrial and aquatic environments affects the sustainability of food production, and hence human health and security.

Changes in insect vector populations due to fertilizer use can affect the spread of communicable diseases. Townsend *et al*. (2003) hypothesized that increasing nutrient availability could often favour disease-causing organisms. They suggested that vector-borne diseases like malaria and West Nile virus may be affected by elevated nitrogen in aquatic environments. Although studies linking fertilizers to diseases transmitted by such vectors appear to be lacking, the fact that studies have demonstrated a positive relationship between fertilizer use and vectors indicates that fertilizer use can increase incidences of some vector-borne diseases in humans.

Studies have suggested that changes in human diet can affect both the environment and human health. For example, Sabaté and Soret (2014) proposed that increased adoption of plant-based diets would have positive effects on the food supply, as well as on the environment and human health. Increased intake of more plant-based foods would likely reduce the amounts of fertilizers used. For example, in a study by Marlow *et al*. (2009) non-vegetarian diets required 13 times more fertilizer than vegetarian ones.

9.5 Key data gaps: environmental and health effects of fertilizers

To a large extent, risk assessment involves the use of scientific principles and methods to conceptualize, assess and manage risks (Aven 2016). The knowledge generated can be used to guide decision-making, e.g., in determining policies.

While there are indications that fertilizers can have adverse effects on the environment and health (Chapters 9.1 and 9.2), some of the potential impacts of these effects may be unknown. The epidemiological reviews carried out for this report bear this out (Box 9.3-2). For example, according to the review of studies linking fertilizer use and health effects, studies were few; they reported either significant, not significant or inconclusive findings for the outcomes studied; and they had limitations (e.g., in regard to study design, poor exposure assessment).

Other reports have highlighted the lack of adequate data on adverse effects for use in decision-making. As pointed out in the FAO publication Soil Pollution: A Hidden Reality (Rodríguez Eugenio, McLaughlin and Pennock 2018), in the case of fertilizers, risk assessment is constrained by, for example, large gaps in available information on health effects as well as in toxicological data. According to plans for the implementation of the EU Fertilizer Regulation (EU 2019), the European Commission shall by 2026 submit to the European Parliament and to the Council a report which shall include "any new relevant scientific information as regards the toxicity and carcinogenicity of contaminants that becomes available, including the risks from uranium contamination in fertilizing products".

Estimating the risks posed by contaminants and pollutants from diffuse sources is difficult. Moreover, for many of those in the environment that could be from fertilizers, fertilizers are not the only source and the contribution of fertilizers to adverse environmental and health effects is difficult to estimate. This problem could partly explain why literature on risk assessment studies for fertilizers is scarce. According to a review of the types of risk in agriculture (Komarek, de Pinto and Smith 2020), a large share of studies has been devoted to production and markets and very few to health issues. Since farmers must cope with and manage different types of agricultural risks, with compounding effects, the authors suggest a shift in the focus of research towards analysing multiple contemporaneous types of risks, including those related to the environmental and health effects of overuse or inappropriate handling of inorganic fertilizer.

References

Abalos, D., Jeffery, S., Sanz-Cobena, A., Guardia, G. and Vallejo, A. (2014). Meta-analysis of the effect of urease and nitrification inhibitors on crop productivity and nitrogen use efficiency. *Agriculture Ecosystems and Environment* 189, 136-144. https://doi.org/10.1016/j.agee.2014.03.036.

Abraham, E., Cavanagh, J., Wood, P., Pearson, A. and Mladenov, P. (2016). Cadmium in New Zealand's agriculture and food systems. In *Integrated Nutrient and Water Management for Sustainable Farming*. *Occasional Report No. 29*. Currie, L.D. and Singh, R. (eds.). Palmerston North, New Zealand: Fertilizer and Lime Research Centre, Massey University. 1-13. http://flrc.massey.ac.nz/workshops/16/Manuscripts/Paper_Abraham_2016.pdf.

Access to Medicine Foundation (2020). *Antimicrobial Resistance Benchmark 2020*. https://accesstomedicinefoundation.org/publications/2020-antimicrobial-resistance-benchmark.

Adeel, M., Song, X., Wang, Y., Francis, D. and Yang, Y. (2017). Environmental impact of estrogens on human, animal and plant life: A critical review. *Environment International* 99, 107-119. https://doi.org/10.1016/j.envint.2016.12.010.

Albanito, F., Lebender, U., Cornulier, T., Sapkota, T.B., Brentrup, F., Stirling, C. and Hillier, J. (2017). Direct nitrous oxide emissions from tropical and sub-tropical agricultural systems – a review and modelling of emission factors. *Scientific Reports* 7, 44235. https://doi.org/10.1038/srep44235.

Ali, B.A., Ahmed, H.O., Ballal, S.G. and Albar, A.A. (2001). Pulmonary function of workers exposed to ammonia: A study in the eastern province of Saudi Arabia. *International Journal of Occupational and Environmental Health* 7(1), 19-22. https://www.researchgate.net/publication/12121352_Pulmonary_Function_of_Workers_Exposed_to_Ammonia_A_Study_in_the_Eastern_Province_of_Saudi_Arabia.

Amahmid, O., Asmama, S. and Bouhoum, K. (1999). The effect of waste water reuse in irrigation on the contamination level of food crops by *giardia* cysts and *ascaris* eggs. *International Journal of food Microbiology* 49(1-2), 19-26. https://doi.org/10.1016/S0168-1605(99)00058-6.

Amann, M. (ed.) (2017). *Measures to Address Air Pollution from Agricultural Sources*. Laxenburg: International Institute for Applied Systems Analysis. https://ec.europa.eu/environment/air/pdf/clean_air_outlook_agriculture_report.pdf.

American Geophysical Union (2020). Warming, acidic oceans may nearly eliminate coral reef habitats by 2021, 17 February. https://news.agu.org/press-release/warming-acidic-oceans-may-nearly-eliminate-coral-reef-habitats-by-2100/. Accessed 21 January 2021.

Anderson, J.C., Park, B.J., Palace, V.P. (2016). Microplastics in aquatic environments: Implications for Canadian ecosystems. *Environmental Pollution* 218. 269-280. https://www.sciencedirect.com/science/article/pii/S0269749116305620.

Aro, A., Alfthan, G. and Varo, P. (1995). Effects of supplementation of fertilizers on human selenium status in Finland. *The Analyst* 120, 841-843. https://doi.org/10.1039/AN9952000841.

Athar, R. and Ahmad, M. (2002). Heavy metal toxicity: Effect on plant growth and metal uptake by wheat, and on free living azotobacter. *Water, Air and Soil Pollution* 138, 165-180. https://www.researchgate.net/publication/225460294_Heavy_Metal_Toxicity_Effect_on_Plant_Growth_and_Metal_Uptake_by_Wheat_and_on_Free_Living_Azotobacter.

Austin, K.G., Harris, N.L., Wijaya, A., Murdiyarso, D., Harvey, T., Stolle, F. and Kasibhatla, P. (2018). A review of land-based greenhouse gas flux estimates in Indonesia. *Environmental Research Letters* 13, 055003. https://doi.org/10.1088/1748-9326/aab531.

Aven, T. (2016). Risk Assessment and Risk Management: Review of recent advances on their foundation. *European Journal of Operational Research* 253(1), 1-13. https://doi.org/10.1016/j.ejor.2015.12.023.

Azevedo, S.M.F.O., Carmichael, W.M., Jochimsen, E.M., Rinehart, K.L., Lau, S., Shaw, G.R. and Eaglesham, G.K. (2002). Human intoxication by microcystins during renal dialysis treatment in Caruaru—Brazil. *Toxicology* 181-182, 441-446. https://doi.org/10.1016/S0300-483X(02)00491-2.

Babrauskas, V. (2016). Explosions of ammonium nitrate fertilizer in storage or transportation are preventable accidents. *Journal of Hazardous Materials* 304, 134-149. https://doi.org/10.1016/j.jhazmat.2015.10.040.

Babrauskas, V. (2017). The West, Texas, ammonium nitrate explosion: A failure of regulation. *Journal of Fire Sciences* 35(5), 396-414. https://doi.org/10.1177/0734904116685723.

Bahadoran, Z., Mirmiran, P., Ghasemi, A., Kabir, A., Azizi, F. and Hadaegh, F. (2015). Is dietary nitrate/nitrite exposure a risk factor for development of thyroid abnormality? A systematic review and meta-analysis. *Nitric Oxide* 47, 65-76. https://doi.org/10.1016/j.niox.2015.04.002.

Ballal, S.G., Ali, B., Albar, A.A., Ahmed, H.O. and Al-Hasan, A.Y. (1998). Bronchial asthma in two chemical fertilizer producing factories in Eastern Saudi Arabia. *The International Journal of Tuberculosis and Lung Disease* 2(4), 330-335. https://www.researchgate.net/publication/13721824_Bronchial_asthma_in_two_chemical_fertilizer_producing_factories_in_Eastern_Saudi_Arabia.

Bandyopadhyay, S. and Samanta, I. (2020). Antimicrobial resistance in agri-food chain and companion animals as a re-emerging menace in post-covid epoch: Low-and middle-income countries perspective and mitigation strategies. *Frontiers in Veterinary Science* 7, 620. https://doi.org/10.3389/fvets.2020.00620.

Barbiero, R.P., Lesht, B.M., Hinchey, E.K. and Nettesheim, T.G. (2018). A brief history of the U.S. EPA Great Lakes National Program Office's water quality survey. *Journal of Great Lakes Research* 44(4), 539-546. https://doi.org/10.1016/j.jglr.2018.05.011.

Bauer, S.E., Tsigaridis, K. and Miller, R. (2016). Significant atmospheric aerosol pollution caused by world food cultivation. *Geophysical Research Letters* 43(10), 5394-5400. https://doi.org/10.1002/2016GL068354.

Bennani, H., Mateus, A., Mays, N., Eastmure, E., Stärk, K.D.C. and Häsler, B. (2020). Overview of evidence of antimicrobial use and antimicrobial resistance in the food chain. *Antibiotics* 9(2), 49. https://doi.org/10.3390/antibiotics9020049.

Bergman, Å., Heindel, J., Jobling, S., Kidd, K. and Zoeller, R.T. (eds.) (2013). *State of the Science on Endocrine Disrupting Chemicals-2012: An Assessment of the State of the Science of Endocrine Disruptors Prepared by a Group of Experts for the United Nations Environment Programme*. Geneva: United Nations Environment Programme and the World Health Organization. http://www.who.int/ceh/publications/endocrine/en/

Berthold, M., Karsten, U., von Weber, M., Bachor, A. and Schumann, R. (2018). Phytoplankton can bypass nutrient reductions in eutrophic coastal water bodies. *Ambio* 47, 146-158. https://doi.org/10.1007/s13280-017-0980-0.

Beusen, A.H.W., Bouwman, A.F., Van Beek, L.P.H., Mogollón, J.M. and Middelburg, J.J. (2016). Global riverine N and P transport to ocean increased during the 20th century despite increased retention along the aquatic continuum. *Biogeosciences* 13(8), 2441-2451. https://doi.org/10.5194/bg-13-2441-2016.

Bhat, M.R. and Ramaswamy, C. (1993). Effect of ammonia, urea and diammonium phosphate (DAP) on lung functions in fertilizer plant workers. *Indian Journal of Physiology and Pharmacology* 37(3), 221-224. https://pubmed.ncbi.nlm.nih.gov/8276500/.

Blekkenhorst, L.C., Bondonno, N.P., Liu, A.H., Ward, N.C., Prince, R.L., Lewis, J.R. *et al*. (2018). Nitrate, the oral microbiome, and cardiovascular health: A systematic literature review of human and animal studies. *The American Journal of Clinical Nutrition* 107(4), 504-522. https://doi.org/10.1093/ajcn/nqx046.

Bownik, A. (2016). Harmful Algae: Effects of cyanobacterial cyclic peptides on aquatic invertebrates – a short review. *Toxicon* 124, 26-35. https://doi.org/10.1016/j.toxicon.2016.10.017.

Boyd, C.E. (2018). Aquaculture Pond Fertilization. *CAB Reviews* 13, 002. https://doi.org/10.1079/PAVSNNR201813002.

Brambilla, G., D'Hollander, W., Oliaei, F., Stahl, T. and Weber, R. (2018). Pathways and factors for food safety and food security at PFOS contaminated sites within a problem based learning approach. *Chemosphere* 129, 192-202. https://doi.org/10.1016/j.chemosphere.2014.09.050.

Bray, N. and Wickings, K. (2019). The roles of invertebrates in the urban soil microbiome. *Frontiers in Ecology and Evolution* 7, 359. https://doi.org/10.3389/fevo.2019.00359.

Breitburg, D., Gregoire, M. and Isensee, K. (2018). *The Ocean is Losing its Breath: Declining Oxygen in the World's Ocean and Coastal Waters – Summary for Policy Makers*. Paris: Intergovernmental Oceanographic Commission of the United Nations Educational, Scientific and Cultural Organization (IOC-UNESCO). https://unesdoc.unesco.org/ark:/48223/pf0000265196.

Breitburg, D., Levin, L.A., Oschilies, A., Grégoire, M., Chavez, F.P., Conley, D.J. *et al*. (2018). Declining oxygen in the global ocean and coastal waters. *Science* 359 (6371), eaam7240. https://doi.org/10.1126/science.aam7240.

Brinton, W.F., Storms, P. and Blewett, T.C. (2009). Occurrence and levels of fecal indicators and pathogenic bacteria in market-ready recycled organic matter composts. *Journal of Food Protection* 72(2), 332-339. https://doi.org/10.4315/0362-028X-72.2.332.

Burke, L., Reytar, K., Spalding, M. and Perry, A. (2011). *Reefs at Risk Revisited*. Washington D.C: World Resources Institute. https://www.wri.org/publication/reefs-risk-revisited.

Cakmak, I. and Kutman, U.B. (2018). Agronomic biofortification of cereals with zinc: A review. *European Journal of Soil Science* 69(1), 172-180. https://doi.org/10.1111/ejss.12437.

Carlson, R.R., Foo, S.A. and Asner, G.P. (2019). Land use impacts on coral reef health: A ridge-to-reef perspective. *Frontiers in Marine Science* 6, 562. https://doi.org/10.3389/fmars.2019.00562.

Cassou, E., Jaffee, S.M. and Ru, J. (2017). *The Challenge of Agricultural Pollution - Evidence from China, Vietnam and the Philippines*. Washington, D.C: World Bank. https://openknowledge.worldbank.org/bitstream/handle/10986/29187/9781464812019.pdf?sequence=2&isAllowed=y.

Chamberlain, P.M., McNamara, N.P., Chaplow, J., Stott, A.W. and Black, H.I. (2006). Translocation of surface litter carbon into soil by collembola. *Soil Biology and Biochemistry* 38(9), 2655-2664. https://doi.org/10.1016/j.soilbio.2006.03.021.

Chaney, R.L. (2012). Chapter Two - Food safety issues for mineral and organic fertilizers. *Advances in Agronomy* 117, 51-116. https://doi.org/10.1016/B978-0-12-394278-4.00002-7.

Charles, A., Rochette, P., Whalen, J.K., Angers, D.A., Chantigny, M.H. and Bertrand, N. (2017). Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: A meta-analysis. *Agriculture, Ecosystems and Environment* 236, 88-98. https://doi.org/10.1016/j.agee.2016.11.021.

Chen, D., Lan, Z., Hu, S. and Bai, Y. (2015). Effects of nitrogen enrichment on belowground communities in grassland: Relative role of soil nitrogen availability vs. soil acidification. *Soil Biology and Biochemistry* 89, 99-108. https://doi.org/10.1016/j.soilbio.2015.06.028.

Chen, H., Yang, X., Wang, P., Wang, Z., Li, M. and Zhao, F.J. (2018). Dietary cadmium intake from rice and vegetables and potential health risk: A case study in Xiangtan, Southern China. *Science of The Total Environment* 639, 271-277. https://doi.org/10.1016/j.scitotenv.2018.05.050.

Cheng, G., Ning, J., Ahmed, S., Huang, J., Ullah, R., An, B. *et al*. (2019). Selection and dissemination of antimicrobial resistance in agri-food production. *Antimicrobial Resistance and Infection Control* 8, 158. https://doi.org/10.1186/s13756-019-0623-2.

Chernysh,Y., Yakhnenko, O., Chubur, V. and Roubík, H. (2021). Phosphogypsum recycling: A review of environmental issues, current trends, and prospects. *Applied Sciences* 11(4), 1575. https://doi.org/10.3390/app11041575.

Chesapeake Bay Foundation (2019). The history of Chesapeake Bay cleanup efforts. https://www.cbf.org/how-we-save-the-bay/chesapeake-clean-water-blueprint/the-history-of-bay-cleanup-efforts.html. Accessed 21 March 2021.

Chesapeake Bay Foundation (2021). 2020 state of the blueprint. https://www.cbf.org/how-we-save-the-bay/chesapeake-clean-water-blueprint/2020-state-of-the-blueprint/. Accessed 21 March 2021.

Chesapeake Bay Program (2019). Agriculture. https://www.chesapeakebay.net/issues/agriculture. Accessed 21 March 2021.

Chorus, I. and Welker, M. (eds.) (2021). *Toxic Cyanobacteria in Water - A Guide to Their Public Health Consequences, Monitoring and Management*. 2nd edition. https://www.taylorfrancis.com/books/oa-edit/10.1201/9781003081449/toxic-cyanobacteria-water-ingrid-chorus-martin-welker.

Chu, D. (2018). Effects of heavy metals on soil microbial community. *IOP Conference Series: Earth and Environmental Science* 113, 012009. https://iopscience.iop.org/article/10.1088/1755-1315/113/1/012009.

Chuong, J., Huxley, J., Sportswood, E.N., Nichols, L., Mariotte, P. and Suding, K.N. (2016). Cattle as dispersal vectors of invasive and introduced plants in a California annual grassland. *Rangeland Ecology & Management* 69(1), 52-58. https://doi.org/10.1016/j.rama.2015.10.009.

Cole, L., Buckland, S.M. and Bardgett, R.D. (2005). Relating microarthropod community structure and diversity to soil fertility manipulations in temperate grassland. *Soil Biology and Biochemistry* 37(9), 1707-1717. https://doi.org/10.1016/j.soilbio.2005.02.005.

Combalbert, S., Bellet, V., Dabert, P., Bernet, N., Balaguer, P. and Hernandez-Raquet, G. (2012). Fate of steroid hormones and endocrine activities in swine manure disposal and treatment facilities. *Water Research* 46(3), 895-906. https://doi.org/10.1016/j.watres.2011.11.074.

Conti, G.O., Ferrante, M., Banni, M., Favara, C., Nicolosi, I., Cristaldi, A. *et al*. (2020). Micro- and nano-plastics in edible fruit and vegetables. The First Diet Risks Assessment for the General Population. *Environmental Research* 187, 1109677. https://doi.org/10.1016/j.envres.2020.109677.

Cox, K.D., Covernton, G.A., Davies, H.L., Dower, J.F., Juanes, F. and Dudas, S.E. (2019). Human consumption of microplastics. *Environmental Science & Technology* 53(12), 7068-7074. https://doi.org/10.1021/acs.est.9b01517.

Cronin, S.J., Manoharan, V., Hedley, M.J. and Loganathan, P. (2000). Fluoride: A review of its fate, bioavailability, and risks of fluorosis in grazed-pasture systems in New Zealand. *New Zealand Journal of Agricultural Research* 43(3), 295-321. https://doi.org/10.1080/00288233.2000.9513430.

Cycoń, M., Mrozik, A. and Piotrowska-Seget, Z. (2019). Antibiotics in the soil environment – degradation and their impact on microbial activity and diversity. *Frontiers in Microbiology* 10, 338. https://doi.org/10.3389/fmicb.2019.00338.

Dai, H., Wei, S. and Skuza, L. (2020). Effects of different soil pH and nitrogen fertilizers on *Bidens pilosa* L. Cd accumulation. *Environmental Science and Pollution Research* 27, 9403-9409. https://doi.org/10.1007/s11356-019-07579-5.

Dai, Z., Su, W., Chen, H., Barberán, A., Zhao, H., Yu, M. *et al*. (2018). Long-term nitrogen fertilization decreases bacterial diversity and favors the growth of actinobacteria and proteobacteria in agro-ecosystems across the globe. *Global Change Biology* 24(8), 3452-3461. https://pubmed.ncbi.nlm.nih.gov/29645398/.

de Graaff, M.-A., Hornslein, N., Throop, H.L., Kardol, P. and van Diepen, L.T.A. (2019). Chapter one - effects of agricultural intensification on soil biodiversity and implications for ecosystem functioning: A meta-analysis. *Advances in Agronomy* 155, 1-44. https://doi.org/10.1016/bs.agron.2019.01.001.

De Paz, J.M., Delgado, J.A., Ramos, C., Shaffer, M.J. and Barbarick, K.K. (2009). Use of a new GIS nitrogen index assessment tool for evaluation of nitrate leaching across a Mediterranean region. *Journal of Hydrology* 365(3-4), 183-194. https://doi.org/10.1016/j.jhydrol.2008.11.022.

de Souza Machado, A.A., Lau, C.W., Kloas, W., Bergmann, J., Bachelier, J.B., Faltin, E. *et al*. (2019). Microplastics can change soil properties and affect plant performance. *Environmental Science & Technology* 53(10), 6044-6052. https://doi.org/10.1021/acs.est.9b01339.

Delahoy, M.J., Wodnik, B., McAliley, L., Penakalapati, G., Swarthout, J., Freeman, M.C. and Levy, K. (2018). Pathogens transmitted in animal feces in low- and middle-income countries. *International Journal of Hygiene and Environmental Health* 221(4), 661-676. https://doi.org/10.1016/j.ijheh.2018.03.005.

Delgado, J.A. (2021). Nitrogen (nutrient) trading tool. In *Managing Water Resources and Hydrological Systems*. *2nd edition*. Fath, B.D. and Jorgensen, S.E. (eds.). Boca Raton, FL: CRC Press. 407-14. https://www.routledge.com/Managing-Water-Resources-and-Hydrological-Systems/Fath-Jorgensen/p/book/9781138342668.

Department of Agriculture, Water and Environment of Australia (2020). Managing and protecting the Great Barrier Reef. https://www.environment.gov.au/marine/gbr/protecting-the-reef. Accessed 20 December 2020.

Dhaliwal, S.S., Naresh, R.K., Gupta, R.K., Panwar, A.S., Mahajan, N.C., Singh, R. and Mandal, A. (2019). Effect of tillage and straw return on carbon footprints, soil organic carbon fractions and soil microbial community in different textured soils under rice-wheat rotation: A review. *Reviews in Environmental Science and Bio/Technology* 19, 103-115. https://doi.org/10.1007/s11157-019-09520-1.

Díaz, R., Rabalais, N.N. and Breitburg, D.L. (2012). *Agriculture's Impact on Aquaculture: Hypoxia and Eutrophication in Marine Waters*. Paris: Organisation for Economic Co-operation and Development. https://www.oecd.org/greengrowth/sustainable-agriculture/49841630.pdf.

Duncan, E.G., O'Sullivan, C.A., Roper, M.M., Biggs, J.S. and Peoples, M.B. (2018). Influence of co-application of nitrogen with phosphorus, potassium and sulphur on the apparent efficiency of nitrogen fertiliser use, grain yield and protein content of wheat: Review. *Field Crops Research* 226, 56-65. https://doi.org/10.1016/j.fcr.2018.07.010.

Edwards, C.A. and Lofty, J.R. (1982). Nitrogenous fertilizers and earthworm populations in agricultural soils. *Soil Biology and Biochemistry* 14(5), 515-521. https://doi.org/10.1016/0038-0717(82)90112-2.

Egertson, C.J. and Downing, J.A. (2004). Relationship of fish catch and composition to water quality in a suite of agriculturally eutrophic lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 61(9), 1784-1796. https://doi.org/10.1139/f04-109.

Ercumen, A., Pickering, A.J., Kwong, L.H., Arnold, B.F., Parvez, S.M., Alam, M. *et al*. (2017). Animal feces contribute to domestic fecal contamination: Evidence from *e. coli* measured in water, hands, food, flies and soil in Bangladesh. *Environmental Science & Technology* 51(15), 8725-8734. https://doi.org/10.1021/acs.est.7b01710.

Eriksson, J., Öborn, I., Jansson, G. and Andersson, A. (1996). Factors influencing Cd-content in crops - results from Swedish field investigations. *Swedish Journal of Agricultural Research* 26(3), 125-133. https://www.researchgate.net/publication/259825147_Factors_Influencing_Cd-Content_in_Crops_Results_from_Swedish_field_investigations.

European Chemicals Agency (2019). *Annex XV Restriction Report - Proposal for a Restriction - Intentionally Added Microplastics*. *Version No. 1.2.* https://echa.europa.eu/documents/10162/05bd96e3-b969-0a7c-c6d0-441182893720.

European Commission (2015). *Statement as Regards the Presence of Perchlorate in Food Endorsed by the Standing Committee on Plants, Animals, Food and Feed on 10 March 2015, updated on 23 June 2015*. https://ec.europa.eu/food/system/files/2016-10/cs_contaminants_catalogue_perchlorate_statement_food_update_en.pdf.

European Commission (2019). *Summary Report. Standing Committee on Plants, Animals, Food and Feed Section Novel Food and Toxicological Safety of the Food Chain (Subsection of Phytopharmaceuticals-Legislation Committee Meeting)*. https://ec.europa.eu/food/system/files/2019-12/reg-com_toxic_20191022_sum.pdf.

European Commission (2020). *Commission Regulation (EU) 2020/685 of 20 May 2020 Amending Regulation (EC) No 1881/2006 as Regards Maximum Levels of Perchlorate in Certain Foods (Text with EEA Relevance)*. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32020R0685&from=EN.

European Commission (n.d.). *Chemical Accident Prevention and Preparedness: Major Accidents Involving Fertilizers*. Lessons Learned Bulletin No. 5. https://minerva.jrc.ec.europa.eu/en/shorturl/minerva/mahb_bulletin_no5_final_forthewebpdf.

European Environment Agency (2017). *Air Quality in Europe – 2017 Report*. https://www.eea.europa.eu/publications/air-quality-in-europe-2017.

European Environment Agency (2018). *European Union Emission Inventory Report 1990-2016 under the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP)*. https://op.europa.eu/en/publication-detail/-/publication/39bfecbd-7ffe-11e8-ac6a-01aa75ed71a1.

European Food Safety Authority (2012). Cadmium dietary exposure in the European population. *EFSA Journal* 10(1), 2551. https://doi.org/10.2903/j.efsa.2012.2551.

European Union (2019). *Regulation (EU) 2019/1009 of the European Parliament and of the Council of 5 June 2019 Laying down Rules on the Making Available on the Market of EU Fertilising Products and Amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and Repealing Regulation (EC) No 2003/2003*. https://eur-lex.europa.eu/eli/reg/2019/1009/oj.

European Union (2020). *European Union Comments on Codex Circular Letter CL 2019/81 OCS/CF: Request for Comments at Step 6 on the Draft Maximum Level for Cadmium in Chocolates Containing or Declaring <30% Total Cocoa Solids on a Dry Matter Basis*. CODEX Committee Contaminants in Food. 14th Session, 20-24 April 2020. https://www.fao.org/fao-who-codexalimentarius/sh-proxy/en/?lnk=1&url=https%253A%252F%252Fworkspace.fao.org%252Fsites%252Fcodex%252FCircular%252520Letters%252FCL%2525202019-81%252Fcl19_81e.pdf.

Fan, S.-K., Zhu, J., Tian, W.-H., Guan, M.-Y., Fang, X.-Z. and Jin, C.-W. (2017). Effects of split applications of nitrogen fertilizers on the cd level and nutritional quality of Chinese cabbage. *Journal of Zhejiang University-Science B* 18, 897-905. https://link.springer.com/article/10.1631/jzus.B1600272.

Farokhi, A., Heederik, D. and Smit, L.A.M. (2018). Respiratory health effects of exposure to low levels of airborne endotoxin – a systematic review. *Environmental Health* 17, 14. https://doi.org/10.1186/s12940-018-0360-7.

Federal Office for the Environment of Switzerland (2019). *État et Évolution des Eaux Souterraines en Suisse - Résultats de l'Observation Nationale des Eaux Souterraines NAQUA, État 2016. Rencontre d'information pour les médias, 15 août 2019, Berne*. [Also available in German, as *Zustand und Entwicklung Grundwasser Schweiz*, but not published in English.] https://www.bafu.admin.ch/bafu/fr/home/themes/eaux/publications/publications-eaux/resultats-observatoire-eaux-souterraines-naqua.html.

Food and Agriculture Organization of the United Nations (2002). *World Agriculture: Towards 2015/2030 - An FAO Perspective*. http://www.fao.org/3/y4252e/y4252e00.htm.

Food and Agriculture Organization of the United Nations (2015). *Status of the World's Soil Resources: Main Report*. http://www.fao.org/documents/card/en/c/c6814873-efc3-41db-b7d3-2081a10ede50/.

Food and Agriculture Organization of the United Nations (2020). *The Share of Agriculture in Total Greenhouse Gas Emissions: Global, Regional and Country Trends 1990-2017*. http://www.fao.org/3/ca8389en/CA8389EN.pdf.

Food and Agriculture Organization of the United Nations and World Health Organization (2011). *Safety Evaluation of Certain Contaminants in Food: Prepared by the Seventy-second Meeting of the Joint FAO/WHO Expert Committee on Food Additives (JEFCA)*. https://apps.who.int/iris/handle/10665/44520.

Food and Agriculture Organization of the United Nations, Intergovernmental Technical Panel on Soil, Global Soil Biodiversity Initiative, Secretariat of the Convention on Biological Diversity and European Commission (2020). *State of Knowledge of Soil Biodiversity – Status, Challenges and Potentialities - Report 2020*. https://doi.org/10.4060/cb1928en.

Fox, J.E., Starcevic, M., Kow, K.Y., Burow, M.E. and McLachlan, J.A. (2001). Endocrine disrupters and flavonoid signalling. *Nature* 413, 128-129. https://doi.org/10.1038/35093163.

Frková, Z., Vystavna, Y., Koubová, A., Kotas, P., Grabicová, K., Grabic, R. *et al*. (2020). Microbial responses to selected pharmaceuticals in agricultural soils: Microcosm study on the roles of soil, treatment and time. *Soil Biology and Biochemistry* 149, 107924. https://doi.org/10.1016/j.soilbio.2020.107924.

Fu, H.-Z., Wang, M.-H. and Ho, Y.-S. (2013). Mapping of drinking water research: A bibliometric analysis of research output during 1992-2011. *Science of The Total Environment* 443, 757-765. https://doi.org/10.1016/j.scitotenv.2012.11.061.

Galic, N., Hawkins, T. and Forbes, V.E. (2019). Adverse impacts of hypoxia on aquatic invertebrates: A meta-analysis. *Science of The Total Environment* 652, 736-743. https://doi.org/10.1016/j.scitotenv.2018.10.225.

Gall, J.E., Boyd, R.S. and Rajakaruna, N. (2015). Transfer of heavy metals through terrestrial food webs: A review. *Environmental Monitoring and Assessment* 187, 201. https://doi.org/10.1007/s10661-015-4436-3.

Garriga, N., Montori, A. and Llorente, G.A. (2017). Impact of ammonium nitrate and sodium nitrate on tadpoles of *Alytes obstetricans*. *Ecotoxicology* 26, 667-674. https://doi.org/10.1007/s10646-017-1799-y.

Gauns, M., Mochemadkar, S., Pratihary, A., Shirodkar, G., Narvekar, P.V. and Naqvi, S.W.A. (2020). Phytoplankton associated with seasonal oxygen depletion in waters of the western continental shelf of India. *Journal of Marine Systems* 204, 103338. https://doi.org/10.1016/j.jmarsys.2020.103308.

Ger, K.A., Urrutia-Cordero, P., Frost, P.C., Hansson, L.-A., Sarnelle, O., Wilson, A.E. and Lürling, M. (2016). The interaction between cyanobacteria and zooplankton in a more eutrophic world. *Harmful Algae* 54, 128-144. https://doi.org/10.1016/j.hal.2015.12.005.

Gibbens, S. (2020). Scientists Are Trying to Save Coral Reefs. Here's what's working, 4 June. *National Geographic*. https://www.nationalgeographic.com/science/2020/06/scientists-work-to-save-coral-reefs-climate-change-marine-parks/. Accessed 20 December 2020.

Gionfra, S. (2018). *Plastic Pollution in Soil*. Brussels: Institute for European Environmental Policy. https://ieep.eu/uploads/articles/attachments/3a12ecc3-7d09-4e41-b67c-b8350b5ae619/Plastic%20pollution%20in%20soil.pdf?v=63695425214.

Gonzáles-Quiroz, M., Pearce, N., Caplin, B. and Nitsch, D. (2018). What do epidemiological studies tell us about chronic kidney disease of undetermined cause in Meso-America? A systematic review and meta-analysis. *Clinical Kidney Journal* 11(4), 496-506. https://doi.org/10.1093/ckj/sfx136.

Goss, M.J., Tubeileh, A. and Goorahoo, D. (2013). A review of the use of organic amendments and the risk to human health. In *Advances in Agronomy*. Sparks, D.L. (ed.). Academic Press. Chapter 5. 275-380. https://www.elsevier.com/books/advances-in-agronomy/sparks/978-0-12-407686-0.

Governments of Canada and Ontario (2018). *Canada-Ontario Lake Erie Action Plan: Partnering on Achieving Phosphorous Loading Reductions to Lake Erie from Canadian Sources*. https://www.canada.ca/content/dam/eccc/documents/pdf/great-lakes-protection/dap/action_plan.pdf.

Graham, J.L., Dubrovsky, N.M. and Eberts, S.M. (2017). *Cyanobacterial Harmful Algal Blooms and U.S. Geological Survey Science Capabilities*. Reston, Virginia: United States Geological Survey. https://doi.org/10.3133/ofr20161174.

Guglielmi, G. (2020). Why Beirut's ammonium nitrate blast was so devastating, 10 August. *Nature*. https://www.nature.com/articles/d41586-020-02361-x. Accessed 28 December 2020.

Guo, H., Xue, S., Nasir, M., Gu, J. and Lv, J. (2021). Impacts of cadmium addition on the alteration of microbial community and transport of antibiotic resistance genes in oxytetracycline contaminated soil. *Journal of Environmental Sciences* 99, 51-58. https://www.sciencedirect.com/science/article/pii/S100107422030156X.

Guo, J.H., Liu, X.J., Zhang, Y., Shen, J.L., Han, W.X., Zhang, W.F. *et al*. (2010). Significant acidification in major chinese croplands. *Science* 327(5968), 1008-1010. https://doi.org/10.1126/science.1182570.

Guo, J.-J., Huang, X.-P., Xiang, L., Wang, Y.-Z., Li, Y.-W., Li, H. *et al*. (2020). Source, migration and toxicology of microplastics in soil. *Environment International* 137, 105263. https://doi.org/10.1016/j.envint.2019.105263.

Gupta, S., Gupta, R.C., Gupta, A.B., Eskiocak, S., Prakasa Rao, E.V.S., Puttanna, K. and Singhvi, A. (2010). Pathophysiology of nitrate toxicity in human and its mitigation measures. *ING Bulletins on Regional Assessment of Reactive Nitrogen* 20 (Raghuram, N., ed.), i-v and 1-78. https://www.researchgate.net/publication/282847297_Pathophysiology_of_Nitrate_Toxicity_in_Human_and_its_Mitigation_Measures.

Gupta, S.K., Gupta, A.B. and Gupta, R. (2017). 28 - Pathophysiology of nitrate toxicity in humans in view of the changing trends of the global nitrogen cycle with special reference to India. In *The Indian Nitrogen Assessment: Sources of Reactive Nitrogen, Environmental and Climate Effects, Management Options, and Policies*. Abrol, Y.P., Adhya, T.K., Aneja, V.P., Raghuram, N., Pathak, H., Kulshrestha, U. *et al*. (eds.) Elsevier. Chapter 28. 459-468. https://www.sciencedirect.com/book/9780128118368/the-indian-nitrogen-assessment#book-description.

Gyenes, Z. and Wood, M.H. (2015). Lessons learned from major accidents involving fertilizers. *Loss Prevention Bulletin* 242, 15-18. https://minerva.jrc.ec.europa.eu/en/shorturl/minerva/lpb242pg15_fertiliserspdf.

Hamilton, H.A., Ivanova, D., Stadler, K., Merciai, S., Schmidt, J., van Zelm, R. *et al*. (2018). Trade and the role of non-food commodities for global eutrophication. *Nature Sustainability* 1, 314-321. https://doi.org/10.1038/s41893-018-0079-z.

Hamscher, G., Pawelzick, H.T., Sczesny, S., Nau, H. and Hartung, J. (2003). Antibiotics in dust originating from a pig-fattening farm: A new source of health hazard for farmers? *Environmental Health Perspectives* 111 (13), 1590-1594. https://doi.org/10.1289/ehp.6288.

Han, Z., Walter, M.T. and Drinkwater, L.E. (2017). N2O emissions from grain cropping systems: A meta-analysis of the impacts of fertilizer-based and ecologically-based nutrient management strategies. *Nutrient Cycling in Agroecosystems* 107, 335-355. https://doi.org/10.1007/s10705-017-9836-z.

Haneklaus, N., Sun, Y., Bol, R., Lottermoser, B. and Schnug, E. (2017). To extract, or not to extract uranium from phosphate rock, that is the question. *Environmental Science & Technology* 51, 753-754. https://doi.org/10.1021/acs.est.6b05506.

Hatch, D.J., Jones, L.H.P. and Burau, R.G. (1988). The effect of pH on the uptake of cadmium by four plant species grown in flowing solution culture. *Plant and Soil* 105(1), 121-126. https://doi.org/10.1007/BF02371149.

He, D., Luo, Y., Lu, S., Liu, M., Song, Y. and Lei, L. (2018). Microplastics in soils: Analytical methods, pollution characteristics and ecological risks. *TrAC Trends in Analytical Chemistry* 109, 163-172. https://doi.org/10.1016/j.trac.2018.10.006.

Helmberger, M.S., Tiemann, L.K. and Grieshop, M.J. (2019). Towards an ecology of soil microplastics. *Functional Ecology* 34(3), 550-560. https://doi.org/10.1111/1365-2435.13495.

Hergoualc'h, K., Akiyama, H., Bernoux, M., Chirinda, N., del Prado, A., Kasimir, Å. *et al*. (2019). N2O emissions from managed soils, and CO2 emissions from lime and urea application. In *2019 Refinement to the 2006 IPCC Guidelines on National Greenhouse Gas Inventories*. *Volume 4: Agriculture, Forestry and Other Land Use*. Switzerland: Intergovernmental Panel on Climate Change. Chapter 11. 11.1-11.48. https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Ch11_Soils_N2O_CO2.pdf.

Hernández, A.J. and Pastor, J. (2008). Relationship between plant biodiversity and heavy metal bioavailability in grasslands overlying an abandoned mine. *Environmental Geochemistry and Health* 30, 127-133. https://doi.org/10.1007/s10653-008-9150-4.

Hoagland, L., Ximenes, E., Ku, S. and Ladisch, M. (2018). Foodborne pathogens in horticultural production systems: Ecology and mitigation. *Scientia Horticulturae* 236, 192-206. https://doi.org/10.1016/j.scienta.2018.03.040.

Hoegh-Guldberg, O., Poloczanska, E.S., Skirving, W. and Dove, S. (2017). Coral reef ecosystems under climate change and ocean acidification. *Frontiers in Marine Science* 4, 158. https://doi.org/10.3389/fmars.2017.00158.

Huijbers, P.M.C. (2016). *Transmission of Antibiotic Resistance from Animals to Humans: Broilers as a Reservoir of ESBL-Producing Bacteria*. PhD thesis. Wageningen: Wageningen University. https://research.wur.nl/en/publications/transmission-of-antibiotic-resistance-from-animals-to-humans-broi.

Huijbers, P.M.C., Blaak, H., de Jong, M.C.M., Graat, E.A.M., Vandenbroucke-Grauls, C.M.J.E. and de Roda Husman, A.M. (2015). Role of the environment in the transmission of antimicrobial resistance to humans: A review. *Environmental Science & Technology* 49(20), 11993-12004. https://doi.org/10.1021/acs.est.5b02566.

Huijbers, P.M.C., Graat, E.A.M., Haenen, A.P.J., van Santen, M.G., van Essen-Zandbergen, A., Mevius, D.J. *et al*. (2014). Extended-spectrum and AmpC β-lactamase-producing *Escherichia coli* in broilers and people living and/or working on broiler farms: Prevalence, risk factors and molecular characteristics. *Journal of Antimicrobial Chemotherapy* 69(10), 2669-2675. https://doi.org/10.1093/jac/dku178.

Humbert, J.-Y., Dwyer, J.M., Andrey, A. and Arlettaz, R. (2016). Impacts of nitrogen addition on plant biodiversity in mountain grasslands depend on dose, application duration and climate: A systematic review. *Global Change Biology* 22(1), 110-120. https://doi.org/10.1111/gcb.12986.

Interagency Coordination Group on Antimicrobial Resistance (2019). *No Time to Wait: Securing the Future from Drug-Resistant Infections*. *Report to the Secretary-General of the United Nations April 2019*. https://www.who.int/docs/default-source/documents/no-time-to-wait-securing-the-future-from-drug-resistant-infections-en.pdf?sfvrsn=5b424d7_6.

Intergovernmental Panel on Climate Change (2018). Summary for policymakers. In *Global Warming of 1.5°C. An IPCC Special Report on the Impacts of Global Warming of 1.5°C above Pre-industrial Levels and Related Global Greenhouse Gas Emission Pathways, in the Context of Strengthening the Global Response to the Threat of Climate Change, Sustainable Development, and Efforts to Eradicate Poverty*. Masson-Delmotte, V., Zhai, P., Pörtner, H.-O., Roberts, D., Skea, J., Shukla, P.R., *et al*. (eds.). https://www.ipcc.ch/site/assets/uploads/sites/2/2019/05/SR15_SPM_version_report_LR.pdf.

International Joint Commission (2014). *A Balanced Diet for Lake Erie: Reducing Phosphorus Loadings and Harmful Algal Blooms – A Report of the Lake Erie Ecosystem Priority*. https://legacyfiles.ijc.org/publications/2014%20IJC%20LEEP%20REPORT.pdf.

International Programme on Chemical Safety (2002). *Global Assessment on the State-of-the-Science of Endocrine Disrupters*. Damstra, T., Barlow, S., Bergman, A., Kavlock, R. and Van Der Kraak, G. (eds.) https://apps.who.int/iris/handle/10665/67357.

Jakubowicz, I., Enebro, J. and Yarahmadi, N. (2021). Challenges in the search for nanoplastics in the environment – a critical review from the polymer science perspective. *Polymer Testing* 93, 106953. https://doi.org/10.1016/j.polymertesting.2020.106953.

Jantke, K., Hartmann, M.J., Rasche, L., Blanz, B. and Schneider, U.A. (2020). Agricultural greenhouse gas emissions: Knowledge and positions of German farmers. *Land* 9(5), 130. https://doi.org/10.3390/land9050130.

Jeppesen, R., Rodriguez, M., Rinde, J., Haskins, J., Hughes, B., Mehner, L. and Wasson, K. (2016). Effects of hypoxia on fish survival and oyster growth in a highly eutrophic estuary. *Estuaries and Coasts* 41, 89-98 https://doi.org/10.1007/s12237-016-0169-y.

Jin, Z., Cen, J., Hu, Y., Li, L., Shi, Y., Fu, G. and Li, F. (2019). Quantifying nitrate sources in a large reservoir for drinking water by using stable isotopes and a Bayesian isotope mixing model. *Environmental Science and Pollution Research* 26, 20364-20376. https://doi.org/10.1007/s11356-019-05296-7.

John, M.K. (1973). Cadmium uptake by eight food crops as influenced by various soil levels of cadmium. *Environmental Pollution* 4(1), 7-15. https://doi.org/10.1016/0013-9327(73)90026-8.

Johnson, P.T.J., Townsend, A.R., Cleveland, C.C., Glibert, P.M., Howarth, R.W., McKenzie, V.J. *et al*. (2010). Linking environmental nutrient enrichment and disease emergence in humans and wildlife. *Ecological Applications* 20(1), 16-29. https://doi.org/10.1890/08-0633.1.

Jubb, T.F., Annand, T.E., Main, D.C. and Murphy, G.M. (1993). Phosphorus supplements and fluorosis in cattle – a northern Australia experience. *Australian Veterinary Journal* 70(10), 379-383. https://doi.org/10.1111/j.1751-0813.1993.tb00818.x.

Kabir, E.R., Rahman, M.S. and Rahman, I. (2015). A review on endocrine disruptors and their possible impacts on human health. *Environmental Toxicology and Pharmacology* 40(1), 241-258. https://doi.org/10.1016/j.etap.2015.06.009.

Khan, A., Khan, S., Khan, M.A., Qamar, Z. and Waqas, M. (2015). The uptake and bioaccumulation of heavy metals by food plants, their effects on plants nutrients, and associated health risk: A review. *Environmental Science and Pollution Research* 22, 13772-13799. https://doi.org/10.1007/s11356-015-4881-0.

Khan, M.A., Khan, S., Khan, A. and Alam, M. (2017). Soil contamination with cadmium, consequences and remediation using organic amendments. *Science of The Total Environment* 601-602, 1591-1605. https://doi.org/10.1016/j.scitotenv.2017.06.030.

Khan, M.N., Mobin, M., Abbas, Z.K. and Alamri, S.A. (2018). Fertilizers and their Contaminants in Soils, Surface and Groundwater. In *The Encyclopedia of the Anthropocene Volume 5: Contaminants*. Elsevier. Dellasala, D.A. and Goldstein, M. (eds). 225-240. https://www.sciencedirect.com/referencework/9780128135761/encyclopedia-of-the-anthropocene#book-description.

Kidd, J., Manning, P., Simkin, J., Peacock, S. and Stockdale, E. (2017). Impacts of 120 years of fertilizer addition on a temperate grassland ecosystem. *PLoS ONE* 12(3), e0174632. https://www.ncbi.nlm.nih.gov/pmc/articles/PMC5369769/.

Kieć-Swierczyńska, M., Krecisz, B. and Swierczyńska-Machura, D. (2003). Most frequent causes of allergic contact dermatitis in farmers: Based on material collected in the Nofer Institute of Occupational Medicine, Łódź [Article in Polish]. *Medycyna Pracy* 54(3), 237-243. https://pubmed.ncbi.nlm.nih.gov/14669576/.

Kiegiel, K., Gajda, D. and Zakrzewska-Kołtuniewicz, G. (2017). Secondary raw materials as a potential source of uranium. In *proceeding of nuclear 2017. The 10th Annual International Conference on Sustainable Development through Nuclear Research and Education. Part 1/3 Nuclear Energy*. Chapter I.3. 132-137. https://inis.iaea.org/collection/NCLCollectionStore/_Public/49/086/49086185.pdf.

Kim, D., Kwak, J.I. and An, Y.-J. (2019). Physiological response of crop plants to the endocrine-disrupting chemical nonylphenol in the soil environment. *Environmental Pollution* 251, 573-580. https://doi.org/10.1016/j.envpol.2019.04.101.

Kolok, A.S., Ali, J.M., Rogan, E.G. and Bartelt-Hunt, S.L. (2018). The fate of synthetic and endogenous hormones used in the us beef and dairy industries and the potential for human exposure. *Current Environmental Health Reports* 5, 225-232. https://doi.org/10.1007/s40572-018-0197-9.

Komarek, A.M., De Pinto, A. and Smith, V.H. (2020). A review of types of risks in agriculture: What we know and what we need to know. *Agricultural Systems* 178, 102738. https://doi.org/10.1016/j.agsy.2019.102738.

Kroon, F.J., Thorburn, P., Schaffelke, B. and Whitten, S. (2016). Towards protecting the Great Barrier Reef from land-based pollution. *Global Change Biology* 22(6), 1985-2002. https://doi.org/10.1111/gcb.13262.

Kubickova, B., Babica, P., Hilscherová, K. and Šindlerová, L. (2019). Effects of cyanobacterial toxins on the human gastrointestinal tract and the mucosal innate immune system. *Environmental Sciences Europe* 31(31). https://doi.org/10.1186/s12302-019-0212-2.

Kumar, K., Gupta, S.C., Baidoo, S.K., Chander, Y. and Rosen, C.J. (2005). Antibiotic uptake by plants from soil fertilized with animal manure. *Journal of Environmental Quality* 34(6), 2082-2085. https://doi.org/10.2134/jeq2005.0026.

Kuzikova, I.L., Zaytseva, T.B., Kichko, A.A., Zinoveva, S.V., Russu, A.D., Mayachkina, N.V. and Medvedeva, N.G. (2019). Effect of nonylphenols on the abundance and taxonomic structure of the soil microbial community. *Eurasian Soil Science* 52, 671-681. https://doi.org/10.1134/S1064229319060073.

Laboureur, D.M., Han, Z., Harding, B.Z., Pineda, A., Pittman, W.C., Rosas, C. *et al*. (2016). Case study and lessons learned from the ammonium nitrate explosion at the west fertilizer facility. *Journal of Hazardous Materials* 308, 164-172. https://doi.org/10.1016/j.jhazmat.2016.01.039.

Laden, F., Schwartz, J., Speizer, F.E. and Dockery, D.W. (2006). Reduction in fine particulate air pollution and mortality: Extended follow-up of the Harvard six cities study. *American Journal of Respiratory and Critical Care Medicine* 173(6), 667-672. https://doi.org/10.1164/rccm.200503-443OC.

Lam, S., Nguyen-Viet, H., Tuyet-Hanh, T.T., Nguyen-Mai, H. and Harper, S. (2015). Evidence for public health risks of wastewater and excreta management practices in Southeast Asia: A scoping review. *International Journal of Environmental Research and Public Health* 12(10), 12863-12885. https://doi.org/10.3390/ijerph121012863.

Larney, F.J. and Blackshaw, R.E. (2003). Weed seed viability in composted beef cattle feedlot manure. *Journal of Environmental Quality* 32(3), 1105-1113. https://doi.org/10.2134/jeq2003.1105.

Laybourn-Parry J. and Pearce, D.A. (2007). The biodiversity and ecology of Antarctic lakes: Models for evolution. *Philosophical Transactions of the Royal Society B* 362, 2273-2289. https://doi.org/10.1098/rstb.2006.1945.

Le Moal, M., Gascuel-Odoux, C., Ménesguen, A., Souchon, Y., Étrillard, C., Levain, A. *et al*. (2019). Eutrophication: A new wine in an old bottle?. *Science of The Total Environment* 651, 1-11. https://geosciences.univ-tours.fr/images/media/20180926103300-lemoal_et_al_2018_stoten.pdf.

Lemanski, K. and Scheu, S. (2015). The influence of fertilizer addition, cutting frequency and herbicide application on soil organisms in grassland. *Biology and Fertility of Soils* 51, 197-205. https://doi.org/10.1007/s00374-014-0963-2.

Li, X., Zhang, Q., Zhang, Y., Zheng, B., Wang, K., Chen, Y. *et al*. (2015). Source contributions of urban PM2.5 in the Beijing-Tianjin-Hebei region: Changes between 2006 and 2013 and relative impacts of emissions and meteorology. *Atmospheric Environment* 123(Part A), 229-239. https://doi.org/10.1016/j.atmosenv.2015.10.048.

Li, X.G., Jia, B., Lv, J., Ma, Q., Kuzyakov, Y. and Li, F.-M. (2017). Nitrogen fertilization decreases the decomposition of soil organic matter and plant residues in planted soils. *Soil Biology and Biochemistry* 112, 47-55. https://doi.org/10.1016/j.soilbio.2017.04.018.

Li, Y.-M., Chaney, R.L. and Schneiter, A.A. (1994). Effect of soil chloride level on cadmium concentration in sunflower kernels. *Plant and Soil* 167, 275-280. https://doi.org/10.1007/BF00007954.

Liesch, T., Hinrichsen, S. and Goldscheider, N. (2015). Uranium in groundwater – fertilizers versus geogenic sources. *The Science of The Total Environment 536*, 981-995. https://doi.org/10.1016/j.scitotenv.2015.05.133.

Liu, P., Jia, S., He, X., Zhang, X. and Ye, L. (2017). Different impacts of manure and chemical fertilizers on bacterial community structure and antibiotic resistance genes in arable soils. *Chemosphere* 188, 455-464. https://doi.org/10.1016/j.chemosphere.2017.08.162.

Liu, T., Chen, X., Hu, F., Ran, W., Shen, Q., Li, H. and Whalen, J.K. (2016). Carbon-rich organic fertilizers to increase soil biodiversity: Evidence from a meta-analysis of nematode communities. *Agriculture, Ecosystems and Environment* 232, 199-207. https://joann-whalen.research.mcgill.ca/publications/Agriculture%20Ecosystems%20and%20Environment%20232--199%E2%80%93207.pdf.

Liu, T., Guo, R., Ran, W., Whalen, J.K. and Li, H. (2015). Body size is a sensitive trait-based indicator of soil nematode community response to fertilization in rice and wheat agroecosystems. *Soil Biology and Biochemistry* 88, 275-281. https://doi.org/10.1016/j.soilbio.2015.05.027.

Liu, X., Tian, G., Jiang, D., Zhang, C. and Kong, L. (2016). Cadmium (Cd) Distribution and contamination in chinese paddy soils on national scale. *Environmental Science and Pollution Research* 23, 17941-17952. https://doi.org/10.1007/s11356-016-6968-7.

Lugon-Moulin, N., Ryan, L., Donini, P. and Rossi, L. (2006). Cadmium content of phosphate fertilizers used for tobacco production. *Agronomy for Sustainable Development* 26(3), 151-155. https://doi.org/10.1051/agro:2006010.

Lunyera, J., Mohottige, D., Von Isenburg, M., Jeuland, M, Patel, U.D. and Stanifer, J.W. (2016). CKD of uncertain etiology: A systematic review. *Clinical Journal of the American Society of Nephrology* 11(3), 379-385. https://doi.org/10.2215/CJN.07500715.

Lwanga, E.H., Gertsen, H., Gooren, H., Peters, P., Salánki, T., van der Ploeg *et al*. (2016). Microplastics in the terrestrial ecosystem: Implications for *Lumbricus Terrestris* (Oligochaeta, Lumbricidae). *Environmental Science & Technology* 50(5), 2685-2691. https://doi.org/10.1021/acs.est.5b05478.

Maaroufi, N.I and De Long, J.R. (2020). Global change impacts on forest soils: Linkage between soil biota and carbon-nitrogen-phosphorus stoichiometry. *Frontiers in Forests and Global Change* 20, 16. https://doi.org/10.3389/ffgc.2020.00016.

Mann, R.M., Hyne, R.V., Choung, C.B. and Wilson, S.P. (2009). Amphibians and agricultural chemicals: Review of the risks in a complex environment. *Environmental Pollution* 157(11), 2903-2927. https://doi.org/10.1016/j.envpol.2009.05.015.

Mao, Z., Zheng, X.-F., Zhang, Y.-Q., Tao, X.-X., Li, Y. and Wang, W. (2012). Occurrence and biodegradation of nonylphenol in the environment. *International Journal of Molecular Science* 13(1), 491-505. https://doi.org/10.3390/ijms13010491.

Marlow, H.J., Hayes, W.K., Soret, S., Carter, R.L., Schwab, E.R. and Sabaté, J. (2009). Diet and the environment: Does what you eat matter? *The American Journal of Clinical Nutrition* 89(5), 1699S-1703S. https://doi.org/10.3945/ajcn.2009.26736Z.

Mateo-Sagastra, J. and Burke, J. (2010). *Agriculture and Water Quality Interactions: A Global Overview*. *SOLAW Background Thematic Report -TR08*. Rome: Food and Agriculture Organization of the United Nations. http://www.fao.org/3/a-bl092e.pdf.

Mateo-Sagasta, J., Zadeh, S.M. and Turral, H. (2018). *More People, More Food, Worse Water? A Global Review of Water Pollution from Agriculture*. Rome and Colombo: Food and Agriculture Organization of the United Nations and International Water Management Institute on behalf of the Water, Land and Ecosystems Research Program of the CGIAR. http://www.fao.org/3/CA0146EN/ca0146en.pdf.

Matthiessen, P., Wheeler, J.R. and Weltje, L. (2018). A review of the evidence for endocrine disrupting effects of current-use chemicals on wildlife populations. *Critical Reviews in Toxicology* 48(3), 195-216. https://doi.org/10.1080/10408444.2017.1397099.

McCarthy, G., Lawlor, P.G., Coffey, L., Nolan, T., Gutierrez, M. and Gardiner, G.E. (2011). An assessment of pathogen removal during composting of the separated solid fraction of pig manure. *Bioresource Technology* 102(19), 9059-9067. https://doi.org/10.1016/j.biortech.2011.07.021.

McLaughlin, M.J., Palmer, L.T., Tiller, K.G., Beech, T.A. and Smart, M.K. (1994). Increased soil salinity causes elevated cadmium concentrations in field-grown potato tubers. *Journal of Environmental Quality* 23(5), 1013-1018. https://doi.org/10.2134/jeq1994.00472425002300050023x.

McLaughlin, M.J., Parker, D.R. and Clarke, J.M. (1999). Metals and micronutrients – food safety issues. *Field Crops Research* 60(1-2), 143-163. https://doi.org/10.1016/S0378-4290(98)00137-3.

McLaughlin, M.J., Stevens, D.P., Keerthisinghe, D.G., Cayley, J.W.D. and Ridley, A.M. (2001). Contamination of soil with fluoride by long-term application of superphosphates to pastures and risk to grazing animals. *Australian Journal of Soil Research* 39(3), 627-640. https://doi.org/10.1071/SR00036.

McLaughlin, M.J., Tiller, K.G., Naidu, R. and Stevens, D.P. (1996). Review: The behavior and environmental impact of contaminants in fertilizers. *Australian Journal of Soil Research* 34(1), 1-54. https://doi.org/10.1071/SR9960001.

Meijer, L.J.J., van Emmerik, T., van der Ent, R., Schmidt, C. and Lebreton, L. (2021). More than 1000 rivers account for 80% of global riverine plastic emissions into the ocean. *Science Advances* 7(18), 5803. https://doi.org/10.1126/sciadv.aaz5803.

Meter, A., Atkinson, R.J. and Laliberte, B. (2019). *Cadmium in Cacao from Latin America and the Caribbean - A Review of Research and Potential Mitigation Solutions*. Rome: Biodiversity International. https://cgspace.cgiar.org/bitstream/handle/10568/102353/Bioversity-CAF-Review-Cad-Cacao-13Nov2019-EN.pdf?sequence=9&isAllowed=y.

Midolo, G., Alkemade, R., Schipper, A.M., Benítez-López, A., Perring, M.P. and De Vries, W. (2019). Impacts of nitrogen addition on plant species richness and abundance: A global meta-analysis. *Global Ecology and Biogeography* 28(3), 398-413. https://doi.org/10.1111/geb.12856.

Minor, M.A. and Norton, R.A. (2004). Effects of soil amendments on assemblages of soil mites (Acari: Oribatida, Mesostigmata) in short-rotation willow plantings in central New York. *Canadian Journal of Forest Research* 34(7), 1417-1425. https://doi.org/10.1139/x04-035.

Mirlean, N. and Roisenberg, A. (2006). The effect of emissions of fertilizer production on the environment contamination by cadmium and arsenic in Southern Brazil. *Environmental Pollution* 143(2), 335-340. https://doi.org/10.1016/j.envpol.2005.11.022.

Mishra, J., Singh, R. and Arora, N.K. (2017). Alleviation of heavy metal stress in plants and remediation of soil by rhizosphere microorganisms. *Frontiers in Microbiology* 8, 1706. https://doi.org/10.3389/fmicb.2017.01706.

Mitchell, L.G., Grant, C.A. and Racz, G.J. (2000). Effect of nitrogen application on concentration of cadmium and nutrient ions in soil solution and in durum wheat. *Canadian Journal of Soil Science* 80(1), 107-115. https://cdnsciencepub.com/doi/10.4141/S98-085.

Miyata, K. and Ose, K. (2012). Thyroid hormone-disrupting effects and the amphibian metamorphosis assay. *Journal of Toxicologic Pathology* 25(1), 1-9. https://doi.org/10.1293/tox.25.1.

Mortvedt, J.J. (1996). Heavy Metal Contaminants in Inorganic and Organic Fertilizers. In *Fertilizers and Environment*. Rodriguez-Barrueco, C. (ed.). Springer Nature. 5-11. https://link.springer.com/chapter/10.1007/978-94-009-1586-2_2.

Mullen, R.A., Hurst, J.J., Naas, K.M., Sassoubre, L.M. and Aga, D.S. (2019). Assessing uptake of antimicrobials by *Zea mays* L. and prevalence of antimicrobial resistance genes in manure-fertilized soil. *Science of The Total Environment* 646, 409-415. https://doi.org/10.1016/j.scitotenv.2018.07.199.

Muloi, D., Kiiru, J., Ward, M.J., Hassell, J.M., Bettridge, J.M., Robinson, T.P. *et al*. (2019). Epidemiology of antimicrobial resistant *Escherichia coli* carriage in sympatric humans and livestock in a rapidly urbanizing city. *International Journal of Antimicrobial Agents* 54(5), 531-537. https://doi.org/10.1016/j.ijantimicag.2019.08.014.

Muturi, E.J., Ramirez, J.L., Rooney, A.P. and Dunlap, C. (2016). Association between fertilizer-mediated changes in microbial communities and *Aedes albopictus* growth and survival. *Acta Tropica* 164, 54-63. https://doi.org/10.1016/j.actatropica.2016.08.018.

Nan, Z., Zhao, C., Li, J., Chen, F. and Sun, W. (2002). Relations between soil properties and selected heavy metal concentrations in spring wheat (*Triticum aestivum L.*) grown in contaminated soils. *Water, Air and Soil Pollution* 133, 205-213. https://doi.org/10.1023/A:1012962604095.

Novo, M., Verdú, I., Trigo, D. and Martínez-Guitarte, J.L. (2018). Endocrine disruptors in soil: effects of bisphenol a on gene expression of the earthworm *Eisenia Fetida*. *Ecotoxicology and Environmental Safety* 150, 159-167. https://doi.org/10.1016/j.ecoenv.2017.12.030.

Organisation for Economic Co-operation and Development (2013). *OECD Compedium of Agri-environmental Indicators*. https://www.oecd.org/environment/oecd-compendium-of-agri-environmental-indicators-9789264186217-en.htm.

Penakalapati, G., Swarthout, J., Delahoy, M.J., McAliley, L., Wodnik, B., Levy, K. and Freeman, M.C. (2017). Exposure to animal feces and human health: A systematic review and proposed research priorities. *Environmental Science & Technology* 51(20), 11537-11552. https://doi.org/10.1021/acs.est.7b02811.

Peng, S., Dolfing, J., Feng, Y., Wang, Y. and Lin, X. (2018). Enrichment of the antibiotic resistance gene *tet*(l) in an alkaline soil fertilized with plant derived organic manure. *Frontiers in Microbiology* 9, 1140. https://doi.org/10.3389/fmicb.2018.01140.

Pennino, M.J., Compton, J.E. and Leibowitz, S.G. (2017). Trends in drinking water nitrates violations across the United States. *Environmental Science & Technology* 51(22), 13450-13460. https://doi.org/10.1021/acs.est.7b04269.

Pérez-Valera, E., Kyselková, M., Ahmed, E., Sladecek, F.X.J., Goberna, M. and Elhottová, D. (2019). Native soil microorganisms hinder the soil enrichment with antibiotic resistance genes following manure applications. *Scientific Reports* 9, 6760. https://doi.org/10.1038/s41598-019-42734-5.

Perkins, M.W., Wong, B., Tressler, J., Rodriguez, A., Sherman, K., Andres, J. *et al*. (2017). Adverse respiratory effects in rats following inhalation exposure to ammonia: Respiratory dynamics and histopathology. *Inhalation Toxicology* 29(1), 32-41. https://doi.org/10.1080/08958378.2016.1277571.

Pino-Otín, M.R., Muñiz, S., Val, J. and Navarro, E. (2017). Effects of 18 pharmaceuticals on the physiological diversity of edaphic microorganisms. *Science of The Total Environment* 595, 441-450. https://doi.org/10.1016/j.scitotenv.2017.04.002.

Pittman, W., Han, Z., Harding, B., Rosas, C., Jiang, J., Pineda, A. and Mannan, S. (2014). Lessons to be learned from an analysis of ammonium nitrate disasters in the last 100 Years. *Journal of Hazardous Materials* 280, 472-477. https://doi.org/10.1016/j.jhazmat.2014.08.037.

Pizolotto, C.A., Maciel, J.L.N., Fernandes, J.M.C. and Boller, W. (2019). Saprotrophic survival of *Magnaporthe oryzae* in infested wheat residues. *European Journal of Plant Pathology* 153, 327-339. https://doi.org/10.1007/s10658-018-1578-5.

Pizzol, M., Smart, J.C.R. and Thomsen, M. (2014). External costs of cadmium emissions to soil: A drawback of phosphorus fertilizers. *Journal of Cleaner Production* 84, 475-483. https://doi.org/10.1016/j.jclepro.2013.12.080.

PlasticsEurope (2019). *Plastics – the Facts 2019*. https://plasticseurope.org/wp-content/uploads/2021/10/2019-Plastics-the-facts.pdf.

Pope, C.A., III, Ezzati, M. and Dockery, D.W. (2009). Fine-particulate air pollution and life expectancy in the United States. *New England Journal of Medicine* 360(4), 376-386. https://doi.org/10.1056/NEJMsa0805646.

Poste, A.E., Hecky, R.E. and Guildford, S.J. (2011). Evaluating microcystin exposure risk through fish consumption. *Environmental Science & Technology* 45(13), 5806-5811. https://doi.org/10.1021/es200285c.

Powlson, D.S., Addiscott, T.M., Benjamin, N., Cassman, K.G., de Kok, T.M., van Grinsven, H. *et al*. (2008). When does nitrate become a risk for humans? *Journal of Environmental Quality* 37(2), 291-295. https://doi.org/10.2134/jeq2007.0177.

Prata, J.C., da Costa, J.P. Lopes, I., Duarte, A.C. and Rocha-Santos, T. (2020). Environmental exposure to microplastics: An overview on possible human health effects. *Science of The Total Environment* 702, 13445. https://doi.org/10.1016/j.scitotenv.2019.134455.

Rabalais, N.N., Cai, W.-J., Carstensen, J., Conley, D.J., Fry, B., Hu, X. *et al*. (2014). Eutrophication-driven deoxygenation in the coastal ocean. *Oceanography* 27(1), 172–183. https://doi.org/10.5670/oceanog.2014.21.

Ragusa, A., Svelato, A., Santacroce, C., Catalano, P., Notarstefano, V., Carnevali, O. *et al.* (2021). Plasticenta: First evidence of microplastics in human placenta. *Environment International* 146, 106274. https://www.sciencedirect.com/science/article/pii/S0160412020322297.

Rahimi, S., Mashhadi, H.R., Banadaky, M.D. and Mesgaran, M.B. (2016). Variation in weed seed fate fed to different holstein cattle groups. *PLoS ONE* 11(4), e0154057. https://doi.org/10.1371/journal.pone.0154057.

Rajapakse, J., Rainer-Smith, S., Millar, G.J., Grace, P., Hutton, A., Hoy, W. *et al*. (2019). Unsafe drinking water quality in remote Western Australian Aboriginal communities. *Geographical Research* 57(2), 178-188. https://doi.org/10.1111/1745-5871.12308.

Reiskind, M.H. and Wilson, M.L. (2004). *Culex Restuans* (Diptera: Culicidae) Oviposition behavior determined by larval habitat quality and quantity in Southeastern Michigan. *Journal of Medical Entomology* 41(2), 179-186. https://doi.org/10.1603/0022-2585-41.2.179.

Research and Markets (2021). Global plastic market report 2021-2028: market size is expected to surpass $750 billion, 28 April. https://www.globenewswire.com/en/news-release/2021/04/28/2218533/28124/en/Global-Plastic-Market-Report-2021-2028-Market-Size-is-Expected-to-Surpass-750-Billion.html. Accessed 30 April 2021.

Ripple, W.J., Wolf, C., Newsome, T.M., Barnard, P. and Moomaw, W.R. (2019). World scientists' warning of a climate emergency. *BioScience* 70(1), 8-12. https://doi.org/10.1093/biosci/biz088.

Roberts, T.L. (2014). Cadmium and phosphorous fertilizers: The issues and the science. *Procedia Engineering* 83, 52-59. https://doi.org/10.1016/j.proeng.2014.09.012.

Rodríguez Eugenio, N., McLaughlin, M. and Pennock, D. (2018). *Soil Pollution: A Hidden Reality*. Rome: Food and Agriculture Organization of the United Nations. http://www.fao.org/3/I9183EN/i9183en.pdf.

Rotz A. (2020). Environmental sustainability of livestock production. *Meat and Muscle Biology* 4(2), 1-18. https://doi.org/10.22175/mmb.11103.

Royal Society and Rand Europe (2018). *The Impact of Ammonia Emissions from Agriculture on Biodiversity: An Evidence Synthesis*. https://royalsociety.org/~/media/policy/projects/evidence-synthesis/Ammonia/Ammonia-report.pdf?la=en-GB.

Saadaoui, E., Ghazel, N., Romdhane, C.B. and Massoudi, N. (2017). Phosphogypsum: Potential uses and problems – A review. *International Journal of Environmental Studies* 74(4), 558-567. https://doi.org/10.1080/00207233.2017.1330582.

Sabaté, J. and Soret, S. (2014). Sustainability of plant-based diets: Back to the future. *The American Journal of Clinical Nutrition* 100 (supplement 1), 476S-482S. https://doi.org/10.3945/ajcn.113.071522.

Salmanzadeh, M., Balks, M.R., Hartland, A. and Schipper, L.A. (2016). Cadmium accumulation in three contrasting New Zealand soils with the same phosphate fertilizer history. *Geoderma Regional* 7(3), 271-278. https://doi.org/10.1016/j.geodrs.2016.05.001.

Sánchez-Bayo, F. and Wyckhuys, K.A.G. (2019). Worldwide decline of the entomofauna: A Review of its drivers. *Biological Conservation* 232, 8-27. https://doi.org/10.1016/j.biocon.2019.01.020.

Scheu, S., Ruess, L. and Bonkowski, M. (2005). Interactions between microorganisms and soil micro- and mesofauna. In *Microorganisms in Soils: Roles in Genesis and Functions*. *Soil Biology vol 3*. Buscot, F. and Varma, A. (eds.). Springer Nature. 253-275. https://www.springer.com/gp/book/9783540222200.

Schnug, E. and Lottermoser, B.G. (2013). Fertilizer-derived uranium and its threat to human health. *Environmental Science & Technology* 47(6), 2433-2434. https://doi.org/10.1021/es4002357.

Schrijver, R., Stijntjes, M., Rodríguez-Baño, J., Tacconelli, E., Rajendran, N.B. and Voss, A. (2018). Review of antimicrobial resistance surveillance programmes in livestock and meat in EU with focus on humans. *Clinical Microbiology and Infection* 24(6), 577-590. https://doi.org/10.1016/j.cmi.2017.09.013.

Schroeder, H.A. and Balassa, J.J. (1963). Cadmium: Uptake by vegetables from superphosphate in soil. *Science* 140(3568), 819-820. https://doi.org/10.1126/science.140.3568.819.

Schwabl, P., Köppel, S., Königshofer, P., Bucsics, T., Trauner, M., Reiberger, T. and Liebmann, B. (2019). Detection of various microplastics in human stool: A prospective case series. *Annals of Internal Medicine* 171(7), 453-457. https://doi.org/10.7326/m19-0618.

Science Advice for Policy by European Academies (SAPEA) (2019). *A Scientific Perspective on Micro-plastics in Nature and Society*. https://doi.org/10.26356/microplastics.

Scott, A.M., Beller, E., Glasziou, P., Clark, J., Ranakusuma, R.W., Byambasuren, O. *et al*. (2018). Is antimicrobial administration to food animals a direct threat to human health? A rapid systematic review. *International Journal of Antimicrobial Agents* 52(3), 316-323. https://doi.org/10.1016/j.ijantimicag.2018.04.005.

Selman, M., Greenhalgh, S., Diaz, R. and Sugg, Z. (2008). *Eutrophication and Hypoxia in Coastal Areas: A Global Assessment of the State of Knowledge*. Washington D.C: World Resources Institute. https://www.wri.org/publication/eutrophication-and-hypoxia-coastal-areas.

Sgro, G.V. and Reavie, E.D. (2018). Lake Erie's ecological history reconstructed from the sedimentary record. *Journal of Great Lakes Research* 44(1), 54-69. https://doi.org/10.1016/j.jglr.2017.11.002.

Shi, T., Zhang, Y., Gong, Y., Ma, J., Wei, H., Wu, X. *et al*. (2019). Status of cadmium accumulation in agricultural soils across China (1975-2016): From temporal and spatial variations to risk assessment. *Chemosphere* 230, 136-143. https://doi.org/10.1016/j.chemosphere.2019.04.208.

Shin, S.-J., Song, E.-S., Kim, J.-W., Lee, J.-H., Gautam, R., Kim, H. *et al*. (2019). Major environmental characteristics of swine husbandry that affect exposure to dust and airborne endotoxins. *Journal of Toxicology and Environmental Health* 82(4), 233-243. https://doi.org/10.1080/15287394.2019.1584596.

Shukla, S. and Saxena, A. (2018). Global status of nitrate contamination in ground water: Its occurrence, health impacts, and mitigation measures. In *Handbook of Environmental Materials Management*. Hussain, C. (ed.). Cham: Springer. 1-20. https://doi.org/10.1007/978-3-319-73645-7_20.

Silva, A.L.P., Prata, J.C., Walker, T.R., Duarte, A.C., Ouyang, W., Barcelò, D. and Rocha-Santos, T. (2021). Increased plastic pollution due to covid-19 pandemic: Challenges and recommendations. *Chemical Engineering Journal* 405,126683. https://doi.org/10.1016/j.cej.2020.126683.

Singer, A.C., Shaw, H., Rhodes, V. and Hart, A. (2016). Review of antimicrobial resistance in the environment and its relevance to environmental regulators. *Frontiers in Microbiology* 7, 1728. https://doi.org/10.3389/fmicb.2016.01728.

Smolders, E. (2001). Cadmium uptake by plants. *International Journal of Occupational Medicine and Environmental Health* 14(2), 177-183. https://www.researchgate.net/publication/11799510_Cadmium_uptake_by_plants.

Smolders, E. (2017). *Scientific Aspects Underlying the Regulatory Framework in the Area of Fertilisers – State of Play and Future Reforms*. *In-Depth Analysis for the IMCO Committee*. Brussels: Office of the European Union. https://op.europa.eu/en/publication-detail/-/publication/04a20719-fff0-11e6-8a35-01aa75ed71a1/language-en.

Solomou, A.D., Sfougaris, A.I., Vavoulidou, E.M. and Csuzdi, C. (2013). Species richness and density of earthworms in relation to soil factors in olive orchard production systems in Central Greece. *Communications in Soil Science and Plant Analysis* 44(1-4), 301-311. https://doi.org/10.1080/00103624.2013.741904.

Song, P., Wu, L. and Guan, W. (2015). Dietary nitrates, nitrites, and nitrosamines intake and the risk of gastric cancer: A meta-analysis. *Nutrients* 7(12), 9872-9895. https://doi.org/10.3390/nu7125505.

Song, Y., Wang, Y., Mao, W., Sui, H., Yong, L., Yang, D. *et al*. (2017). Dietary cadmium exposure assessment among the Chinese population. *PLoS ONE* 12(5), e0177978. https://doi.org/10.1371/journal.pone.0177978.

Srivastava, V., Sarkar, A., Singh, S., Singh, P., de Araujo, A.S.F. and Singh, R. (2017). Agroecological responses of heavy metal pollution with special emphasis on soil health and plant performances. *Frontiers in Environmental Science* 5, 64. https://doi.org/10.3389/fenvs.2017.00064.

Steinmetz, Z., Wollmann, C., Schaefer, M., Buchmann, C., David, J., Tröger, J. *et al*. (2016). Plastic mulching in agriculture. Trading short-term agronomic benefits for long-term soil degradation? *The Science of the Total Environment* 550, 690-705. https://doi.org/10.1016/j.scitotenv.2016.01.153.

Steyn D.G. (1961). An outbreak of urea poisoning among Bantu farms labourers in the Porgietersrust Distric, Transvaal. *South African Medical Journal* 35(9), 721-722. https://www.ajol.info/index.php/samj/article/view/176752.

Stocker, M.D., Pachepsky, Y.A., Hill, R.L. and Shelton, D.R. (2015). Depth-dependent survival of *Escherichia coli* and enterococci in soil after manure application and simulated rainfall. *Applied and Environmental Microbiology* 81(14), 4801-4808. https://doi.org/10.1128/AEM.00705-15.

Suman, J., Uhlik, O., Viktorova, J. and Macek, T. (2018). Phytoextraction of heavy metals: A promising tool for clean-up of polluted environment? *Frontiers in Plant Science* 9, 1476. https://doi.org/10.3389/fpls.2018.01476.

Sundblad, B.-M., Larsson, B.-M., Acevedo, F., Ernstgård, L., Johanson, G., Larsson, K. and Palmberg, L. (2004). Acute respiratory effects of exposure to ammonia on healthy persons. *Scandinavian Journal of Work Environment & Health* 30(4), 313-321. https://doi.org/10.5271/sjweh.800.

Svirčev, Z., Lalić, D., Savić, G.B., Tokodi, N., Backović, D.D., Chen, L. *et al*. (2019). Global geographical and historical overview of cyanotoxin distribution and cyanobacterial poisonings. *Archives of Toxicology* 93, 2429-2481. https://doi.org/10.1007/s00204-019-02524-4.

Tang, K.L., Caffrey, N.P., Nóbrega, D.B., Cork, S.C., Ronksley, P.E., Barkema, H.W. *et al*. (2017). Restricting the use of antibiotics in food-producing animals and its associations with antibiotic resistance in food-producing animals and human beings: a systematic review and meta-analysis. *Lancet Planetary Health* 1(8), E316-E327. https://doi.org/10.1016/S2542-5196(17)30141-9.

Thapa, R., Chatterjee, A., Awale, R., McGranahan, D.A. and Daigh, A. (2016). Effect of enhanced efficiency fertilizers on nitrous oxide emissions and crop yields: A meta-analysis. *Soil Science Society of America Journal* 80(5), 1121-1134. https://doi.org/10.2136/sssaj2016.06.0179.

Thorburn, P.J., Biggs, J.S., Weier, K.L. and Keating, B.A. (2003). Nitrate in groundwaters of intensive agricultural areas in Coastal Northeastern Australia. *Agriculture, Ecosystems & Environment* 94(1), 49-58. https://doi.org/10.1016/S0167-8809(02)00018-X.

Thorburn, P.J., Wilkinson, S.N. and Silburn, D.M. (2013). Water quality in agricultural lands draining to the Great Barrier Reef: A review of causes, management and priorities. *Agriculture, Ecosystems & Environment* 180, 4-20. https://doi.org/10.1016/j.agee.2013.07.006.

Tian, D. and Niu, S. (2015). A global analysis of soil acidification caused by nitrogen addition. *Environmental Research Letters* 10(2), 024019. https://iopscience.iop.org/article/10.1088/1748-9326/10/2/024019.

Tian, H., Xu, R., Canadell, J.G., Thompson, R.L., Winiwarter, W., Suntharalingam, P. *et al*. (2020). A comprehensive quantification of global nitrous oxide sources and sinks. *Nature* 586, 248-256. https://doi.org/10.1038/s41586-020-2780-0.

Tovar-Sánchez, E., Hernández-Plata, I., Martínez, M.S., Valencia-Cuevas, L. and Galante, P.M. (2018). Heavy metal pollution as a biodiversity threat. In *Heavy Metals.* Saleh H.E.M. and Aglan, R.F. (eds.). IntechOpen. Chapter 21. http://dx.doi.org/10.5772/intechopen.74052.

Townsend, A.R., Howarth, R.W., Bazzaz, F.A., Booth, M.S., Cleveland, C.C., Collinge, S.K. *et al*. (2003). Human health effects of a changing global nitrogen cycle. *Frontiers in Ecology and the Environment* 1(5), 240-246. https://doi.org/10.1890/1540-9295(2003)001[0240:HHEOAC]2.0.CO;2.

Treseder, K.K. (2008). Nitrogen additions and microbial biomass: A meta-analysis of ecosystem studies. *Ecology Letters* 11, 1111-1120. https://doi.org/10.1111/j.1461-0248.2008.01230.x.

Tubiello, F.N., Salvatore, M., Ferrara, A.F., House, J., Federici, S., Rossi, S. *et al*. (2015). The contribution of agriculture, forestry and other land use activities to global warming, 1990-2012. *Global Change Biology* 21(7), 2655-2660. https://doi.org/10.1111/gcb.12865.

Tubiello, F.N., Salvatore, M., Rossi, S., Ferrara, A., Fitton, N. and Smith, P. (2013). The FAOSTAT database of greenhouse gas emissions from agriculture. *Environmental Research Letters* 8(1), 015009. https://iopscience.iop.org/article/10.1088/1748-9326/8/1/015009/meta.

Tulsidas, H., Gabriel, S., Kiegiel, K. and Haneklaus, N. (2019). Uranium resources in EU phosphate rock imports. *Resources Policy* 61, 151-156. https://doi.org/10.1016/j.resourpol.2019.02.012.

Udeigwe, T.K., Teboh, J.M., Eze, P.N., Stietiya, M.H., Kumar, V., Hendrix, J. *et al*. (2015). Implications of leading crop production practices on environmental quality and human health. *Journal of Environmental Management* 151, 267-279. https://doi.org/10.1016/j.jenvman.2014.11.024.

United Nations Environment Programme (2010). *Final Review of Scientific Information on Cadmium. Version of December 2010*. https://wedocs.unep.org/xmlui/bitstream/handle/20.500.11822/27636/Cadmium_Review.pdf?sequence=1&isAllowed=y.

United Nations Environment Programme (2017). *Antimicrobial Resistance: Investigating the Environmental Dimension*. http://wedocs.unep.org/bitstream/handle/20.500.11822/22263/Frontiers_2017_CH1_EN.pdf?sequence=1&isAllowed=y.

United Nations Environment Programme (2019). *Global Chemicals Outlook II: From Legacies to Innovative Solutions*. https://www.unenvironment.org/explore-topics/chemicals-waste/what-we-do/policy-and-governance/global-chemicals-outlook.

United Nations Environment Programme (2021). *From Pollution to Solution – A Global Assessment of Marine Litter, including Plastics and Microplastics*. https://www.unep.org/resources/pollution-solution-global-assessment-marine-litter-and-plastic-pollution.

United Nations Environment Programme (2022). *Synthesis Report on Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them*. https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

United Nations Environment Programme and Climate and Clean Air Coalition (2021). *Global Methane Assessment: Benefits and Costs of Mitigating Methane Emissions*. https://www.unep.org/resources/report/global-methane-assessment-benefits-and-costs-mitigating-methane-emissions.

United States Environmental Protection Agency (2011). *Toxicological Review of Urea (CAS No. 57-13-6) In Support of Summary Information on the Integrated Risk Information System (IRIS)*. https://cfpub.epa.gov/ncea/iris/iris_documents/documents/toxreviews/1022tr.pdf.

United States Environmental Protection Agency (2015). *Chemical Advisory: Safe Storage, Handling, and Management of Solid Ammonium Nitrate Prills*. https://www.epa.gov/sites/production/files/2015-06/documents/an_advisory_6-5-15.pdf.

United States Environmental Protection Agency (2016). *Toxicological Review of Ammonia Noncancer Inhalation [CASRN 7664-41-7].* https://iris.epa.gov/static/pdfs/0422tr.pdf.

United States Environmental Protection Agency (2018). *U.S. Action Plan for Lake Erie. 2018-2023. Commitments and Strategy for Phosphorus Reduction*. https://www.epa.gov/sites/production/files/2018-03/documents/us_dap_final_march_1.pdf.

United States Environmental Protection Agency (2020a). What is acid rain?. https://www.epa.gov/acidrain/what-acid-rain. Accessed 28 December 2020.

United States Environmental Protection Agency (2020b). *Policy Assessment for the Review of the National Ambient Air Quality Standards for Particulate Matter*. https://www.epa.gov/system/files/documents/2021-10/final-policy-assessment-for-the-review-of-the-pm-naaqs-01-2020.pdf.

United States Environmental Protection Agency (2020c). Health and environmental effects of particulate matter (PM). https://www.epa.gov/pm-pollution/health-and-environmental-effects-particulate-matter-pm. Accessed 28 December 2020.

United States Environmental Protection Agency (2020d). Lake Erie. https://www.epa.gov/greatlakes/lake-erie. Accessed 28 December 2020.

United States Environmental Protection Agency (2020e). *U.S. Action Plan for Lake Erie*. https://www.epa.gov/glwqa/us-action-plan-lake-erie.

United States Environmental Protection Agency (2020f). Restoration of the Chesapeake Bay. https://www.epa.gov/restoration-chesapeake-bay. Accessed 28 December 2020.

United States Environmental Protection Agency (2020g). Radioactive material from fertilizer production. https://www.epa.gov/radtown/radioactive-material-fertilizer-production. Accessed 28 December 2020.

United States Environmental Protection Agency (2020h). Health effects from cyanotoxins. https://www.epa.gov/cyanohabs/health-effects-cyanotoxins. Accessed 28 December 2020

United States Geological Survey (2020). *Cadmium. Mineral Commodity Summaries.* https://pubs.usgs.gov/periodicals/mcs2020/mcs2020-cadmium.pdf.

United States Oceanic and Atmospheric Administration (n.d.). What are microplastics?. https://oceanservice.noaa.gov/facts/microplastics.html. Accessed 28 December 2020.

University of Nebraska-Lincoln (2020). Pathogens and organic matter. https://water.unl.edu/article/animal-manure-management/pathogens-and-organic-matter. Accessed 28 December 2020.

UN-Water (2021). *Summary Progress Update 2021 – SDG 6 – Water and Sanitation for All: March 2021*. https://www.unwater.org/new-data-on-global-progress-towards-ensuring-water-and-sanitation-for-all-by-2030/.

Vacchi-Suzzi, C., Kruse, D., Harrington, J., Levine, K. and Meliker, J.R. (2016). Is urinary cadmium a biomarker of long-term exposure in humans? a review. *Current Environmental Health Reports* 3(4), 450. https://doi.org/10.1007/s40572-016-0107-y.

Vanderplancke, G., Claireaux, G., Quazuguel, P., Huelvan, C., Corporeau, C., Mazurais, D. and Zambonino-Infante, J.-L. (2014). Exposure to chronic moderate hypoxia impacts physiological and developmental traits of European sea bass (*Dicentrarchus labrax*) larvae. *Fish Physiology and Biochemistry* 41, 233-242. https://doi.org/10.1007/s10695-014-0019-4.

Venkatesan, A.K. and Halden, R.U. (2013). National inventory of perfluoroalkyl substances in archived U.S. biosolids from the 2001 EPA National Sewage Sludge Survey. *Journal of Hazardous Materials* 252-253, 413-418. https://doi.org/10.1016/j.jhazmat.2013.03.016.

Victor, T.J. and Reuben, R. (2000). Effects of organic and inorganic fertilizers on mosquito populations in rice fields of Southern India. *Medical and Veterinary Entomology* 14(4), 361-368. https://www.researchgate.net/publication/12198638_Effects_of_organic_and_inorganic_fertilizers_on_mosquito_populations_in_rice_fields_of_Southern_India.

Wales, A.D. and Davies, R.H. (2015). Co-selection of resistance to antibiotics, biocides and heavy metals, and its relevance to foodborne pathogens. *Antibiotics* 4(4) 567-604. https://doi.org/10.3390/antibiotics4040567.

Wall, D.H., Nielsen, U.N. and Six, J. (2015). Soil biodiversity and human health. *Nature* 528, 69-76. https://doi.org/10.1038/nature15744.

Wang, J., Liu, X., Li, Y., Powell, T., Wang, X., Wang, G. and Zhang, P. (2019). Microplastics as contaminants in the soil environment: A mini-review. *Science of The Total Environment* 691, 848-857. https://doi.org/10.1016/j.scitotenv.2019.07.209.

Wang, K., Qiao, Y., Zhang, H., Yue, S., Li, H., Ji, X. and Crowley, D. (2018). Influence of cadmium-contaminated soil on earthworm communities in a subtropical area of China. *Applied Soil Ecology* 127, 64-73. https://doi.org/10.1016/j.apsoil.2018.02.026.

Wang, M., Chai, L., Zhao, H., Wu, M. and Wang, H. (2015). Effects of nitrate on metamorphosis, thyroid and iodothyronine deiodinases expression in *Bufo gargarizans larvae*. *Chemosphere* 139, 402-409. https://doi.org/10.1016/j.chemosphere.2015.07.037.

Wang, P., Chen, H., Kopittke, P.M. and Zhao, F.-J. (2019). Cadmium contamination in agricultural soils of China and the impact on food safety. *Environmental Pollution* 249, 1038-1048. https://doi.org/10.1016/j.envpol.2019.03.063.

Wang, S., Chen, H.Y.H., Tan, Y., Fan, H. and Raun, H. (2016). Fertilizer regime impacts on abundance and diversity of soil fauna across a poplar plantation chronosequence in Coastal Eastern China. *Scientific Reports* 6, 20816. https://doi.org/10.1038/srep20816.

Wang, S., Lu, W., Waly, N., Ma, C., Zhang, Q. and Wang, C. (2017). Recovery and germination of seeds after passage through the gut of Kazakh sheep on the north slope of the Tianshan Mountains. *Seed Science Research* 27(1), 43-49. https://doi.org/10.1017/S0960258517000022.

Ward, M.H., Jones, R.R., Brender, J.D., de Kok, T.M., Weyer, P.J., Nolan, B.T. *et al*. (2018). Drinking water nitrate and human health: An updated review. *International Journal of Environmental Research and Public Heath* 15(7), 1557. https://doi.org/10.3390/ijerph15071557.

Watson, C. and Cichra, C.E. (2013). *Fertilization of Fresh Water Fish Ponds*. *Fact Sheet FA-17*. Gainesville: University of Florida, Institute of Food and Agricultural Sciences. http://agrilife.org/fisheries2/files/2013/10/Fertilization-of-Fresh-Water-Fish-Ponds.pdf.

Wear, S.L. (2016). Missing the boat: Critical threats to coral reefs are neglected at global scale. *Marine Policy* 74, 153-157. https://doi.org/10.1016/j.marpol.2016.09.009.

Weber, R., Herold, C., Hollert, H., Kamphues, J., Blepp, M. and Ballschmiter, K. (2018). Reviewing the relevance of dioxin and PCB sources for food from animal origin and the need for their inventory, control and management. *Environmental Sciences Europe* 30, 42. https://doi.org/10.1186/s12302-018-0166-9.

Weithmann, N., Möller, J.N., Löder, M.G.J., Piehl, S., Laforsch, C. and Freitag, R. (2018). Organic fertilizer as a vehicle for the entry of microplastic into the environment. *Science Advances* 4(4), eaap8060. https://doi.org/10.1126/sciadv.aap8060.

Wesseling, C., Glaser, J., Rodríguez-Guzmán, J., Weiss, I., Lucas, R., Peraza, S. *et al*. (2020). Chronic kidney disease of non-traditional origin in Mesoamerica: A disease primarily driven by occupational heat stress. *Revista Panamericana de Salud Pública* 44, e15. https://doi.org/10.26633/RPSP.2020.15.

White, P. (2017). *Aquaculture Pollution: An Overview of Issues with a Focus on China, Vietnam and the Philippines*. Washington, D.C.: World Bank. https://openknowledge.worldbank.org/handle/10986/29249.

Wiggenhause, M., Bigalke, M., Imseng, M., Keller, A., Rehkämper, M., Wilcke, W. and Frossard, E. (2019). Using isotopes to trace freshly applied cadmium through mineral phosphorus fertilization in soil-fertilizer-plant systems. *Science of The Total Environment* 648, 779-786. https://doi.org/10.1016/j.scitotenv.2018.08.127.

World Health Organization (2003). *Nitrate and Nitrite in Drinking Water*. *Background Document for Development of WHO Guidelines for Drinking-water Quality*. WHO/SDE/WSH/04.03/56. https://apps.who.int/iris/handle/10665/75380.

World Health Organization (2016). *Nitrate and Nitrite in Drinking-water*. *Background Document for Development of WHO Guidelines for Drinking-water Quality.* WHO/FWC/WSH/16.52. https://pdfslide.net/documents/nitrate-and-nitrite-in-drinking-water-whoint-iii-preface-access-to-safe-drinking-water.html?page=2.

World Health Organization (2017a). *Guidelines for Drinking-water Quality: Fourth Edition - Incorporating the First Addendum*. https://www.who.int/publications/i/item/9789241549950.

World Health Organization (2017b). Antimicrobial resistance: The food chain, 1 November. https://www.who.int/news-room/q-a-detail/antimicrobial-resistance-in-the-food-chain. Accessed 29 December 2020.

World Health Organization (2018). Ambient (outdoor) air pollution. https://www.who.int/news-room/fact-sheets/detail/ambient-(outdoor)-air-quality-and-health. Accessed 29 December 2020.

World Health Organization (2019a). *Preventing Disease through Healthy Environments: Exposure to Cadmium: A Major Public Health Concern*. https://apps.who.int/iris/handle/10665/329480.

World Health Organization (2019b). *Microplastics in Drinking-water*. https://www.who.int/publications/i/item/9789241516198.

World Health Organization (2020a). Soil-transmitted helminth infections. https://www.who.int/news-room/fact-sheets/detail/soil-transmitted-helminth-infections. Accessed 28 December 2020.

World Health Organization (2020b). Antimicrobial resistance. https://www.who.int/news-room/fact-sheets/detail/antimicrobial-resistance. Accessed 30 December 2020

World Health Organization, Food and Agriculture Organization of the United Nations and World Organisation for Animal Health (2020). *Technical Brief on Water, Sanitation, Hygiene and Wastewater Management to Prevent Infections and Reduce the Spread of Antimicrobial Resistance*. http://www.fao.org/3/ca9120en/CA9120EN.pdf.

World Resources Institute (2013). Eutrophication and hypoxia map data set. https://www.wri.org/resources/data-sets/eutrophication-hypoxia-map-data-set. Accessed 21 January 2021.

Wu, N., Zhang, W., Xie, S., Zeng, M., Liu, H., Yang, J. *et al*. (2020). Increasing prevalence of antibiotic resistance genes in manured agricultural soils in Northern China. *Frontiers of Environmental Science & Engineering* 14, 1. https://doi.org/10.1007/s11783-019-1180-x.

Wu, Y., Gu, B., Erisman, J.W., Reis, S., Fang, Y., Lu, X. and Zhang, X. (2016). PM2.5 pollution is substantially affected by ammonia emissions in China. *Environmental Pollution* 218, 86-94. https://doi.org/10.1016/j.envpol.2016.08.027.

Xie, Y., Fan, J., Zhu, W., Amombo, E., Lou, Y., Chen, L. and Fu, J. (2016). Effect of heavy metals pollution on soil microbial diversity and bermudagrass genetic variation. *Frontiers in Plant Science* 7, 755. https://doi.org/10.3389/fpls.2016.00755.

Xu, R., Tian, H., Pan, S., Prior, S.A., Feng, Y., Batchelor, W.D. *et al*. (2019). Global ammonia emissions from synthetic nitrogen fertilizer applications in agricultural systems: Empirical and process-based estimates and uncertainty. *Global Change Biology* 25(1), 314-326. https://doi.org/10.1111/gcb.14499.

Yu, G., Wang, J., Liu, L., Li, Y., Zhang, Y. and Wang, S. (2020). The analysis of groundwater nitrate pollution and health risk assessment in rural areas of Yantai, China. *BMC Public Health* 20, 437. https://doi.org/10.1186/s12889-020-08583-y.

Zhang, J.N., Yang, L., Zhang, M., Liu, Y.-S., Zhao, J.-L., He, L.-Y. *et al*. (2019). Persistence of androgens, progestogens, and glucocorticoids during commercial animal manure composting process. *Science of The Total Environment* 665, 91-99. https://doi.org/10.1016/j.scitotenv.2019.02.120.

Zhang, P., Ma, G., Wang, C., Lu, H., Li, S., Xie, Y. *et al*. (2017). Effect of irrigation and nitrogen application on grain amino acid composition and protein quality in winter wheat. *PLoS ONE* 12(6), e0178494. https://doi.org/10.1371/journal.pone.0178494.

Zhang, W.-F., Dou, Z.-X., He, P., Ju, X.-T., Powlson, D., Chadwick, D. *et al*. (2013). New technologies reduce greenhouse gas emissions from nitrogenous fertilizer in China. *Proceedings of the National Academy of Sciences of the United States* 110(21), 8375-8080. https://pubmed.ncbi.nlm.nih.gov/23671096/.

Zhang, Y., Lu, J., Wu, J., Wang and Luo, Y. (2020). Potential risks of microplastics combined with superbugs: Enrichment of antibiotic resistant bacteria on the surface of microplastics in mariculture system. *Ecotoxicology and Environmental Safety* 187, 109852. https://www.sciencedirect.com/science/article/abs/pii/S0147651319311832.

Zhou, Z. (2015). *A Global Assessment of Nitrate Contamination in Groundwater*. *Internship Report*. International Groundwater Resources Assessment Centre and Wageningen University. https://www.un-igrac.org/sites/default/files/resources/files/A%20Global%20Assessment%20of%20Nitrate%20Contamination%20in%20Groundwater.pdf.

Zieritz, A., Lopes-Lima, M., Bogan, A.E., Sousa, R., Walton, S., Rahim, K.A.A. *et al*. (2016). Factors driving changes in freshwater mussel (Bivalvia, Unionida) diversity and distribution in Peninsular Malaysia. *Science of The Total Environment* 571, 1069-1078. https://doi.org/10.1016/j.scitotenv.2016.07.098

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them

Envisioning A Chemical-Safe World

Chapter 10 of 12

The impact of fertilizer use

Contents

1 Global drivers, actors and policies affecting pesticides and fertilizer use

2 Status and trends of pesticide use

3 The regulatory and policy environment for pesticide management

4 Environmental and health effects of pesticide use

5. The environmental, human health and economic impacts of pesticides

6 Current pesticide risk reduction and risk management

7 Status and trends of fertilizer use

8 The regulatory and policy environment of fertilizer management and use

9 Environmental and health effects of fertilizer use

About ii

10 The impact of fertilizer use 1

10.1 Overview 1

10.2 Environmental and health costs 2

10.2.1 Human health 2
10.2.2 Ecosystem services and planetary boundaries 4
10.2.3 Total monetary costs of human and environmental damage 5

10.3 Agronomic, economic, health and environmental benefits 5

10.3.1 Agronomic benefits 5
10.3.2 Economic benefits 6
10.3.3 Health 7
10.3.4 Environmental benefits 8

10.4 Data gaps and challenges 9

References 10

11 Current fertilizer risk reduction and risk management

12 Transformative actions to minimize the adverse impacts of pesticide and fertilizers

List of Figures

Figure 10.2-1 Human fatalities involving ammonium nitrate, 1916-2019. 3

About

In December 2017, Resolution 4 of the 3rd Session of the United Nations Environment Assembly (UNEA 3) requested "the Executive Director to present a report on the environmental and health impacts of pesticides and fertilizers and ways of minimizing them, given the lack of data in that regard, in collaboration with the World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO) and other relevant organizations by the fifth session of the United Nations Environment Assembly". In response to this request, UNEP published a *Synthesis Report on the Environmental and Health Impacts of Pesticides and Fertilizers and Ways to Minimize Them*[1] in February 2022 (United Nations Environment Programme [UNEP] 2022).

The overall goal of the synthesis report is to provide the information base to enable other advocacy actions to be taken by stakeholders to minimize the adverse impacts of pesticides and fertilizers. Specific objectives of the synthesis report are to:

- Update understanding of current pesticide and fertilizer use practices;
- Present major environmental and health effects of pesticides and fertilizers, during their life cycle, and identify key knowledge gaps;
- Review current management practices, legislation and policies aimed at reducing risks in the context of the global chemicals, environmental and health agenda;
- Identify opportunities to minimize environmental and health impacts, including proven and innovative approaches.

This chapter on "The impact of fertilizer use" is the 10th in a series of 12 chapters that make up a comprehensive compilation of scientific information. The chapters were developed to both inform and further elaborate on the information provided in the synthesis report. Please note that the disclaimers and copyright from the synthesis report apply.

1 The Synthesis report is available at https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

10

The impact of fertilizer use

10.1 Overview

> With fertilizers, we face a kind of Goldilocks problem. Some places have too few nutrients and therefore poor crop production, whereas others have too much, leading to pollution. (Foley 2011, pp. 64-65)

Much is still unknown about the adverse environmental and health effects of fertilizers. As emphasized in Chapter 9, the links between fertilizer use and human health, for example, are not fully understood and quantifying the effects of fertilizer use is complex. Large information gaps exist with regard to the impact of fertilizers. In the case of some of the information currently available, there are concerns about reliability and the comparability of data from different studies.

More is known about fertilizers' agronomic and economic benefits. But even in the case of agronomic and economic effects, fertilizer is usually part of a technology package. Where data exist, it is difficult to compare those from different studies since assumptions may vary among the studies. [Chapters 10.2, 10.3 and 10.4]

Fertilizers are used for their "provisional" role. Their main benefits include provision of food, feed and fibre. Fertilizers support food security and can contribute to the provision of nutritious food and feeds (in terms of both quality and quantity) through agronomic biofortification (Global Panel on Agriculture and Food Systems for Nutrition 2016a; Joy *et al*. (2017). (Chapters 10.1, 10.3.1 and 10.3.3]

Few attempts have been made to quantify the adverse effects of fertilizer use, including through their overuse and contaminants. Doing so can be complex. Studies have suggested that the monetary costs could be substantial (van Grinsven *et al*. 2013; Sobota *et al*. 2015; Zhang *et al*. 2020). [Chapters 10.2.1 and 10.2.3]

Direct health impacts of fertilizers are rare, but may occur through, for example, inhalation of ammonia and dusts from manure. Storage and transport accidents involving fertilizers may cause significant loss of human life (Zhao 2016; Guglielmi 2020). [Chapter 10.2.1]

The planetary boundary for nitrogen and phosphorus flows is considered to have been crossed (Stockholm Resilience Centre 2020). Agricultural production has been identified as a major driver for the exceedance of planetary boundaries, with overfertilization and low nutrient use efficiencies contributing to nutrient losses (Campbell *et al* 2017). [Chapter 10.2.2]

The use of fertilizers provides economic benefits for various stakeholders. Their use by farmers is mostly profitable and can contribute to better livelihoods for farming households (Donkor *et al*. 2019; Martey, Kuwornu and Adjebeng-Danquah 2019). Nevertheless, farmers sometimes incur fertilizer-related losses (Kelly 2005). [Chapter 10.3.2]

Excessive use increases fertilizers' negative effects on ecosystem services. On the other hand, their sustainable use can reduce these effects and even generate ecosystem benefits, for example through increasing soil carbon or avoiding conversion of forests and pastures to cropland, which in turn reduces loss of ecosystem services (Han *et al*. 2016; Tang *et al*. 2019). [Chapters 10.2.2 and 10.3.4].

10.2 Environmental and health costs

Despite indications that fertilizers can have negative effects on the environment and health, there are no global studies on the total costs of fertilizer pollution. However, studies have attempted to estimate the global or regional impacts of some pollutants that could potentially be from fertilizer.

Following the literature review conducted under this report, scientific research appears to be more focused on nitrogen than on other potential pollutants from fertilizers. Scientists have attempted to bring the world's attention to the nitrogen issue through numerous publications and the creation of an international platform (International Nitrogen Management System 2021). In 2019 scientists sent an open letter to the Secretary-General of the United Nations requesting that global action on nitrogen be mobilized (UK Centre for Ecology and Hydrology 2019).

10.2.1 Human health

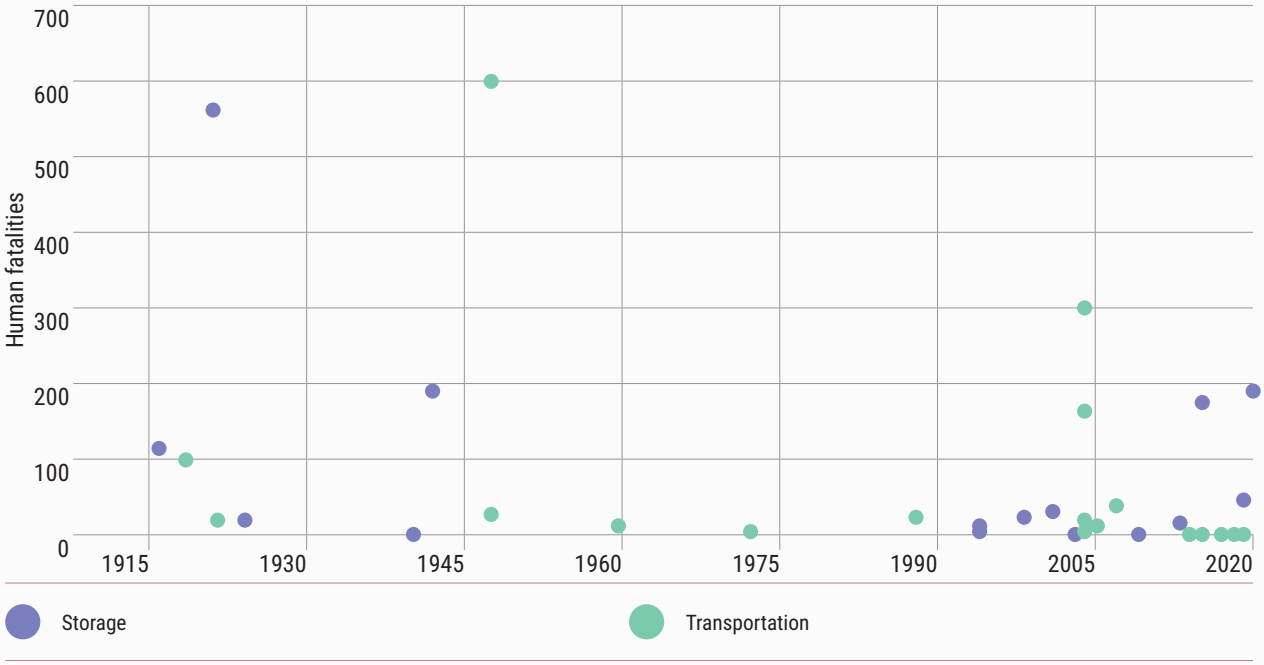
Premature deaths

There are no estimates of global premature deaths due to fertilizers. Existing studies suggest that fertilizers could contribute to a significant number of these deaths annually.

According to estimates by Pozzer *et al*. (2017), a 50 per cent reduction in agricultural emissions could reduce the number of premature deaths by 250,000 per year; assuming no agricultural emissions premature deaths would be reduced by 800,000. It should be noted that on this topic no other publications with global coverage were found. However, a study on maize (corn) production in the United States associated 4,300 premature deaths with reduced air quality due to nitrogen from inorganic fertilizer and manure (Hill *et al*. 2019). According to Anderson *et al*. (2019), about 67,000 deaths per year in 33 Organisation for Economic Co-operation and Development (OECD) and European Union (EU)/European Economic Area (EEA) countries could be due to antimicrobial resistance (AMR). The contribution of manure is not estimated in that study.

Deaths from accidents involving fertilizers have been reported, but they are rare. Dozens of major incidents involving (reactive) nitrogen fertilizers, especially ammonium nitrate, have been documented in different parts of the world (United States Environmental Protection Agency 2015). While guidelines have been developed on the transport, storage and handling of ammonium nitrate fertilizer (Chapter 1.7), accidents resulting in fatalities do occur (Figure 10.2-1). For example, in 2015 an accident in the port city of Tianjin,

Figure 10.2-1 Human fatalities involving ammonium nitrate, 1916-2019. Source of data: Compiled by UNEP



China killed 173 people and injured nearly 800 (Zhao 2016); in 2020 at least 220 people died and many more were injured in an accident in Beirut, Lebanon that left about 300,000 people homeless and destroyed enormous amounts of property. According to authorities, the latter accident was the result of 2,750 tons of ammonium nitrate having been stored improperly for six years in a port warehouse (Guglielmi 2020).

Human health costs

There are no global estimates of the total human health costs of fertilizer use. However, attempts have been made to estimate the costs of pollution by nitrogen and other pollutants, pathogens, and AMR, to all of which fertilizer is a contributor. Despite lack of data, this report presents the few published studies. It should be noted that the data in these studies are subject to much uncertainty.

Several studies have estimated the damage costs to human health of agricultural nitrogen. In the United States, Hill *et al*. (2019) estimated that the air quality-related health costs of using nitrogen fertilizer (inorganic and manure) in maize production could be in the range of United States dollars (USD) 14-64 billion (average USD 39 billion).

For potentially toxic trace elements like cadmium, health costs could be minimal. This is especially true in the case of inorganic fertilizers, as they are used in quantities that are too small to affect human health (Lugon-Moulin *et al*. 2006; Pizzol, Smart and Thomsen 2014). According to Pizzol Smart and Thomsen (2014), the health costs of cadmium in inorganic fertilizer were estimated to average 15.53 euros/year for each square kilometre of Danish agricultural soil to which this fertilizer was applied, while the health costs of cadmium in organic inputs were almost double this amount (28.60 euros/year for sludge and 37.04 euros/year for pig manure). These studies suggest that toxic trace elements in organic inputs can pose a slightly higher risk than those in inorganic fertilizers, but that the risks posed by organic inputs may not be significant in the short term.

In the case of pathogens, the monetary value of health costs could be at least USD 1 billion. For example, the global burden of cystic echinococcosis, a health outcome of helminths, has been estimated to be USD 0.8 billion per year (Budke, Deplazes and Torgerson 2006) and health costs related to foodborne pathogens have been estimated to be about USD 380 million/year (Tegtmeier and Duffy 2004).

The study by Anderson *et al*. (2019) of AMR-related health costs in OECD and EU/European Economic Area (EEA) countries found that, according to OECD models, improved use of antibiotics could reduce deaths from AMR by half and save about 2.3 billion euros. (The contribution of manure was to AMR was not estimated.) Since about 70-80 per cent of antibiotics produced are used in livestock (World Economic Forum 2017), it is possible that a proportion of these costs is due to handling and use of manure. However, according to a study by Miller *et al*. (2019), transmission of AMR to food grown in soil to which animal manure has been applied may be minimal.

10.2.2 Ecosystem services and planetary boundaries

There is no publication presenting total global estimates of ecosystem damage due to nitrogen. However, cost estimates by Sutton *et al*. (2013) of global damage caused by nitrogen include costs to the environment. Regional estimates based on the available literature are presented below. In addition to the monetary value of damage to ecosystem services, the planetary boundary for nitrogen and phosphorus flows is considered to have been exceeded.

Sutton *et al*. (2019) estimated global damage to the coastal aquatic environment by nitrogen and phosphorus, associated with hypoxia, to be about USD 170 billion annually. These costs vary among regions. Studies have estimated the costs of eutrophication in the EU and the United States to be in the billions of dollars. In the United States annual losses due to eutrophication in respect of recreational water usage, waterfront property values, threatened and endangered species recovery efforts, and spending on drinking water were estimated to be USD 2.2 billion by Dodds *et al*. (2009). In the EU in 2008 the costs of eutrophication were estimated by van Grinsven *et al*. (2018) to be 93 billion euros (58 billion euros and 35 billion euros for nitrogen leaching and ammonia deposition, respectively). Since nitrogen and phosphorus both contribute to eutrophication (Chapter 9, Figure 9.3-1), the damage cost of phosphorus to the aquatic environment is probably covered in all these studies.

Little is known about the costs of nutrient losses with regard to ecosystem services such as water regulation and biodiversity. For example, no data were found concerning the impacts of nutrient loads on water regulation services provided by wetlands, although wetlands provide services such as water supply and the control of pollution and flooding (International Water Management Institute 2006). Loss of coastal habitats and coral reefs increases risks from floods and hurricanes to life and property for 100-300 million people (IPBES 2019). In the EU in 2008 the costs of degradation of the terrestrial ecosystem by ammonia deposition was estimated to be 5 billion euros (van Grinsven *et al*. 2018). It has been pointed out by de Graaff *et al*. (2019) that more studies are needed on the links between environmental change, soil biodiversity and ecosystem functioning.

The concept that there are nine planetary boundaries beyond which anthropogenic change will put the Earth system outside a safe operating space for humanity was introduced in 2009 (Rockström *et al*. 2009; Stockholm Resilience Centre 2020). It has been further developed by Steffen *et al*. (2015), Willet *et al*. (2019) and others. The nine planetary boundaries are land-system change, freshwater use, biogeochemical flows, biosphere integrity, climate change, ocean acidification, stratospheric ozone depletion, atmospheric aerosol loading, and novel entities. Nitrogen and phosphorus flows to the biosphere and oceans are considered to be beyond the estimated safe planetary boundary for biochemical flows (Stockholm Resilience Centre 2020). However, it should be noted that the actual positioning of biochemical flows and other boundaries – and how to interpret these boundaries – continues to be debated (Mace *et al*. 2014; Vargas, Willemen and Hein 2018; Willet *et al*. 2019; Pickering and Perrson 2020).

Agricultural production has been identified as a major driver for the exceedance of planetary boundaries, with overfertilization and low nutrient use efficiencies contributing to nutrient losses (Campbell *et al*. 2017). Gerten *et al*. (2020) estimate that half of current global food production depends on planetary boundary transgressions. The key prerequisites identified

by the authors for a transformation towards more sustainable production and consumption patterns are spatially redistributed cropland, improved water-nutrient management, food waste reduction and dietary changes.

10.2.3 Total monetary costs of human and environmental damage

Some regional and national studies, and one global study, have attempted to assess the damage costs of nitrogen damage for both the environment and human health.

In the United States, Sobota *et al*. (2015) estimated that potential damage to the environment and human health from nitrogen use in agriculture could be in the range of USD 59-340 billion/year (median USD 157 billion). In the EU-27, damage from agricultural nitrogen was estimated by van Grinsven *et al*. (2013) to be in the range of 35-230 billion euros/year. In China, according to a study on ammonia emissions by Zhang *et al*. (2020), reducing agricultural ammonia emissions could result in environmental and human health benefits of about USD 18-42 billion. Another study in Europe by von Blottnitz *et al*. (2006) estimated that one kilogram of nitrogen fertilizer resulted in damage costs of about 0.3 euros.

Globally, the costs of nitrogen damage associated with ecosystems, climate change and human health could, according to Sutton *et al*. (2013), be in the range of 300-3,000 billion euros (the equivalent of around USD 340-3,400 billion in 2019). However, the authors emphasize that this estimate is largely based on EU estimates (van Grinsven *et al*. 2013) about which there are considerable uncertainties. Furthermore, there is a lack of data on fertilizers' contribution to global estimates of the costs of nitrogen damage although that contribution could be substantial. Sutton *et al*. (2013) suggested that improving nutrient use efficiency by 20 per cent could reduce damage costs with regard to biodiversity, the climate and humans by about USD 170 billion per year. Furthermore, in the EU study by van Grinsven et al. (2013) agriculture was estimated to contribute about 47 per cent of the damage costs.

Environmental and health costs associated with nutrient pollution are often much greater than the costs of measures to abate these pollutants (Sutton *et al*. 2013), as demonstrated in the following examples. The benefits of a theoretical complete phase-out of ammonia emissions from agriculture in the EU would outweigh the costs of these measures by at least ten times (Giannadaki *et al*. 2018). In China, Zhang *et al*. (2020) estimate that mitigating agricultural ammonia emissions by 38-67 per cent would cost about USD 6-11 billion and that the overall societal benefits would be about USD 18-42 billion. According to a World Water Assessment Programme report (United Nations Educational, Scientific and Cultural Organization [UNESCO] 2017), for every dollar spent on sanitation, the estimated return to society is USD 5.5 and simultaneous recovery of both energy and nutrients from wastewater can be profitable. However, recovering nutrients from wastewater is unlikely to be profitable for the recycling industry (Molinos-Senante *et al*. 2011; Kanter, Zhang and Mauzerall 2014). Options to encourage recycling can include offering subsidies to industry and integrating energy and fertilizer policies.

10.3 Agronomic, economic, health and environmental benefits

10.3.1 Agronomic benefits

Historically, the use of fertilizers, in combination with improved varieties and better management practices, has resulted in increased yields and increased production. For example, cotton yields tripled in West Africa between 1960 and 1985 with increased fertilizer use and better management practices (Pieri 1989, as cited by Bationo and Waswa 2011).

Similarly, the Green Revolution was associated with increased use of fertilizers combined with improved varieties and better management practices. In Asia

it resulted in a doubling of cereal production (from 313 million to 650 million tons/year) between 1970 and 1995 (Hazell 2009). Although the contribution of fertilizers to increased yields has not been quantified, it is likely to have been substantial. For example, according to a long-term study by Stewart *et al*. (2005), the average percentage of yields attributable to fertilizer generally ranged from about 40 to 60 per cent in the United States and the United Kingdom and tended to be much higher in the tropics.

10.3.2 Economic benefits

Global data on the economic benefits of fertilizer use are lacking. Furthermore, the available data do not allow such an overview. However, the value of agricultural crop production globally outweighs the value of the fertilizers used. For example, in 2016 agricultural crop production was valued at USD 2.6 trillion (IPBES 2019). If the contribution of fertilizers to yield is about 50 per cent, as reported in the literature (Stewart et al. 2005; Hazell 2009), in 2016 they could have contributed produce worth about USD 1.3 trillion (i.e., 50 per cent of the USD 2.6 trillion estimated by IPBES). These values should, however, be taken with caution.

According to a report by the International Fertilizer Association, in 2018 global revenue from sales of inorganic fertilizers was about USD 151 billion (International Fertilizer Association [IFA] 2020). There are no global estimates of the monetary value of organic inputs (e.g., manure). If the 110 million tons of nitrogen in manure estimated to be left in pasture or applied to land annually (Chapter 7) were replaced with nitrogen from inorganic fertilizers, the cost of inorganic fertilizers, based on the revenue reported by IFA (2020), could be slightly above USD 70 billion dollars. (This estimate is based on the assumption that inorganic fertilizers supply about 110 million tons of nitrogen per year; see Chapter 7, Figure 7.3-5). The actual value of nitrogen in manure could, however, be less than USD 70 billion. For example, according to studies by Rurangwa, Vanlauwe and Giller (2018) and Leip *et al*. (2019), the cost of a kilogram of nitrogen in inorganic fertilizer in Rwanda could be more than double that of a kilogram of nitrogen in manure. Manure also supplies other nutrients (e.g., phosphorus and potassium), but there are no global estimates of the quantities of these nutrients. Overall, there are no global estimates of the total value of fertilizers used in agriculture (i.e., the combined value of inorganic and organic fertilizers). It is not possible to produce such estimates for this report owing to lack of data.

Studies have demonstrated that fertilizer use is mostly economically profitable, but that it can also be unprofitable. In Africa, Asia and Latin America the value of additional yields obtained due to fertilizer use was more than twice the cost of the fertilizer applied in many cereal experiments (Kelly 2005). However, in some cases the cost of fertilizer was more than, or equal to, the value of the additional yield. The rule of thumb is that the increase in produce value must be at least double the fertilizer cost before a farmer will consider fertilizer use (Kelly 2005).

Fertilizer use can have numerous benefits for farmers. For example, in cassava production in Nigeria it was associated with reduced food insecurity, reduced poverty, and improved well-being of rural farmers (Donkor *et al*. 2019). In Ghana fertilizer use was associated with increased household income in rice farming (Martey, Kuwornu and Adjebeng-Danquah 2019). However, while literature quantifying the contribution of such benefits compared with total benefits are lacking, overall, the increase in fertilizer use per unit of land, as well as in total global use over the years, is an indicator that it has economic benefits.

Global studies comparing the economic benefits of using fertilizers (for farmers) with damage costs are lacking. Existing studies suggest that the cost of pollution could either outweigh or be outweighed by the economic benefits of fertilizer use. For example, estimates by van Grinsven *et al*. (2013) suggest that the cost of pollution from nitrogen use in agricultural production in the EU-27 in 2008 could have been greater than the economic benefit (35-230 billion euros/year vs. 20-80 billion euros/year). In China Tang *et al*. (2019) reported that an increase in the net economic value of vegetables due to use of fertilizers (inorganic and organic) outweighed environmental and health costs

(i.e., damage to ecosystems and human health and contribution to global warming) by ≥15 times. The benefits increased by about 133,000-222,000 Yuan/ha/year and the damage increased by 6,000-8,000 Yuan/ha/year. However, with an increase in the amount of fertilizer used, the increase in benefits is bound to slow down while damage costs continue to rise.

The question of who pays for pollution damage requires careful consideration. In the case of non-point pollution, identifying polluters can be difficult and expensive (FAO 2018a; FAO 2018b). Passing costs on to polluters provides incentives for sustainable use of fertilizers. However, if the costs are too high, industry and farmers can be pushed out of business. On the other hand, passing costs on to consumers makes food more expensive.

10.3.3 Health

Globally, malnutrition represents the greatest risk factor in regard to the global burden of disease (Global Panel on Agriculture and Food Systems for Nutrition 2016b). The health costs of all forms of malnutrition could be up to USD 3.5 trillion (of which USD 0.5 trillion for overnutrition, i.e., overweight and obesity) (United Nations Children's Fund [UNICEF] 2018).

Undernourishment

Fertilizers have contributed to increased food production globally (Bruulsema *et al*. 2012). Erisman *et al*. (2008) estimated that in 2008 nitrogen fertilizers produced using the Haber-Bosch process were responsible for feeding about 48 per cent of the world population. They concluded that the lives of about half of world population had been made possible through the use of these fertilizers.

Data on fertilizers' global impact on undernourishment is lacking. However, evidence from the Green Revolution demonstrates that they have made a substantial contribution to human health. Hazell (2009) reported that the Green Revolution in Asia resulted in an increase of nearly 30 per cent in the amount of calories available per person despite a 60 per cent population increase between 1970 and 1995.

Today almost 0.8 billion people are undernourished (Peng and Berry 2018), 149 million children are stunted (Development Initiatives 2020), and about 45 per cent of deaths among children under five are associated with undernutrition (WHO 2020). As the global population continues to increase, the number of undernourished people may continue to grow. Seachinger *et al*. (2019) estimated that to feed a projected world population of 10 billion in 2050, calorie production will have to be about 56 per cent more than in 2010. In sub-Saharan Africa, where the population is expected to double by 2050, providing healthy diets to all will require increased fertilizer use (Elrys *et al*. 2020). Overall, responsible use of fertilizers can contribute to reducing undernourishment

Micronutrient deficiencies (hidden hunger)

Globally, over 2 billion people are estimated to suffer from micronutrient deficiencies caused by dietary deficiencies of nutrients and vitamins (FAO *et al*. 2020). The estimate by Peng and Berry (2018) that 33-50 per cent of the world population is micronutrient deficient is consistent with the estimate of 2 billion people in FAO *et al*. (2020) In 2019, 340 million children were estimated to suffer from micronutrient deficiencies.

The micronutrients for which agronomic biofortification has been carried out, or which have been the subject of experiments, include selenium, zinc, iron and iodine. About a billion people suffer from iron deficiency (Global Burden of Disease Study 2013 Collaborators 2015). Zinc deficiency could be affecting 1.1 billion people (Wessells and Brown 2012). In 2002 WHO reported that 1.4 per cent of deaths worldwide (0.8 million) were attributable to zinc deficiency (1.4 per cent in males and 1.5 per cent in females). Attributable DALYs were higher (28 million worldwide), with zinc deficiency accounting for about 2.9 per cent of worldwide loss of healthy life years (WHO 2002). Two decades ago, some 0.5-1 billion people could have been suffering from selenium deficiency (Combs 2001).

Case studies on the health benefits of consuming foods biofortified through fertilizer use are scarce. The only case found was that of the national programme in Finland, where selenium has

been added to fertilizers since 1985 to reduce deficiencies in humans (reviewed by Alfthan *et al*. 2015). The authors reported that the concentration of selenium in cereals increased and that dietary intake in 2014 was double the intake in 1985. They argued, however, that the impact of the programme on human health could not be determined due to lack of a selenium deficiency disease and of a reference population.

The benefits of addressing malnutrition can outweigh the costs. Addressing global micronutrient deficiency through supplementation, food biofortification and agronomic biofortification would incur annual costs of about USD 1.2 billion and benefits of about USD 15.3 billion (Global Panel on Agriculture and Food Systems for Nutrition 2016a). Joy *et al*. (2017) reported that agronomic biofortification with zinc has both agronomic and health benefits. They estimated that the yield benefits of fertilizer biofortification were more than 10 times the cost of fertilizer, while DALYs decreased by about 40 per cent and gross returns from reduction in disease burden was at least eight times greater on average than the costs of fertilizer.

Other health benefits

The existing literature suggests that nitrates can have beneficial effects. A systematic literature search by Blekkenhorst *et al*. (2018) concluded there is strong evidence that dietary nitrate (e.g., derived from vegetables) can reduce cardiovascular disease in humans. A narrative review by Wikoff et al. (2018) came to a similar conclusion about the benefits of nitrates in regard to cardiovascular disease. Organic nitrates are also among the commonly prescribed medications for treatment and management of cardiovascular disorders (Adams and Khan 2020). Since nitrogen from fertilizer has been linked to nitrates in, for example, vegetables (Salehzadeh *et al*. 2020), it is likely that nitrogen fertilizers can have a positive effect on cardiovascular disease. However, evidence of the link between nitrates from fertilizer and health benefits is lacking, so that the positive impact remains unknown or undocumented.

10.3.4 Environmental benefits

Soil carbon

Although experiments (Ge, Zhu and Jiang 2018) and meta-analysis of published experimental data (Han *et al*. 2016) have reported that fertilizers can have positive effects on soil carbon, literature concerning the quantification of the impacts of fertilizer on soil carbon is scarce. Baveye, Baveye and Gowdy (2016), who reviewed the evolution of research on soil ecosystem services during the last two decades, reported that scientists have made efforts to quantify soil services but have shown minimal interest in monetary valuation. Still, scientists (Lal 2014; Brady *et al*. 2015) have made efforts to put values on changes in soil carbon. Furthermore, Tang *et al*. (2019) quantified the change in ecosystem services from carbon due to use of fertilizers. Based on a field experiment in China, they reported that the economic value of the regulating services provided by soil carbon showed modest increases with organic inputs (194-333 Yuan/ha/year, or about USD 30-50).

Saving forests

Fertilizer use has contributed, to some extent, to saving forests. According to estimates by Stevenson *et al*. (2013), about 17.9-26.7 million hectares of land under pasture and forests would have been converted to cropland between 1965 and 2004 were it not for the Green Revolution. The avoided loss of ecosystem services provided by forests could have been in the billions of dollars. For example, FAO has estimated that the value of the loss of ecosystem services due to deforestation could be about USD 1,600/ha of forest lost (FAO 2014). This implies that the avoided loss of ecosystem services due to saving of 17.9-26.7 million hectares of pasture and forest land could have been about USD 30-40 billion/year. Furthermore, the avoided loss of carbon emissions could have been substantial. A global study by Burney, Davis and Lobell (2010) on greenhouse gas (GHG) emissions between 1961 and 2005 estimated that yield gains since 1961 resulting from intensification had avoided overall carbon emissions of up to 161 gigatons of carbon (GtC).

10.4 Data gaps and challenges

There are large data gaps with regard to the impacts of fertilizer use. Furthermore, assessments of the global social costs of nutrient pollution are subject to many uncertainties (Sutton *et al*. 2013). In addition, the existing data are not disaggregated by gender, age or fertilizer type (i.e., on organic fertilizers in comparison to inorganic fertilizers).

As pointed out in Chapter 9, there are large data gaps on the environmental effects of fertilizer use and on the impact of fertilizers on human health. This could be one reason why there are very few impact studies. While a key objective of governments is to improve people's health and welfare, their investment priorities may not include monitoring fertilizers' adverse impacts.

Where data exist, it is difficult to compare the values of pollution costs from different studies. Varying assumptions are made, e.g., in monetizing the different species of reactive nitrogen (Kanter, Zhang and Mauzerall 2014). In addition, it is not easy to predict or estimate damage costs as these are influenced by, for example, fertilizer type, management practices and ecosystem characteristics.

For nitrogen the situation is more complex. Costs largely depend on the pathway the nitrogen follows and the different transformations of the nitrogen that occur (Keeler *et al*. 2016). As in this study, a recent systematic review on the health costs of eutrophication by Kouakou and Poder (2019) demonstrates the difficulties of making comparisons among studies or even combining costs reported in different studies.

Using regional data to make global estimates has some drawbacks. According to Sutton *et al*. (2013), the global estimates presented in that report were calculated by extrapolating values estimated for the EU. The authors further explain that the EU estimates were based on citizens' willingness-to-pay in order to reduce health risks, restore damaged ecosystems and reduce GHG emissions, and that such willingness largely depends on income. This demonstrates the need for the collection of region-specific data (i.e., more granular data).

For different forms of nitrogen pollutants, the estimates made depend to some extent on the approach used to arrive at those estimates. For example, Thompson *et al*. (2019) demonstrated that estimates of nitrous oxide emissions can differ in different models. Furthermore, estimates tend to have large uncertainty ranges (Huang *et al*. 2017). This is supported by Keeler et al. (2016), who concluded that nitrogen-related costs depend on where nitrogen moves and the location, vulnerability and preferences of affected populations.

The data gaps highlighted in this chapter are not the only ones which exist. For example, some studies have associated increases in insect pest populations with the use of fertilizers. If these increases do occur, fertilizer use can result in increased use of pesticides. There is also a need to examine leakages along the food chain. Of the 36 per cent of calories produced that are estimated to be fed to livestock, 12 per cent are estimated to reach human beings in the form of animal products (Cassidy *et al*. 2013). About one-third (by weight) of all food produced is lost or wasted (FAO 2011; FAO 2021). While it is important to be aware of the uncertainties of data on food loss and food waste (Lipinski *et al*. 2013; Xue *et al*. 2017), it is crucial to reduce leakages and recycle nutrients in order to meet global sustainability goal

References

Adams, J. and Khan, E. (2020). Medications used for the cardiovascular system. In *Understanding Pharmacology in Nursing Practice*. Hood, P. and Khan, E. (eds.). Cham: Springer, 167-217. https://doi.org/10.1007/978-3-030-32004-1_7.

Alfthan, G., Eurola, M., Ekholm, P., Venalainen, E., Root, T., Korkalainen, K. *et al*. (2015). Effects of nationwide addition of selenium to fertilizers on foods, and animals and human health in Finland: From deficiency to optimal selenium status of the population. *Journal of Trace Elements in Medicine and Biology* 31, 142-147. https://doi.org/10.1016/j.jtemb.2014.04.009.

Anderson, M., Clift, C., Schulze, K., Sagan, A., Nahrgang, S., Ouakrim, D. and Mossialos, E. (2019). *Averting the AMR Crisis: What Are the Avenues for Policy Action for Countries in Europe? Policy Brief 32*. World Health Organization/European Observatory on Health Systems and Policies. https://www.euro.who.int/__data/assets/pdf_file/0005/397652/PolicyBrief_PB32_FINAL_WEB.pdf.

Bationo, A. and Waswa, B. (2011). New challenges and opportunities for integrated soil fertility management in Africa. In *Innovations as Key to the Green Revolution in Africa*. *Exploring the Scientific Facts*. Bationo, A., Waswa, B., Okeyo, J., Maina, F. and Kihara, J. (eds.). Dordrecht: Springer. 3-17. https://doi.org/10.1007/978-90-481-2543-2_1.

Baveye, P., Baveye, J. and Gowdy, J. (2016). Soil "ecosystem" services and natural capital: Critical appraisal of research on uncertain ground. *Frontiers in Environmental Science* 4, 41. https://doi.org/10.3389/fenvs.2016.00041.

Blekkenhorst, L., Bondonno, N., Liu, A., Ward, N., Prince, R., Lewis, J. *et al*. (2018). Nitrate, the oral microbiome, and cardiovascular health: A systematic literature review of human and animal studies. *American Journal of Clinical Nutrition* 107, 504-522. https://doi.org/10.1093/ajcn/nqx046.

Brady, M., Hedlund, K., Cong, R., Hemerik, L., Hotes, S., Machado, S. *et al*. (2015). Valuing supporting soil ecosystem services in agriculture: A natural capital approach. *Agronomy Journal* 107(5), 1809-1821. https://doi.org/10.2134/agronj14.0597.

Bruulsema, T., Heffer, P., Welch, R., Cakmak, I. and Moran, K. (2012). Introduction/Executive Summary. In *Fertilizing Crops to Improve Human Health: A Scientific Review*. International Plant Nutrition Institute (IPNI) and International Fertilizer Industry Association (IFA). 1-9. http://www.ipni.net/article/IPNI-3269.

Budke, C., Deplazes, P. and Torgerson, P. (2006). Global socioeconomic impact of cystic echinococcosis. *Emerging Infectious Diseases* 12(2), 296-303. https://doi.org/10.3201/eid1202.050499.

Burney, J., Davis, S. and Lobell, D. (2010). Greenhouse gas mitigation by agricultural intensification. *Proceedings of the National Academy of Sciences* 107(26), 12052-12057. https://doi.org/10.1073/pnas.0914216107.

Campbell, B.M., Beare, D.J., Bennett, E.M., Hall-Spencer, J.M., Ingram, J.S.I., Jaramillo, F. *et al*. (2017). Agriculture production as a major driver of the Earth system exceeding planetary boundaries. *Ecology and Society* 22(4), 8. https://doi.org/10.5751/ES-09595-220408.

Cassidy, E., West, P., Gerber, J. and Foley, J. (2013). Redefining agricultural yields: From tonnes to people nourished per hectare. *Environmental Research Letters* 8, 034015. https://iopscience.iop.org/article/10.1088/1748-9326/8/3/034015/meta.

Combs, G. (2001). Selenium in global food systems. *British Journal of Nutrition* 85, 517-547. https://doi.org/10.1079/BJN2000280.

de Graaff, M., Hornslein, N., Throop, H., Kardol, P., and van Diepen, L. (2019). Effects of agricultural intensification on soil biodiversity and implications for ecosystem functioning: A meta-analysis. *Advances in Agronomy* 55, 1-44. https://doi.org/10.1016/bs.agron.2019.01.001.

Development Initiatives (2020). *2020 Global Nutrition Report*. https://globalnutritionreport.org/reports/2020-global-nutrition-report/2020-global-nutrition-report-context-covid-19/.

Dodds, W., Bouska, W., Eitzmann, J., Pilger, T., Pitts, K., Riley *et al*. (2009). Eutrophication of U.S. freshwaters: Analysis of potential economic damages. *Environmental Science and Technology* 43(1), 12-19. https://doi.org/10.1021/es801217q.

Donkor, E., Onakuse, S., Bogue, J. and De Los Rios-Carmenado, I. (2019). Fertiliser adoption and sustainable rural livelihood improvement in Nigeria. *Land Use Policy* 88, 104193. https://doi.org/10.1016/j.landusepol.2019.104193.

Elrys, A., Desoky, E., Ali, A., Zhang, J., Cai, Z. and Cheng, Y. (2020). Sub-Saharan Africa's food nitrogen and phosphorus footprints: A scenario analysis for 2050. *Science of The Total Environment* 752, 141964. https://doi.org/10.1016/j.scitotenv.2020.141964.

Erisman, J., Sutton, M., Galloway, J., Klimont, Z. and Winiwarter, W. (2008). How a century of ammonia synthesis changed the world. *Nature Geoscience* 1, 636-639. https://doi.org/10.1038/ngeo325.

Foley, J. (2011). Can we feed the world and sustain the planet? *Scientific American* 61, 60-65. https://www.scientificamerican.com/article/can-we-feed-the-world/.

Food and Agriculture Organization of the United Nations (2011). *Global Food Losses and Food Waste: Extent, Causes and Prevention*. Rome. http://www.fao.org/3/a-i2697e.pdf.

Food and Agriculture Organization of the United Nations (2014). *Food Wastage Footprint: Full cost-accounting*. http://www.fao.org/3/a-i3991e.pdf.

Food and Agriculture Organization of the United Nations (2018a). Pollutants from agriculture a serious threat to world's water, 20 June. http://www.fao.org/news/story/en/item/1141534/icode/. Accessed 28 January 2021.

Food and Agriculture Organization of the United Nations (2018b). *More People, More Food, Less Water?* A *Global Review of Water Pollution from Agriculture*. Rome and Colombo: FAO and the International Water Management Institute (IWMI) on behalf of the Water Land and Ecosystems research program of the CGIAR. http://www.fao.org/3/CA0146EN/ca0146en.pdf.

Food and Agriculture Organization of the United Nations (2021). Food waste and loss database. http://www.fao.org/food-loss-and-food-waste/flw-data. Accessed 27 January 2021.

Food and Agriculture Organization of the United Nations, International Fund for Agricultural Development, United Nations Children's Fund, World Food Programme and World Health Organization (2020). *The State of Food Security and Nutrition in the World 2020*. *Transforming food systems for affordable healthy diets*. Rome: FAO.

Ge, S., Zhu, Z. and Jiang, Y. (2018). Long-term impact of fertilization on soil pH and fertility in an apple production system. *Journal of Soil Science and Plant Nutrition* 18(1), 282-293. http://dx.doi.org/10.4067/S0718-95162018005001002.

Gerten, D., Heck, V., Jagermeyr, J., Bordirsky , B.L., Jalava, M., Kummu, M. *et al*. (2020). Feeding ten billion people is possible within four terrestrial planetary boundaries. *Nature Sustainability* 3, 200-208. https://doi.org/10.1038/s41893-019-0465-1.

Giannadaki, D., Giannakis, E., Pozzer, A. and Lelieveld, J. (2018). Estimating health and economic benefits of reductions in air pollution from agriculture. *Science of The Total Environment* 622-623, 1304-1316. https://doi.org/10.1016/j.scitotenv.2017.12.064.

Gibb, H.J., Barchowsky, A., Bellinger, D., Bolger, P.M., Carrington, C., Havelaar, A.H. *et al*. (2019). Estimates of the 2015 global and regional disease burden from four foodborne metals – arsenic, cadmium, lead and methylmercury. *Environmental Research* 174, 188-194. https://doi.org/10.1016/j.envres.2018.12.062.

Global Burden of Disease Study 2013 Collaborators (2015). Global, regional, and national incidence, prevalence, and years lived with disability for 301 acute and chronic diseases and injuries in 188 countries, 1990-2013: A systematic analysis for the Global Burden of Disease Study 2013. *Lancet 386*, 743-800. https://www.thelancet.com/action/showPdf?pii=S0140-6736%2815%2960692-4.

Global Panel on Agriculture and Food Systems for Nutrition (2016a). *The Cost of Malnutrition*. *Why policy Action is Urgent*. *Technical Brief No*. *3*. *Summary for Decision Makers*. https://www.glopan.org/cost-of-malnutrition/.

Global Panel on Agriculture and Food Systems for Nutrition (2016b). *Food Systems and Diets: Facing the Challenges of the 21st Century*. https://glopan.org/sites/default/files/ForesightReport.pdf.

Guglielmi, G. (2020). Why Beirut's ammonium nitrate blast was so devastating. *Nature*, 10 August. https://www.nature.com/articles/d41586-020-02361-x. Accessed 28 January 2021.

Han, P., Zhang, W., Wang, G., Sun, W. and Huang, Y. (2016). Changes in soil organic carbon in croplands subjected to fertilizer management: A global meta-analysis. *Scientific Reports* 6, 27199. https://doi.org/10.1038/srep27199.

Havelaar, A., Kirk, M., Torgerson, P., Gibb, H., Hald, T., Lake, R. *et al*. (2015). World Health Organization global estimates and regional comparisons of the burden of foodborne disease in 2010. *PLoS Medicine* 12(12), e1001923. https://doi.org/10.1371/journal.pmed.1001923.

Hazell, P. (2009). *The Asian Green Revolution*. IFPRI Discussion Paper 00911. https://www.ifpri.org/publication/asian-green-revolution.

Hill, J., Goodkind, A., Tessum, C., Thakrar, S., Tilman, D., Polasky, S. *et al*. (2019). Air-quality-related health damages of maize. *Nature Sustainability* 2, 397-403. https://doi.org/10.1038/s41893-019-0261-y.

Huang, T., Zhu, X., Zhong, Q., Yun, X., Meng, W., Li, B. *et al*. (2017). Spatial and temporal trends in global emissions of nitrogen oxides from 1960 to 2014. *Environmental Science and Technology* 51(14), 7992-8000. https://doi.org/10.1021/acs.est.7b02235.

Institute of Health Metrics and Evaluation (2017). *Global Burden of Disease Study 2017 (GBD 2017) Data Resources*. *Datasets*. http://ghdx.healthdata.org/gbd-2017.

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (2019). *Global Assessment Report on Biodiversity and Ecosystem Services*. https://ipbes.net/global-assessment.

International Fertilizer Association (2020). *Global Sustainability Report 2019*. https://www.scribd.com/document/460442439/2020-IFA-Sustainability-Report-pdf-pdf.

International Nitrogen Management System (2021). Towards the establishment of an international nitrogen management system. https://www.inms.international. Accessed 28 January 2021.

International Water Management Institute (2006). *Working Wetlands: A New Approach to Balancing Agricultural Development with Environmental Protection*. Water Policy Briefing Issue 21. http://www.iwmi.cgiar.org/Publications/Water_Policy_Briefs/PDF/WPB21.pdf.

Joy, E.J.M., Ahmad, W., Zia, M.H., Kumssa, D.B., Young, S.D., Ander *et al*. (2017). Valuing increased zinc (Zn) fertiliser-use in Pakistan. *Plant and Soil* 411, 139-150. https://doi.org/10.1007/s11104-016-2961-7. Accessed 28 January 2021.

Kanter, D., Zhang, X. and Mauzerall, D. (2014). Reducing nitrogen pollution while decreasing farmers' costs and Increasing fertilizer industry profits. *Journal of Environmental Quality* 44(2), 325-335. https://doi.org/10.2134/jeq2014.04.0173.

Keeler, B., Gourevitch, J., Polasky, S., Isbell, F., Tessum, C., Hill, J. and Marshall, J. (2016). The social costs of nitrogen. *Science Advances* 2(10), e1600219. https://doi.org/10.1126/sciadv.1600219.

Kelly, V. (2005). *Farmers' Demand for Fertilizer in Sub-Saharan Africa*. Staff Paper 2005-20. Michigan State University. https://ageconsearch.umn.edu/record/11612.

Kouakou, C. and Poder, T. (2019). Economic impact of harmful algal blooms on human health: A systematic review. *Journal of Water and Health* 17(4), 499-416. https://doi.org/10.2166/wh.2019.064.

Lal, R. (2014). Societal value of soil carbon. *Journal of Soil and Water Conservation* 69(6), 186A-192A. https://doi.org/10.2489/jswc.69.6.186A.

Leip, A., Ledgard, S., Uwizeye, A., Palhares, J., Aller, M., Amon, B. *et al*. (2019). The value of manure - Manure as co-product in life cycle assessment. *Journal of Environmental Management* 241, 293-304. https://doi.org/10.1016/j.jenvman.2019.03.059.

Lipinski, B., Hanson , C., Lomax, J., Kotinoja, L., Waite, R., and Searchinger, T. (2013). *Reducing Food Loss and Food Waste*. *Working Paper, Installment 2 of Creating a Sustainable Food Future*. Washington DC: World Resources Institute. https://www.wri.org/publication/reducing-food-loss-and-waste.

Lugon-Moulin, N., Ryan, L., Donini, P., and Rossi, L. (2006). Cadmium content of phosphate fertilizers used for tobacco production. *Agronomy for Sustainable Development 26*, 151-155. https://doi.org/10.1051/agro:2006010.

Mace, G.M. (2014). Approaches to defining a planetary boundary for biodiversity. *Global Environmental Change* 28, 289-297. https://doi.org/10.1016/j.gloenvcha.2014.07.009.

Martey, E., Kuwornu, J. and Adjebeng-Danquah. (2019). Estimating the effect of mineral fertilizer use on land productivity and income: Evidence from Ghana. *Land Use Policy* 85, 463-475. https://doi.org/10.1016/j.landusepol.2019.04.027.

Miller, C., Spiehs, M., Arthur, T., Woodbury, B., Cortus, C., Chatterjee, A. *et al*. (2019). Cropland amendment with beef cattle manure minimally affects antimicrobial resistance. *Journal of Environmental Quality* 48, 1683-1693. https://doi.org/10.2134/jeq2019.02.0042.

Molinos-Senante, M., Hernández-Sancho, F., Sala-Garrido, R. and Garrido-Baserba, M. (2011). Economic feasibility study for phosphorus recovery processes. *Ambio 40*, 408-416. https://doi.org/10.1007/s13280-010-0101-9.

Peng, W. and Berry, E. (2018). Global nutrition 1990-2015: A shrinking hungry, and expanding fat world. *PLoS ONE* 13(3), e0194821. https://doi.org/10.1371/journal.pone.0194821.

Pickering, J. and Persson, Å (2020). Democratising planetary boundaries: Experts, social values and deliberative risk evaluation in Earth system governance. *Journal of Environmental Policy and Planning* 22(1), 59-71. https://doi.org/10.1080/1523908X.2019.1661233.

Pizzol, M., Smart, J. and Thomsen, M. (2014). External costs of cadmium emissions to soil: A drawback of phosphorus fertilizers. *Journal of Cleaner Production* 84, 475-483. https://doi.org/10.1016/j.jclepro.2013.12.080.

Pozzer, A., Tsimpidi, A., Karydis, V., de Meij, V. and Lelieveld, J. (2017). Impact of agricultural emission reductions on fine-particulate matter and public health. *Atmospheric Chemistry and Physics* 17, 12813-12826. https://doi.org/10.5194/acp-17-12813-2017.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F., Lambin, E. *et al*. (2009). A safe operating space for humanity. *Nature* 461, 472-475. https://doi.org/10.1038/461472a.

Rurangwa, E., Vanlauwe, B. and Giller, K. (2018). Benefits of inoculation, P fertilizer and manure on yields of common bean and soybean also increase yield of subsequent maize. *Agriculture, Ecosystems and Environment* 261, 219-229. https://doi.org/10.1016/j.agee.2017.08.015.

Salehzadeh, H., Maleki, A., Rezaee, R., Shahmoradi, B. and Ponnet, K. (2020). The nitrate content of fresh and cooked vegetables and their health-related risks. *PLoS ONE* 15(1), e0227551. https://doi.org/10.1371/journal.pone.0227551.

Seachinger, T., Waite, R., Hanson, C., Ranganathan, J., Dumas, P. and Matthews, E. (2019). *Creating a Sustainable Food Future: A Menu of Solutions to Feed Nearly 10 Billion People by 2050*. *Final Report*. World Resources Institute. https://www.wri.org/publication/creating-sustainable-food-future.

Sobota, D., Compton, J., McCrackin, M. and Singh, S. (2015). Cost of reactive nitrogen release from human activities to the environment in the United States. *Environmental Research Letters* 10, 1-13. https://iopscience.iop.org/article/10.1088/1748-9326/10/2/025006/meta.

Steffen, W., Richardson, K., Rockström, J., Cornell, S., Fetzer, I., Bennett, E. *et al*. (2015). Planetary boundaries: Guiding development on a changing planet. *Science* 347(6223), 1259855. https://doi.org/10.1126/science.1259855.

Stevenson, J., Villoria, N., Byerlee, D., Kelley, T. and Maredia, M. (2013). Green Revolution research saved an estimated 18 to 27 million hectares from being brought into agricultural production. *Proceedings of the National Academy of Sciences* 110(21), 8363-8368. https://doi.org/10.1073/pnas.1208065110.

Stewart, W., Dibb, D., Johnston, A. and Smyth, T. (2005). The contribution of commercial fertilizer nutrients to food production. *Agronomy Journal* 97 (1), 1-6. https://doi.org/10.2134/agronj2005.0001.

Stockholm Resilience Centre (2020). Planetary boundaries. https://www.stockholmresilience.org/research/planetary-boundaries.html. Accessed 28 January 2021.

Sutton, M.A., Bleeker, A., Howard, C.M., Bekunda, M., Grizzetti, B., de Vries, W. *et al*. (2013). *Our Nutrient World: The Challenge to Produce More Food and Energy with Less Pollution*. Edinburgh: Centre for Ecology and Hydrology. http://nora.nerc.ac.uk/id/eprint/500700/1/N500700BK.pdf.

Sutton M.A., Howard C.M., Adhya T.K., Baker E., Baron J., Basir A. *et al*. (2019). *Nitrogen: Grasping the Challenge*. *A Manifesto for Science-in-Action through the International Nitrogen Management System*. *Summary Report*. Edinburgh: Centre for Ecology and Hydrology. https://www.inms.international/sites/inms.international/files/Nitrogen%20Grasping%20the%20Challenge%20%28Summary%20Version%29.pdf.

Tang, Q., Ti, C., Xia, L., Xia, Y., Wei, Z. and Yan, X. (2019). Ecosystem services of partial organic substitution for chemical fertilizer in a peri-urban zone in China. *Journal of Cleaner Production* 224, 779-788. https://doi.org/10.1016/j.jclepro.2019.03.201.

Tegtmeier, E., and Duffy, M. (2004). External costs of agricultural production in the United States. *International Journal of Agricultural Sustainability* 2(1), 1-20. https://www.leopold.iastate.edu/files/pubs-and-papers/2004-01-external-costs-agricultural-production-united-states_0.pdf.

Thompson, R., Lassaletta, L., Patra, P., Wilson, C., Wells, K., Gressent, A. *et al*. (2019). Acceleration of global N2O emissions seen from two decades of atmospheric inversion. *Nature Climate Change 9*, 993-998. https://doi.org/10.1038/s41558-019-0613-7.

Troeger, C., Colombara, D., Rao, P., Khalil, I., Brown, A., Brewer, T. *et al*. (2018). Global disability-adjusted life-year estimates of long-term health burden and undernutrition attributable to diarrhoeal diseases in children younger than 5 years. *Lancet Global Health* 6, e255-69. https://doi.org/10.1016/S2214-109X(18)30045-7.

UK Centre for Ecology and Hydrology (2019). We must wake up to devastating impact of nitrogen, say scientists, 23 October. https://www.ceh.ac.uk/news-and-media/news/we-must-wake-devastating-impact-nitrogen-say-scientists. Accessed 20 January 2021.

United Nations Children's Fund (2018). 2018 Global Nutrition Report reveals malnutrition is unacceptably high and affects every country in the world, but there is also an unprecedented opportunity to end it, 28 November. *Press Release*. https://www.unicef.org/press-releases/2018-global-nutrition-report-reveals-malnutrition-unacceptably-high-and-affects. Accessed 20 January 2021.

United Nations Educational. Social and Cultural Organization (2017). *2017 UN World Water Development Report*. *Wastewater: The Untapped Resource*. *Executive Summary*. Perugia, Italy: United Nations World Water Assessment Programme (WWAP). https://unesdoc.unesco.org/ark:/48223/pf0000247552.

United Nations Environment Programme (2022). *Synthesis Report on Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them*. https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

United States Environmental Protection Agency (2015). *Chemical Advisory: Safe Storage, Handling, and Management of Solid Ammonium Nitrate Prills*. US EPA, Occupational Safety and Health Administration (OSHA) and Bureau of Alcohol, Tobacco, Firearms and Explosives (ATF). https://www.epa.gov/sites/production/files/2015-06/documents/an_advisory_6-5-15.pdf.

van Grinsven, H., Holland, M., Jacobsen, B. and Klimont, Z. (2013). Costs and benefits of nitrogen for Europe and Implications for mitigation. *Environmental Science and Technology* 47, 3571-3579. https://doi.org/10.1021/es303804g.

van Grinsven, H., van Dam, J., Lesschen, J., Timmers, M., Velthof, G., and Lassaletta, L. (2018). Reducing external costs of nitrogen pollution by relocation of pig production between regions in the European Union. *Regional Environmental Change 18*, 2403-2415. https://doi.org/10.1007/s10113-018-1335-5.

Vargas, L., Willemen, L. and Hein, L. (2018). Linking planetary boundaries and ecosystem accounting, with an illustration for the Colombian Orinoco river basin. *Regional Environmental Change* 18, 1521-1534. https://doi.org/10.1007/s10113-018-1282-1.

von Blottnitz, H., Rabl, A., Boiadjiev, D., Taylor, T. and Arnold, S. (2006). Damage costs of nitrogen fertilizer in Europe and their internalization. *Journal of Environmental Planning and Management,* 49(3), 413-433. https://doi.org/10.1080/09640560600601587.

Wessells, K. and Brown, K. (2012). Estimating the global prevalence of zinc deficiency: Results Based on zinc availability in national food supplies and the prevalence of stunting. *PLoS ONE* 7(11), e50568. https://doi.org/10.1371/journal.pone.0050568.

Wikoff, D., Thompson, C., Rager, J., Chappell, G., Fitch, S. and Doepker, C. (2018). Benefit-risk analysis for foods (BRAFO): Evaluation of exposure to dietary nitrates. *Food and Chemical Toxicology* 120, 709-723. https://doi.org/10.1016/j.fct.2018.08.031.

Willet, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S. *et al*. (2019). Food in the anthropocene: The EAT–Lancet Commission on healthy diets from sustainable food systems. *The Lancet Commissions* 393(10170), 447-492. https://doi.org/10.1016/S0140-6736(18)31788-4.

World Economic Forum (2017). Three-quarters of antibiotics are used on animals. Here's why that's a major problem, 24 November. https://www.weforum.org/agenda/2017/11/three-quarters-of-antibiotics-are-used-on-animals-heres-why-thats-a-major-problem. Accessed 28 January 2021.

World Health Organization (2002). *The World Health Report 2002 - Reducing Risks, Promoting Healthy Life*. Geneva. https://apps.who.int/iris/handle/10665/42510.

World Health Organization (2020). Malnutrition. https://www.who.int/news-room/fact-sheets/detail/malnutrition. Accessed 26 January 2021.

Xue, L., Liu, G., Parfitt, J., Liu, X., Van Herpen, E., Stenmarck, A. *et al*. (2017). Missing food, missing data? A critical review of global food losses and food waste data. *Environmental Science and Technology* 51(12), 6618-6633. https://doi.org/10.1021/acs.est.7b00401.

Zhang, X., Gu, B., van Grinsven, H., Lam, S., Liang, X., Bai, M., and Chen, D. (2020). Societal benefits of halving agricultural ammonia emissions in China far exceed the abatement costs. *Nature Communications* 11, 4357. https://doi.org/10.1038/s41467-020-18196-z.

Zhao, B. (2016). Facts and lessons related to the explosion accident in Tianjin Port, China. *Natural Hazards* 84, 7070713. https://doi.org/10.1007/s11069-016-2403-0.

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them

Envisioning A Chemical-Safe World

Chapter 11 of 12

Current fertilizer risk reduction and risk management

Contents

1 Global drivers, actors and policies affecting pesticides and fertilizer use

2 Status and trends of pesticide use

3 The regulatory and policy environment for pesticide management

4 Environmental and health effects of pesticide use

5. The environmental, human health and economic impacts of pesticides

6 Current pesticide risk reduction and risk management

7 Status and trends of fertilizer use

8 The regulatory and policy environment of fertilizer management and use

9 Environmental and health effects of fertilizer use

10 The impact of fertilizer use

About ii

11 Current fertilizer risk reduction and risk management 1

11.1 Overview 1

11.2 Assessing current risk reduction 3

11.2.1 Regulatory measures 3
11.2.2 Market-based measures 5
11.2.3 Training and awareness building 7
11.2.4 Research and development 9

11.3 Strengthening fertilizer management: options for actions 14

11.3.1 Strengthening governance of the production, trade and use of fertilizers 14
11.3.2 Strengthening monitoring of fertilizers use and effects 15
11.3.3 Innovating fertilizer management through targeted R&D 15
11.3.4 Informing and educating for change 16
11.3.5 The need for policy coherence 17

12 Transformative actions to minimize the adverse impacts of pesticide and fertilizers

List of Boxes

Box 11.2-1	Fertilizer-related measures in China.	7
Box 11.2-2	Use of sewage sludge in Sweden.	11
Box 11.2-3	Ensuring access to federally funded research in the United States.	12
Box 11.2-4	Examples of international initiatives to make data and publications accessible.	13

List of Tables

Table 11.2-1	Regulatory and policy measures used to reduce the environmental and health risks of fertilizers: strengths, limitations, opportunities and challenges.	5
Table 11.2-2	Market-based measures used to reduce the environmental and health risks of fertilizers: strengths, limitations opportunities and challenges.	6
Table 11.2-3	Educational measures used to reduce the environmental and health risks of fertilizers: strengths, limitations, opportunities and challenges.	10
Table 11.2-4	Research and development measures used to reduce the environmental and health risks of fertilizers: strengths, limitations, opportunities and challenges.	11

About

In December 2017, Resolution 4 of the 3rd Session of the United Nations Environment Assembly (UNEA 3) requested "the Executive Director to present a report on the environmental and health impacts of pesticides and fertilizers and ways of minimizing them, given the lack of data in that regard, in collaboration with the World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO) and other relevant organizations by the fifth session of the United Nations Environment Assembly". In response to this request, UNEP published a *Synthesis Report on the Environmental and Health Impacts of Pesticides and Fertilizers and Ways to Minimize Them*[1] in February 2022 (United Nations Environment Programme [UNEP] 2022).

The overall goal of the synthesis report is to provide the information base to enable other advocacy actions to be taken by stakeholders to minimize the adverse impacts of pesticides and fertilizers. Specific objectives of the synthesis report are to:

- Update understanding of current pesticide and fertilizer use practices;
- Present major environmental and health effects of pesticides and fertilizers, during their life cycle, and identify key knowledge gaps;
- Review current management practices, legislation and policies aimed at reducing risks in the context of the global chemicals, environmental and health agenda;
- Identify opportunities to minimize environmental and health impacts, including proven and innovative approaches.

This chapter on "Current fertilizer risk reduction and risk management" is the 11th in a series of 12 chapters that make up a comprehensive compilation of scientific information. The chapters were developed to both inform and further elaborate on the information provided in the synthesis report. Please note that the disclaimers and copyright from the synthesis report apply.

1 The Synthesis report is available at https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

Current fertilizer risk reduction and risk management

11

11.1 Overview

The content of this chapter is guided by: information compiled in the previous chapters (Chapters 7, 8, 9, and 10); contributions made by experts during a physical meeting of experts held in Geneva, Switzerland, in November 2019; a survey on challenges and opportunities for minimizing risks carried out in 2020; and the published literature. Information from previous chapters and information obtained from experts was used to confirm/identify measures for inclusion in the report. The published literature provides evidence concerning measures that have worked or not worked.

International conventions address fertilizer use indirectly. Their effectiveness in this regard remains undocumented. Furthermore, these conventions mainly address a specific issue while fertilizer use is associated with multiple issues. International conventions, as well as international partnerships that address fertilizer distribution and use, can provide an opportunity or platform for global discussions on sustainable fertilizer use.

International partnerships and initiatives addressing fertilizer distribution and use, such as the Intergovernmental Technical Panel on Soils of the Global Soil Partnership (GSP) (FAO 2019) and the International Nitrogen Management System (INMS) project (INMS 2021), have successfully provided scientific input to international decisions and outputs.

In many cases international conventions and regional agreements form a basis for measures taken at national level. Moreover, international and regional trade agreements have the potential to influence members to use fertilizer responsibly. [Chapter 11.2.1]

Regional agreements on fertilizers are more advanced in developed countries with shared markets (e.g., in the European Union), as are regional agreements on pollution, especially in the case of shared water bodies (e.g., in Europe and North America). Harmonization and implementation of regional fertilizer policies are hampered by a number of factors, including national interests and differences among stakeholders, and are yet to take off in some regions (e.g., Asia and Africa). [Chapter 11.2.1]

Many countries lack policies dedicated to fertilizers or there are gaps in their policies, their policies are fragmented, or other policies conflict with fertilizer policies. Developing countries are the worst performing. Many have rudimentary regulatory frameworks. Constraints on policy development include inadequate scientific evidence, inadequate funding and inadequate staff capacity. Implementation of regulations

is hampered by inadequate staff, inadequate laboratory facilities, lack of funding, the complexity of the problem of pollution from fertilizers, and overlapping mandates among ministries. [Chapter 11.2.1]

Many developing countries subsidize fertilizer, as food security is a key priority. Potential drawbacks of subsidies include an unbalanced supply of nutrients to the soil, inefficient fertilizer use, pressure on national budgets, and interference with the market. Switching to "smart subsidies" has improved their success. Subsidization of technologies has been proposed.

Few countries have environmentally related fertilizer taxes. Taxes have failed to address pollution, partly because they are often designed to raise revenue rather than to encourage sustainable fertilizer use. In addition, they are costly to enforce, and price changes have little or no effect on demand. [Chapter 11.2.2]

The private sector (fertilizer producers and distributors) has continued to promote best practices, for example through generating and promoting customized fertilizer products and participating in knowledge development and dissemination.

Educating and training knowledge disseminators and farmers encourages the adoption of best practices. Knowledge dissemination is sometimes hampered by lack of updated curricula on environmental issues in education programmes, a low and declining number of agricultural students and extension staff (especially in developing countries), reduced funding, weak research-extension-farmers links, and lack of harmonized messages. Non-traditional knowledge disseminators increasingly play an important role in knowledge dissemination.

Use of information and communications technology (ICT) in agriculture has contributed to an increase in the number of young farmers. It could also attract students to agriculture. Youth entering the agriculture sector bring in ICT expertise and cheaper communication technologies.

In many countries, fertilizer recommendations are not based on scientific evidence. Knowledge gaps with regard to adverse impacts of fertilizer use are a strong indication that recommendations do not usually take such impacts into consideration.

In some cases the public is not aware of the impact of fertilizer use or is misinformed. An informed public can make environmentally sound decisions on food choices and participate effectively in environmental conversations. However, actions by the public may not be effective due to lack of interest – or lack of coordination and funding – while laws may not support public participation. [Chapter 11.2.3]

Research is needed to address the large data gaps in the body of knowledge about fertilizer. However, such research is constrained by the decline in the number and funding of researchers, together with lack of coordination and collaboration. Research outputs are not always freely available to the public. Use of open science can improve the reliability of research and increase the number of research outputs. [Chapter 11.2.4]

In general, risk reduction and management require a combination of measures for success: "An efficient and effective policy approach to risk management in agriculture must take into account the interactions and trade-offs between different risks, on-farm strategies, and government policies" (Organisation for Economic Co-operation and Development [OECD] 2018, p. 1). While harmonization of policies at national level improves chances of success, harmonization at regional or even global level can improve these chances further. In the case of some environmental issues (e.g., nitrogen pollution), holistic approaches are likely to be more effective than fragmented ones. Improving efficiencies in the food system will make greater contributions to reducing the negative impact of fertilizer use than improving food production alone.

Actions that can contribute to reduced risks from fertilizer use include strengthening governance of the production, trade and use of fertilizers; strengthening the monitoring of fertilizers' use and effects; and providing support to R&D and extension services. [Chapter 11.3]

Bringing together nitrogen policies, which are currently fragmented among different environmental domains, would increase efficiencies and reduce trade-offs. Joint discussions on policies/initiatives for nitrogen and for phosphorus would potentially improve the efficiency of policies addressing both nutrients.

Sustainable nutrient management requires not only improving fertilizer use practices and technologies, but also making dietary changes towards more plant-based diets and reducing food loss and waste – thus contributing to a more efficient "from industry-to-farm-to-fork" fertilizer chain.

11.2 Assessing current risk reduction

11.2.1 Regulatory measures

International conventions and regional agreements are frequently a basis for measures taken at national level, including fertilizer evaluation, registration and labelling as well as control, enforcement and monitoring (Table 11.2-1). Measures at all levels of decision-making should be based on scientific evidence and the precautionary principle.

International conventions, voluntary international instruments, and partnerships

International conventions do not address fertilizer use directly, but several of them address fertilizer use indirectly. Their effectiveness in reducing the risks of fertilizer use remains unknown (Chapter 8.2). Neither is it clear whether existing international conventions and policy instruments are sufficient to tackle the potential harmful effects of fertilizer. Conventions mostly address a specific issue, while fertilizer use is associated with many issues.

The existence of international partnerships and initiatives addressing fertilizer distribution and use (Chapter 8.3) has helped to ensure the provision of scientific input to international decisions, as demonstrated through the progress made by the International Technical Panel on Soils of the Global Soil Partnership (GSP), hosted by the FAO, in developing outputs such as the International Code of Conduct for the Sustainable Use and Management of Fertilizers (FAO 2019). The Code provides a locally adaptable framework and voluntary set of practices and is targeted at diverse stakeholders. Another example is provided by recent developments regarding nitrogen, in which the International Nitrogen Management System (INMS) project, implemented by the UNEP and executed by the International Nitrogen Institute (INI) and its partners, has continued to play a leading role (INMS 2021).

Bottlenecks exist in the case of many conventions, as well as international partnerships and initiatives, as their success depends on the willingness and capacity of the Parties or members to comply with them. On the other hand, their existence provides an opportunity or platform for global and regional discussions on sustainable fertilizer use.

Regional agreements

In general, regional agreements tend to focus more on trade than on environmental issues (Chapter 8.4). However, trade agreements increasingly include environmental provisions (George 2014) and could therefore contribute towards sustainable nutrient management.

There has been a push to achieve regional harmonization of direct fertilizer policies (e.g., harmonization of quality pricing and sustainable use), but progress has been variable. Harmonization of regional fertilizer policies can benefit both producers and consumers by

opening up markets, reducing trading costs, and encouraging sustainable use of quality fertilizers. The harmonization and implementation of such policies is more advanced in developed countries with shared markets (e.g., in the European Union [EU]) than in developing countries (e.g., in sub-Saharan Africa) (Chapter 8.4).

The existence of regional platforms in many regions offers an opportunity to develop and implement regional agreements on fertilizers. Nevertheless, attempts to harmonize environmental policies, as well as the success of such measures, are hampered by national interests, lack of funds and lack of capacity.

National legislation and regulations

National legislation

Legislation in many countries targets fertilizer use. However, in many other countries legislation dedicated to fertilizers may be lacking; legislation may be present although gaps exist; or legislation may be fragmented (Chapter 8.5).

Bottlenecks in the development of national policies include lack of adequate scientific evidence for policy development, lack of adequate funding, and inadequate staff capacity. Furthermore, some existing legislation may not support sustainable fertilizer use or may even conflict with fertilizer policies that it would be desirable to implement. For example, in developed countries there has been an emphasis on specialization instead of diversification, so that livestock regions are sometimes far from crop producing regions. This can make it difficult to use manure in crop production (e.g., due to high transportation costs). Where land is limited, disposal of such manure also becomes a problem.

Fertilizer evaluation, registration, licensing and labelling

The evaluation and registration of fertilizer products, and the registration of dealers, can help ensure the safety and quality of fertilizers on the market. Labelling can also provide assurance to buyers of the quality and quantity of fertilizer products.

In general, sub-Saharan African (SSA) countries are the worst performing in this respect (Chapter 8.5). In many of these countries there are only rudimentary regulatory frameworks for fertilizer registration; registration time is very long and costs are high; and quality control is poor. In some countries fertilizers are considered new products even when they are made by mixing registered fertilizers. Moreover, in many countries the long and expensive process of registration and licensing discourages small firms from entering the fertilizer industry. Many SSA countries lack personnel and laboratory facilities adequate to implement regulations efficiently. Countries in other regions (Asia and Latin America) also perform poorly or fairly poorly with regard to fertilizer registration, according to the World Bank's Enabling the Business of Agriculture (EBA) study, which collects data on regulations that affect farmers (World Bank 2021a). No reviews or reports were found on the reasons behind this poor performance.

Countries that have not yet been successful in implementing regulatory and policy measures can learn from those that have been more successful. Increasing public participation (e.g., through food labelling) can provide a way to improve the effectiveness and efficiency of these measures. Food labels in most countries include information on nutritional value and expiry dates, but rarely on products' environmental footprint. The European Commission intends to propose that food labels include information concerning climate, environmental and social impacts (European Commission [EC] 2020). Support for such moves could come from bodies or platforms that encourage documentation of the environmental footprint of food production, such as the European Food Sustainable Consumption and Production Round Table (EC 2019) (Chapter 8.4). Key bottlenecks include lack of funding and the unwillingness of governments to put supporting policies in place; lack of quality data; administrative costs; and the variability of the supply chain.

Control, enforcement and monitoring measures

While control, enforcement and monitoring of fertilizer quality is required in many countries, implementation of these measures is often poor

(Table 11.2-1). In the case of manure, enforcement varies across countries and some countries have contradictory policies (Chapter 8.5.2). Monitoring programmes for pollutants are lacking in some countries.

Implementation of these measures is mainly constrained by lack of capacity (e.g., trained staff, laboratories and other facilities, finance) and the complexity of environmental damage caused by fertilizers. In some countries several institutions with overlapping mandates are involved in implementation, which reduces efficiency and effectiveness. Furthermore, it is easier to implement measures that target the industry (e.g., fertilizer distributors and sellers) than those targeting farmers.

As with other measures, experience gained elsewhere can be applied to improve the success of regulations. Greater awareness of fertilizers' health impacts on the part of policymakers, together with public pressure, can influence government decisions. There is also a need to establish food, feed and water monitoring programmes for pollutants that could potentially

Table 11.2-1 Regulatory and policy measures used to reduce the environmental and health risks of fertilizers: strengths, limitations, opportunities and challenges.

Measure	Strengths	Limitations	Opportunities	Challenges
International conventions (Chapter 8.1)	Address transboundary risks, set international standards for nutrient pollution risk reduction	None specific to fertilizers	Existence of international frameworks provides an opportunity for international discussions	Effectiveness mostly depends on national governments' willingness and capacity to comply
Regional agreements (Chapter 8.3)	Address transboundary risks and set regional standards	More focused on trade and less on environmental issues	Existing regional frameworks and trade agreements offer opportunities to implement fertilizer agreements	Effectiveness depends on national governments' willingness and capacity to comply
National fertilizer legislation, and legislation that indirectly addresses fertilizer use (Chapter 8.4)	Provide a legal basis to require risk reduction measures	Lack of legislation that comprehensively addresses risks arising from the use of fertilizers	Synergies among different pieces of legislation can be exploited	Lack of scientific evidence for policy development Unwillingness of governments to develop legislation Policies could be fragmented
Fertilizer evaluation, registration, licensing and labelling (Chapter 8.4)	Most countries require fertilizers to be evaluated before registration and to have labels, which contribute to increased awareness	Personnel and laboratories limited Registration time consuming and expensive Duplication of efforts by regulatory authorities	Existence of efficient and effective measures provides a learning opportunity Increased awareness of environmental issues among stakeholders	Governments may be unwilling to provide adequate resources, or to put supporting policies in place
Control, enforcement and monitoring (Chapter 8.4)	Mostly required for fertilizer before it gets to the farmer Required in many developed countries, but rarely in developing ones	Control, enforcement and monitoring are often inadequate The three measures are sometimes carried out by different authorities	Some experience exists in monitoring ecosystem damage Increased awareness of environmental issues among stakeholders	Governments may be unwilling to provide adequate resources or to reorganize the fertilizer regulatory process

be from fertilizers, and to establish risk assessment goals and methods. Information collected through such programmes can be used to inform decision-makers.

Major drawbacks to the success of control, enforcement and monitoring measures include unwillingness by governments to provide support, and to reorganize and reduce the number of authorities involved.

11.2.2 Market-based measures

Key market-based measures include taxes, subsidies, grants, and commodity price support for harvested produce. This report mainly focuses on subsidies and taxes.

Subsidies

Many developing countries provide fertilizer subsidies to increase food production, improve farmers' incomes and support the growth of the fertilizer industry. However, subsidies have been associated with an unbalanced supply of nutrients to the soil, inefficient fertilizer use, substantial pressure on national budgets, and market distortions (Chapter 7.5.1; Chapter 8.5.4; Table 11.1-2).

Some countries have had more success with subsidies than others, although this could present a learning opportunity for the latter. For example, subsidies became more successful in sub-Saharan Africa when modifications were made to their design. In addition, technologies can be subsidized that improve the efficiency of fertilizers and their use (e.g., precision farming) and encourage recycling of nutrients to address both food demand and environmental issues (Table 11.2-2). For example, in China, government is trying to reduce growth in demand for inorganic fertilizers by phasing out subsidies for inorganic fertilizers and subsidizing the manufacture and use of organic fertilizers (Box 11.2-1). Successful use of subsidies can be hampered not only by the pressure they exert on national budgets, but also by smuggling of subsidized fertilizers to neighbouring countries.

Taxes

Taxes on fertilizers may contribute to sustainable nutrient management and reduced pollution.

Table 11.2-2 Market-based measures used to reduce the environmental and health risks of fertilizers: strengths, limitations opportunities and challenges.

Measure	Strengths	Limitations	Opportunities	Challenges
Subsidies (Chapter 8.4)	Keep fertilizer prices low, increase fertilizer use Improve food security in developing countries Subsidies for proven technologies can reduce nutrient losses	Not always efficient Can distort the market Can encourage imbalanced nutrition	Experience from successful countries can benefit other countries More successful when combined with other policy measures	Funds may be lacking Subsidized fertilizer and produce sometimes sold in neighbouring countries Some subsidy programmes lack an exit strategy International and regional trade agreements discourage use of subsidies Can encourage smuggling of fertilizers
Taxes (Chapter 8.4)	Fertilizer taxes can reduce fertilizer use and pollution. Tax credits (e.g. for organic inputs) can encourage their use	Have had limited success Mostly designed to raise revenues and not to reduce pollution	The funds raised can be used to pay for implementation, e.g. of regulatory measures	Can reduce competitiveness of produce on the regional or international market Can encourage smuggling of fertilizers

Box 11.2-1 Fertilizer-related measures in China.

China is the world's leading consumer of inorganic fertilizer. At the same time, pollution from agriculture in that country is of increasing concern. Nutrient loss from crop and livestock systems has been associated with air pollution and contamination of surface water and groundwater (Ma, Hou and Bai 2020). Agriculture has also been linked to greenhouse gas (GHG) emissions. For example, according to an analysis by Mamun, Martin and Tokgoz (2019) inorganic fertilizer emissions in 2015 accounted for 29.2 per cent of all agriculture-related GHG emissions in China compared to the global average of 13.4 per cent.

In general, crop and livestock systems are decoupled in China, which has led to less recycling of nutrients in animal manure back to land (Ma, Hou and Bai 2020). Furthermore, where farmers grow crops and own livestock, manure production sometimes exceeds nutrient requirements for crop growth (Jin *et al*. 2021). In addition, nutrient use efficiencies have declined (Ma, Hou and Bai 2020).

In the past, government fertilizer policies encouraged fertilizer production, distribution and use, and livestock policies encouraged livestock production (Ma, Hou, and Bai 2020). According to a World Bank report by Searchinger *et al*. (2020), until 2015 China provided subsidies for the production of fertilizers and also imposed limitations on imports. For example, under the Agricultural Input Comprehensive Subsidies programme it spent a total of United States dollars (USD) 16.6 billion in 2014 (OECD 2018, cited in Mamun, Martin and Tokgoz 2019). Fertilizer subsidies likely contributed to excessive use of nitrogen fertilizers, contributing to nutrient pollution in surface and groundwater in agricultural areas (Searchinger *et al*. 2020).

In 2015 the Action Plan for the Zero Increase of Fertilizer Use was implemented (Ji, Liu and Shi 2020). Its key objective was to achieve zero growth of inorganic fertilizer use on major crops by 2020. After the plan was implemented, use of inorganic fertilizers declined while the amount of food produced did not decrease (Ji, Liu and Shi 2020). The lack of yield reduction can be attributed partly to the excessive amounts of fertilizers previously used, as well as to increased promotion of soil testing and use of formulated fertilizers (i.e. fertilizers with a chemical balance formulated to meet local needs) (Shuqin and Fang 2018; Ji, Liu and Shi 2020).

China has currently phased out fertilizer subsidies (Searchinger *et al*. 2020) and, as of 2019, applied a zero export tariff on all fertilizers and potash products (Argus 2018). It has also increased efforts to support pilot programmes targeting nitrogen use efficiency (Searchinger *et al*. 2020). Existing nitrogen use efficiency programmes have been expanded (e.g. for maize) and new ones initiated (e.g. for vegetables) (Searchinger *et al*. 2020).

The government has set a 75 per cent target for manure use by 2020 (Ministry of Agriculture 2017, as cited by Ma, Hou, and Bai 2020). There are also increased efforts to use livestock manure in crop production. For example, in a programme initiated on the substitution of inorganic fertilizers by organic ones, subsidies are provided to organic fertilizer manufacturers to increase affordability and uptake (Searchinger *et al*. 2020).

Additional measures include providing tax holidays for the producers of organic fertilizers; supporting entities for the collection, storage and transportation of raw materials of organic fertilizer; implementing a subsidy policy on organic fertilizer usage; and providing subsidies for equipment used in the production and use of spread organic fertilizer (SOHU 2018). According to a review by Zhang *et al*. (2020), in China partial substitution of inorganic fertilizers by manure can increase crop yields and, depending on conditions, can decrease ammonia and nitrous oxide emissions.

The government has also scaled up programmes to improve water use efficiency and soil quality (Cassou 2018). Suggestions have been made for additional actions, such as improving linkages between livestock and croplands at a regional scale (Jin *et al*. 2020) and between farmers and R&D (Searchinger *et al*. 2020).

They can also result in reduced crop production (Chapter 8.5.4; Table 11.2-2). However, few countries have environmentally related fertilizer taxes and, where these taxes have been used, they have not been very successful.

Lack of success with taxes on fertilizers is partly due to their often being designed primarily to raise revenue rather than to curb nutrient pollution. They also tend to have low elasticities (i.e., price changes have little or no effect on demand), are sometimes set too high, are costly to enforce, and may disadvantage a country on the international market. Ploughing agricultural tax revenues back into the agricultural sector can increase the political acceptability of this tax (Sud 2020).

11.2.3 Training and awareness building

Awareness of the of the adverse impacts of fertilizers and ways to minimize these impacts differs among knowledge disseminators, farmers and other stakeholders. Measures that encourage adoption of best practices include effective education and training programmes for knowledge disseminators and farmers; effective extension systems; and ensuring that adoption of best practices will improve farmers' economic welfare (Chapter 7.5.1). No literature was found on improper use of fertilizer and gender at the time the report was being drafted, however the research focus on this topic was not exhaustive.

Education and training programmes

A key objective of agriculture-related education programmes is to equip knowledge disseminators with appropriate knowledge and tools. These programmes sometimes have limitations. Not all of them have an up to date curriculum addressing environmental issues. This may be especially true in developing countries, where the focus has largely been on food security and poverty reduction whereas in developed countries the focus has included surplus production and environmental degradation (Suvedi and Kaplowitz 2016). However, there have been efforts in many universities globally to ensure that the Sustainable Development Goals (SDGs) and sustainability are included in teaching programmes (Filho *et al*. 2019).

There is need in many countries to promote ecological agriculture curricula and courses and to support updating of the curricula of fertilizer-related training programmes to include the potential risks of fertilizers and the benefits of sustainable nutrient management. There is also a need for regular updating of the curricula for both formal and informal training to correspond to changing priorities and emerging research evidence. In addition, in developing countries there has been a decline in the number of students taking agricultural courses since they prefer careers that are more financially rewarding and see little prospect of earning a good living in agricultural professions.

The existence of national agricultural education and national accreditation boards provides an opportunity for external oversight of the education curriculum. Including environmental experts in such bodies helps to ensure that environmental issues are not neglected and that curricula are kept up to date. Increased use of information and communications technology (ICT) in agriculture has contributed to an increase in the number of young farmers and may also contribute to an increase in the number of agricultural students. Nevertheless, the issue of funding will need to be addressed.

Extension services and knowledge dissemination

Extension services convey knowledge to farmers and train them on new practices. Extension staff In many countries are well trained. However, there has been a decline over the years in the number of public extension staff, along with high staff turn-over in some countries, as reported in Zambia by Somanje, Crespo and Zinyengere (2017) and in Ethiopia by Bitzer (2016). This has made extension coverage worse, especially in developing countries due to the large number of smallholder farmers in those countries. In addition, in some countries funding has declined and links between extension, research and farmers are sometimes weak. Furthermore, where there is more than one service provider, extension messages from different providers are sometimes not harmonized.

Apart from government sponsored extension services, non-traditional knowledge disseminators (e.g., the fertilizer industry, non-governmental organizations [NGOs], research organizations, service providers, and farmer groups and cooperatives) have proved effective in many parts of the world (Mittal, Padmaja and Ajay 2018; Ingram and Mills 2019). A recent study on changes in agricultural extension by Norton and Alwang (2020) reported that knowledge transfer has increasingly shifted from the public sector to include private advisory services, although more so in developed than in developing countries. It associates that shift with agricultural commercialization.

Currently in developing countries there are efforts to empower (jobless) youth to become knowledge disseminators, for example in sheep farming in Ethiopia (Wamatu and Ephrem 2020). Involving youth brings in knowledge of ICT and can help make it possible for a greater number of people to use cheap communication technologies.

Knowledge dissemination can be improved through strengthening the training of agronomists, extension agents, input distributors and other knowledge disseminators on new technologies. Since adulteration of fertilizer is an issue in some countries (Chapter 7.4.6), training farmers how to identify counterfeit and adulterated fertilizers (where such information exists) can contribute to efficient fertilizer use.

Adoption of practices and technologies by farmers

According to a survey of experts carried out for this report, in developing countries many fertilizer recommendations are not based on scientific research. Moreover, the information gap with regard to the adverse environmental and human health effects of fertilizers (Chapters 9 and 10) is a strong indication that recommendations do not usually take these adverse effects into consideration. Knowledge about fertilizer use and its potential impact also varies among farmers. For example, many farmers need to be trained on (or made aware of) fertilizer options that have proven successful (e.g., use of neem-coated fertilizers, precision farming, precision conservation, nutrient recycling, mixed cropping, the 4R approach and manure management) (Chapters 7.2.6 and 7.5.1). Adoption of such options can be enhanced through participatory and experience-based learning, with specific attention given to building the capacity of youth and women. Farmers may be unwilling to adopt such measures due to cost implications. Subsidizing sustainable practices is likely to encourage farmers' adoption of environmental measures.

Public awareness and action

Most fertilizer-related conversations are between policymakers, researchers, knowledge disseminators, fertilizer users and the industry (Chapters 7 and 8). Since fertilizer use can have negative social impacts (Chapter 10), including relevant NGOs in decision-making can improve the chances of success of fertilizer policies. For example, involving the public in nutrient recycling research could reduce consumer aversion to food grown with sewage. Furthermore, participation of environmental civil societies, for example environmental non-governmental organisations (ENGOs) in such conversations has been associated with reduced emissions, for example carbon emissions (Grand and Vasi 2017).

An informed public can push for development and implementation of environmental policies (Table 11.2-3). However, lack of adequate knowledge by the public sometimes discourages the adoption, for example, it may reject food grown with recycled nutrients (Box 11.2-2).

Impediments to the public's contribution to reduced nutrient pollution include the fact that the issue may not be a priority for the public, that public activities may not be well coordinated, and that such activities are rarely funded. Furthermore, the public may be unwilling to act, especially where proposed changes affect their lives. For example changing dietary habits has been proposed as one way to reduce fertilizer use (Chapter 7.5.2), but some people are unwilling to make such changes despite their awareness of adverse impacts (Macdiarmid, Douglas and Campbell 2016).

To improve the chances of success of stakeholder participation, lessons learned from successful public actions can be applied and existing civil society organizations can serve as platforms to push for government action. Challenges to public actions include lack of support for such actions in national laws, and the fact that donors may be unwilling to fund such actions.

11.2.4 Research and development

Research and development (R&D) can influence risks through generating new knowledge. Evidence-based policies and interventions/ recommendations are key to maximizing the benefits of fertilizers and reducing their risks.

Table 11.2-3 Educational measures used to reduce the environmental and health risks of fertilizers: strengths, limitations, opportunities and challenges.

Measure	Strengths	Limitations	Opportunities	Challenges
Education, training programmes	Train knowledge disseminators on best practices for healthy environment	Education curricula are sometimes outdated Declining numbers of agricultural students, especially in developing countries	Existence of committees of environmental experts and national accreditation boards, provides an opportunity for curriculum oversight Use of ICT in agriculture encourages (and is encouraged by) youth interest	Reduction of training budget by governments
Extension services	Convey messages on sustainable fertilizer use and management to farmers Extensive coverage Qualified staff	Decline in numbers of extension staff Inadequate training Inadequate funding Weak links with research and farmers Lack of harmonized messages among extension actors	Increased use of ICT Increased collaboration, e.g. public-private partnerships Multiple extension service providers present Jobless youth in some countries are trained as extension agents	Government and donors may be unwilling to invest more in extension services
Application of environmental measures by farmers	Adoption can contribute to healthy environment	Many recommendations are not based on scientific research Farmers lack adequate knowledge about fertilizer use and risks Farmers are primarily interested in economic benefits and not environmental issues Resistance to change Farmers are too numerous to be effectively trained and monitored	Increasing environmental awareness among both farmers and consumers Existence of trade agreements and voluntary standards that aim to reduce environmental damage	Global competition among producers Lack of an enabling environment
Public awareness and action	The public can push for development and implementation of environmental policies Society is interested in environmental issues	Public awareness does not necessarily translate to cleaner environment Lack of coordinated action by public Unwillingness to change	Experience of successful public actions can be applied Existence of civil society organizations for environmental issues	National laws may not support public action Funding may be lacking

Generation of new knowledge and technologies

As demonstrated in earlier chapters, there are large information gaps in the body of knowledge about fertilizer use and management. For example, many studies have been carried out on the agronomic and economic benefits of fertilizer use, a fair number on the environmental effects of nutrient pollution, and a few on health benefits, but studies exploring all these areas together are lacking (Chapter 10). Furthermore, there is a need to develop and improve recommendations. Where it is necessary to address all parameters in the same recommendation, this can be challenging.

Box 11.2-2 Use of sewage sludge in Sweden. Kirchmann *et al.* (2017).

In Sweden measures to control metal emissions in sewage sludge have been very successful. For example, concentrations of heavy metals in municipal sewage sludge decreased by up to 90 per cent between the 1970s and 2010. This has resulted in reduced heavy metal concentrations in the soil in general, and in increased soil microbial biomass.

However, only about 20 per cent of the total amount of organic water produced is recycled. Reasons for this lack of use range from resistance among industries and consumers to products grown on land treated with sewage sludge, to that associated with bulkiness and concerns about the profitability of applying the sludge to land. Extracting nutrients from waste to produce concentrated fertilizers, instead of recycling wastes directly to soil, could encourage nutrient recycling. However, it calls for new extraction technologies.

Research activities have continued to be affected by shifts in research priorities, the decline in the number of researchers, and a decline in funding in some regions (Table 11.2-4). In the United States funding of agricultural research by the public sector fell from the early 2000s, while private sector funding increased (United States Department of Agriculture [USDA] 2019). The role of the private sector in agricultural research, including through funding, has continued to expand (Fuglie 2016). In developing countries research is underfunded, although in some countries (e.g., China, a key middle income country) funding has continued to increase (Beintema and Echeverría 2020). Research efforts have sometimes been unbalanced; for example, studies on plant biomass have tended to focus on above ground biomass and ignore root biomass (Chapter 7.5.2). Furthermore, research efforts in the past were uncoordinated, but today there is increased collaboration among donors, scientists and other stakeholders.

To contribute to closing data and technology gaps, there is a need to support the formation of open data inventories (e.g., for scientific publications) at international, regional and national levels to make data freely accessible to scientists and other users. Sharing data among scientists, for example through Open Science, can speed up the development of research outputs and minimize costs, although some scientists are unwilling to share their data (Wolkovich, Regetz and O'Connor 2012; Haddaway 2018). Some countries, such as the United States (Box 11.2-3), as well as organizations and institutions (Box 11.2-4), are making efforts to improve public access to publications and data generated by scientists. To fill knowledge gaps,

Table 11.2-4 Research and development measures used to reduce the environmental and health risks of fertilizers: strengths, limitations, opportunities and challenges.

Measure	Strengths	Limitations	Opportunities	Challenges
Generation of new knowledge	Generates knowledge on sustainable fertilizer use and management Interdisciplinary approach	Decline in number of national research staff, lack of adequate funding Research is sometimes uncoordinated and unbalanced	Existing collaborations can be involved in research Donors are increasingly pushing for data sharing to speed up production of quality scientific outputs Demand for research outputs Existence of supportive polices	Research is costly and funds are sometimes lacking Donor fatigue Research priorities may differ between a country and donor

Measure	Strengths	Limitations	Opportunities	Challenges
Transfer of research findings to knowledge disseminators	Allows application of research findings	Funds for knowledge transfer not always included in project budget Researchers more interested in data publishing than dissemination Language used by scientists not easily understood by non-scientists	Existing collaborations. e.g. the Global Partnership on Nutrient Management (GPNM), regional soil partnerships and INI, can be used to disseminate knowledge Increased use of ICT	Donors are sometimes unwilling to fund End user priorities can change Lack of consensus among scientists sometimes leads to conflicting messages

Box 11.2-3 Ensuring access to federally funded research in the United States.

It is estimated that between 2015 and 2017 the United States Government spent an average of USD 135 billion per year on scientific research and development across various disciplines, including medical sciences, public health and agriculture (United States Government Accountability Office [GAO] 2019). Efforts within the government to ensure that scientific papers and publications resulting from federally funded research are freely and easily accessible have grown in recent decades. For example, the Congress passed legislation in 2008 (United States Congress 2008) mandating that research papers funded by the National Institutes of Health (NIH) be made publicly available, at no charge, no later than one year after publication through the National Library of Medicine's PubMed portal (United States National Library of Medicine 2021).

With the goal of expanding these earlier legislative efforts, in 2013 the Office of Science and Technology Policy (OSTP) issued a memorandum that directed federal agencies with over USD 100 million in annual research expenditures to develop a plan to ensure public access to products from federally funded research within 12 months of publication of a given article (United States Office of Science and Technology Policy [OSTP] 2013). By 2017, the 19 federal agencies that met the annual expenditure requirement in the 2013 memorandum had developed the requested access plans (The White House, United States 2016). For example, as part of the Department of Agriculture's plan to increase access to federally funded research the National Agricultural Library developed PubAg, an online search portal that provides access to over 300,000 scientific journal articles focusing on agricultural research (USDA 2021). Similarly, the United States Geological Survey (USGS) launched the USGS Publications Warehouse, which currently provides access to over 160,000 publications written by USGS scientists (United States Geological Survey 2021). Along with these efforts by federal agencies, there have been bipartisan attempts by members of Congress to enact companion legislation, such as the Fair Access to Science and Technology Research Act (FASTR) (United States Senate 2017), with the same goals as the 2013 OSTP memorandum but better accounting for publications produced by extramural researchers working outside the federal government.

A 2019 analysis of the public access plans by the GAO found that there had been overall progress across the government in increasing public access to federally funded research, but that some additional steps could be taken by agencies (GAO 2019). The GAO made 37 recommendations to the 19 agencies about ways they could comply more fully with their obligations under the 2013 OSTP memorandum. For example, it recommended that USDA complete development of training for employees involved in reviewing the merits of researchers' proposed data management plans. Agencies by and large concurred with the GAO's recommendations and committed to working during the next few years to address the specific recommendations. OSTP is currently reviewing public comments on a Request for Information (RFI) issued in February 2020 (OSTP 2020) on several topics related to furthering public access to federally funded scientific research.

These efforts to ensure easy and free access to federally funded research mirror efforts in Europe to require that State-funded research products are published in an Open Access format. The Plan S initiative, launched in 2018 by cOAlition S, is a consortium of national research agencies and funders from 12 European countries (Plan S 2021).

Box 11.2-4 Examples of international initiatives to make data and publications accessible.

Scientists around the world publish thousands of articles every year in journals that cover research from various disciplines. Access to most of these scientific journals by scientists looking to publish their research, or anyone who wants to read the articles, may require the payment of fees that can sometimes be very costly. While there have been recent efforts by many publishers to make articles Open Access (which means anyone can download material), this service relies on authors of the paper paying even higher submission and publishing fees. Therefore, obtaining access to high-quality, cutting-edge scientific research is a problem not only for members of the scientific community, but also for policymakers who need to be able to read reliable, evidence-based, peer-reviewed scientific papers to inform their law-making efforts.

Open access databases include organization databases with data accessible to all, such as those of the FAO (FAOSTAT 2021) and the World Bank (World Bank 2021b), and initiatives such as Global Open Data for Agriculture and Nutrition (GODAN), which focuses on building high-level policy, and public and private institutional support for open data (Global Open Data for Agriculture and Nutrition [GODAN] 2021).

Initiatives exist that aim to make scientific publications more accessible – in terms of cost and technology – to institutions and individuals, with some efforts placing emphasis on ensuring access to those in low income countries. For example, Google Scholar is not only a search engine for scientific literature from journals and libraries around the world, but also provides users with copies of papers if they are available. Research4Life Programmes provides low and middle income countries with access to scientific and professional resources through the following programmes:

- AGORA (Access to Global Online Research in Agriculture), a partnership between FAO and Cornell University in the United States, along with international publishers (AIMS 2020);
- ARDI (Access to Research for Development and Innovation), coordinated by the World Intellectual Property Organization (WIPO) and international publishers (World Intellectual Property Organization 2020);
- GOALI (Research for Global Justice), managed by the International Labour Organization (ILO) in collaboration with the Cornell Law School Library and the Lillian Goldman Law Library at Yale Law School, both in the United States (International Labour Organization 2021).
- Hinari Research for Health, managed by the World Health Organization (WHO) in partnership with the Yale University Library in the United States and international publishers (World Health Organization 2021);
- OARE (Online Access to Research in the Environment), managed by the United Nations Environment Programme (UNEP) in partnership with the Yale University Library and international publishers (UNEP 2021);

In addition, academic social networks like ResearchGate allow researchers to share their publications with others interested in their research (ResearchGate 2021). There are concerns about the potential for such sites to violate intellectual property laws in various countries. Nevertheless, these sources generally not only provide users with access to some publications posted by the authors, but also make it possible to communicate with them directly.

providing access to information can be coupled with regular systematic reviews and synthesis of existing knowledge.

In addition to data gaps on the impacts of fertilizer use, research efforts can include approaches to the development of fertilizer recommendations that are simple to use and do not require too much data; breeding and screening for improved nutrient use efficiency; and improved nutrient recycling technologies.

Transferring knowledge from research

Ideally, research findings should be communicated to end users, but this does not always occur in practice.

Shifting from the traditional linear research approach (in which scientists generate new knowledge and then hand it over to end users or for knowledge dissemination) to working with stakeholders on knowledge generation has had some success. This change in approach has partly been due to pressure from donors who insist that research projects have an impact. However, it, too, is challenged by lack of funding.

The success of R&D can be improved by enhancing collaboration between stakeholders (e.g., strengthening public agricultural R&D systems; supporting collaboration between R&D systems, the public and private sector, and farmers; supporting the development and dissemination of targeted innovations for the use of different types of farmers; and promoting farmer characterization and using such information in technology dissemination). International partnerships and initiatives providing support to knowledge transfer efforts include the Global Partnership on Nutrient Management (GPNM) (UNEP n.d.) and the Sustainable Agriculture Initiative (SAI) Platform (Sustainable Agriculture Initiative [SAI] Platform 2021) (Chapter 8.3).

The use of scientific knowledge can also be improved by strengthening science-policy interfaces at international, regional and national levels; having communication strategies in place at all levels; and designing and developing responsive, audience-oriented knowledge products that target different audiences (e.g., policymakers, extension agents, farmers) and incorporate feedback loops.

11.3 Strengthening fertilizer management: options for actions

In this section incremental actions are proposed that can strengthen existing risk reduction and risk management measures for fertilizers.

In Chapter 12, transformative actions are proposed. These actions would more fundamentally change the way nutrients and fertilizers are managed.

As discussed in the previous sections of this chapter and in Chapters 7, 8, 9, and 10, in many countries there are already measures promoting sustainable use and management of fertilizers. However, the degree of implementation and the success of these measures vary. In addition, there are countries that do not have the relevant policies or in which there are policy gaps. This section sets out some of the actions that have already been undertaken, but may require incremental adjustments to achieve sustainable fertilizer use and management.

11.3.1 Strengthening governance of the production, trade and use of fertilizers

Update national fertilizer legislation to include all elements of the fertilizer life cycle

In many countries there is need to expand fertilizer policies to include not just inorganic fertilizers, but also organic fertilizers, as well as all elements of the fertilizer life cycle. Fertilizers are subject to various types of legislation and regulations related to production, trade, distribution, marketing, safety and use, which can vary among, or within, countries (FAO 2019a) (see Box 8.5-1). Reviewing and updating policies to address gaps is essential for sustainable nutrient management. There is a need, subsequently, to develop, review and update legislation in line with the expanded fertilizer policies.

Fertilizers can be directly regulated by specific fertilizer legislation (addressing, for example, the registration/authorization of fertilizers on the market and their quality, labelling and packaging) or can be addressed in general agrochemicals or agricultural inputs legislation. There are also other laws that may influence the regulatory framework for fertilizers and introduce specific rules or restrictions (see Box 8.5-1).

Enhance enforcement capacity to bolster effective implementation of national legislation

Where monitoring is hampered by lack of trained personnel and equipped laboratories, enhancing enforcement capacity can bolster effective implementation of national policies and related legislation. Strengthening implementation of fertilizer quality inspection and control systems, for example by setting up of a spot check system, training staff and updating analytical equipment, is also necessary.

Establish or strengthen regional collaboration on fertilizer management and, where relevant, regionally harmonize fertilizer legislation

Regional collaboration on fertilizer management does not exist in some regions or, if it does, collaboration may not be effective. This is true even in regions with cross-border trade of fertilizers and shared water bodies. Yet such collaboration would make it easier to control the quality of fertilizer used in regions and make fertilizer registered in one country readily acceptable in other countries. Hence, establishing or strengthening regional collaborations is beneficial for all countries in terms of the environment and human health, economic costs and time (see chapter 8).

Strengthen national collaboration on policies and programmes

National policies and programmes for sustainable use and management of fertilizer are sometimes in conflict with, for example, policies and programmes targeting food security and economic welfare. Harmonizing such policies and programmes (e.g., between the agricultural sector and related sectors such as water, energy and health) can reduce or eliminate incoherence and create synergies. Furthermore, establishing and supporting interdisciplinary platforms for stakeholder engagement can strengthen coordination, implementation and monitoring of fertilizer policies

Promote mainstreaming of fertilizer policies in relevant international and regional agreements

Although many international/regional agreements and agendas (e.g., on food quality) include sustainable development among their guiding principles, minimizing nutrient pollution is not an objective of some of them. Mainstreaming fertilizer policies in such agreements and agendas can contribute to increased stakeholder actions on sustainable nutrient management.

Ensuring the availability of suitable and affordable fertilizers

The amount of fertilizer used per unit of land in some low and middle income countries is low. It is particularly minimal (averaging below 50 kg/hectare/year) in sub-Saharan Africa, where yields are low and many people do not have access to sufficient and nutritious food. A major reason for lack of access to affordable fertilizer is that fertilizer prices there are higher than in other regions. Provision of affordable fertilizer in all world regions should be accompanied by controls (e.g., on quality and the use of suitable practices and technologies to minimize adverse impacts).

11.3.2 Strengthening monitoring of fertilizers use and effects

Collect statistics on manufacturing, importation, sales, use and storage of fertilizers

Not all countries have reliable and readily available national data on fertilizer manufacturing, importation, trade, use and bulk storage (in particular, for hazardous fertilizers such as ammonium nitrate). Yet data on manufacturing, importation, sales and use are crucial for national planning and budgeting, while hazardous fertilizers need to be stored safely. Ensuring the collection and sharing of such data should contribute to improved decision-making.

Establish monitoring programmes on pollutants that could potentially be from fertilizers

Countries in which there are no national monitoring programmes for key pollutants that could potentially be from fertilizers should consider

setting up such programmes, as monitoring data help to identify problems and determine whether control measures are working. Setting up national monitoring programmes for such pollutants in food, feed and drinking water, as well as in coastal waters and in other environmental matrices is becoming increasingly urgent due to intensification. These programmes should consider all key potential sources of such pollutants.

Ensure feedback into policy and decision-making

There is need to set up national reporting systems to ensure that information generated on the fertilizer life cycle informs decision-making concerning the authorization and use of fertilizers. This information could also be shared and used at both regional and international levels.

11.3.3 Innovating fertilizer management through targeted R&D

Promote multi-stakeholder innovation

Collaborative partnerships are essential for efficient and effective research. Developing fertilizer recommendation tools and fertilizer question and answer (Q&A) platforms, in collaboration with fertilizer manufacturers, other service providers and end users, can enhance the adoption of innovations and knowledge.

Strengthening public agricultural research and development systems through capacity building and funding, and supporting related collaborative public-private R&D partnerships, should be considered in countries where the number of public researchers is limited and public research activities are underfunded.

Assess costs and benefits

The large data gaps described in this report demonstrate that there is a need for increased research efforts, especially on the impacts of fertilizer use but also on sustainable management options for various agroecological conditions. Including key economic variables associated with the use of fertilizers in such assessments would provide comprehensive evidence concerning the environmental and human health effects of fertilizers; provide information about the actual status of the problem of nutrient pollution; and contribute to innovations in nutrient management.

Regularly review existing knowledge

To increase accessibility of information, countries should consider providing support to the formation of open data inventories (e.g., for scientific publications and research data) at national, regional and international levels; ensure the compilation of fertilizer statistics; and establish innovative approaches for regular systematic reviews of existing knowledge. This would improve the usefulness of research information in decision-making.

11.3.4 Informing and educating for change

Ensure provision of information to farmers and other stakeholders

Scientifically sound decision-making should be promoted at agroecosystem and regional level. Related information should be made available to all stakeholders in appropriate forms.

Countries should ensure that farmers have access to and exchange information about new technologies for fertilizer use and nutrient management. Where necessary (in sub-Saharan Africa, for example, where literacy rates are low), countries should promote participatory and experience-based learning to ensure long-term adoption of sustainable practices and technologies, with specific attention given to building the capacity of youth and women.

Use of ICT and social media is increasing even in low and middle income countries, although the increase is less for women and the rural population. Making better use of these technologies to provide information and complement education and training, while ensuring that marginalized groups are included, is a cost-effective way to support communication.

Include sustainable nutrient management in educational curricula

Countries should promote ecological agriculture courses in schools, universities and other relevant training entities in order to close the gap between the training offered and the development agenda. Supporting the updating of the curricula of fertilizer-related training programmes to include the potential risks of fertilizers and the benefits of sustainable nutrient management, integrated nutrient management, and environmentally friendly nutrient management technologies and practices can ensure that students are better equipped to guide farmers in sustainable nutrient management.

Train farmers on how to identify adulterated fertilizers

Countries where counterfeit and adulterated fertilizers are common should consider training farmers to identify such fertilizers, where such information exists. That knowledge will allow farmers to have more confidence in the quality of the fertilizer they purchase and discourage overapplication. Training should be accompanied by putting in place a system for reporting such malpractice.

Strengthen extension services

Dissemination of information to farmers, especially in middle and low income countries, is hampered by lack of adequate extension capacity. Countries should consider strengthening public agricultural extension systems, and other independent advisory services, and coordinating advisory services.

Support development of national, regional and global communication strategies

There is need to develop global, regional and national communication strategies to allow effective communication of new information on fertilizer use and sustainable nutrient management to all relevant stakeholders. Support can also be provided through the design and development of responsive knowledge products targeting different audiences (e.g., policymakers, extension agents and farmers) and incorporating feedback loops.

11.3.5 The need for policy coherence

Conventions

Many international and regional initiatives are already contributing (or have the potential to contribute) to the reduction of pollution from fertilizers (Chapter 8.2-8.4). Increased collaboration among these initiatives would also contribute to reducing pollution. Conventions and policy instruments could work better together to address pollution from fertilizers by exploiting the synergies among them and with their partners. The implementation plan for the UNEP-led Towards a Pollution-Free Planet initiative (UNEP 2017) demonstrates how this can be done. It provides an overarching framework to address pollution in various dimensions (i.e., air, water, land/soil, marine and coastal pollution, and the cross-cutting issues of chemicals and waste), with Implementation to be achieved through collaborative efforts among different frameworks and players.

Nitrogen policies

Nitrogen policies and the SDGs are fragmented among different environmental domains (e.g., air pollution, freshwater and marine environments, health and food security). This fragmentation, which is also present in Multilateral Environmental Agreements (MEAs) (Sutton, Raghuram and Adhya 2019), has resulted in policy trade-offs. Nitrogen policies should therefore be brought together to take advantage of synergies among them, so as to speed up progress in addressing pollution. In addition, there is a need for an integrated approach in which nitrogen and phosphorus are considered jointly. Oenema (2019) suggests that including phosphorus initiatives in nitrogen discussions could be useful in the case of both nutrients. For example, both nitrogen and phosphorus contribute to pollution in the aquatic environment (Chapter 9.3.3) and can be recycled together (Chapter 7.2). A study by Kanter and Brownlie (2019) of 15 countries representing 75 per cent of both GHG emissions and nitrogen and phosphorus consumption concluded that a joint approach could be beneficial for all the national climate plans analysed and for the achievement of seven of the 17 SDGs. The authors also reported that

integration between science, policies and practice on this issue was lacking (as of 2018). Gu *et al*. (2018) argued that there it is crucial to take carbon sinks (which are necessary to reduce atmospheric carbon dioxide [CO_2] concentrations) into consideration when addressing nitrogen pollution, as large reductions of the latter may reduce the size of carbon sinks. They also suggested that this relationship should be included in global climate change modelling.

Efforts are already under way to bring together Multilateral Environmental Agreements (MEAs) with the potential to influence nitrogen pollution. For example, the Fourth Meeting of the International Nitrogen Management System (INMS-4) endorsed the proposal in the UNEA-4 nitrogen resolution to establish an Inter-convention Nitrogen Coordination Mechanism (Chapter 8.3.4) and identified the next steps (INMS 2019).

References

AIMS (2020). AGORA (Access to Global Online Research in Agriculture). http://aims.fao.org/agora_r4l. Accessed 26 January 2021.

Argus (2018). China announces fertilizer tariffs for 2019, 24 December. https://www.argusmedia.com/en/news/1817027-china-announces-fertilizer-tariffs-for-2019. Accessed 26 January 2021.

Beintema, N. and Echeverría, R. (2020). *Evolution of CGIAR funding*. ASTI Program Note. https://asti.cgiar.org/sites/default/files/pdf/asti20/CGIAR-funding-ASTI-note.pdf.

Bitzer, V. (2016). *Incentives for Enhanced Performance of Agricultural Extension Systems*. KIT Working Papers 2016-6. https://www.kit.nl/wp-content/uploads/2018/08/Incentives-for-enhanced-performance-of-agricultural-extension-systems.pdf.

Cassou, E. (2018). *The Greening of Farm Support Programs: International Experiences with Agricultural Subsidy Reform*. Washington, D.C.: World Bank. http://documents1.worldbank.org/curated/en/827371554284501204/pdf/The-Greening-of-Farm-Support-Programs-International-Experiences-with-Agricultural-Subsidy-Reform.pdf.

European Commission (2019). *The European Food Sustainable Consumption and Production Round Table Working Group 1 Workshop - Towards the ENVIFOOD Protocol: Deriving Scientifically-Sound Rules from Existing Methodological Alternatives – Summary Report*. https://ec.europa.eu/jrc/en/publication/books/european-food-sustainable-consumption-and-production-round-table-working-group-1-workshop-towards.

European Commission (2020). *Communication from the Commission to the European Parliament, The Council, The European Economic and Social Committee and The Committee of the Regions. A Farm to Fork Strategy for a Fair, Healthy and Environmentally-Friendly Food System. COM(2020) 381 final*. https://eur-lex.europa.eu/resource.html?uri=cellar:ea0f9f73-9ab2-11ea-9d2d-01aa75ed71a1.0001.02/DOC_1&format=PDF.

Filho, W. Paço, A., Mifsud, M., Ávila, L.V., Brandli, L.L., Molthan-Hill, P. *et al*. (2019). Sustainable Development Goals and sustainability teaching at universities: Falling behind or getting ahead of the pack? *Journal of Cleaner Production* 232, 285-294. https://core.ac.uk/download/pdf/211231862.pdf.

Food and Agriculture Organization of the United Nations (2019). *The International Code of Conduct for the Sustainable Use and Management of Fertilizers*. Rome. http://www.fao.org/3/ca5253en/CA5253EN.pdf.

Food and Agriculture Organization of the United Nations (2021). Food and agriculture data. https://www.fao.org/statistics/faostat#. Accessed 26 January 2021.

Fuglie, K. (2016). The growing role of the private sector in agricultural research and development world-wide. *Global Food Security* 10, 29-38. https://www.sciencedirect.com/science/article/abs/pii/S2211912416300190?via%3Dihub.

George, C. (2014). *Environment and Regional Trade Agreements: Emerging Trends and Policy Drivers*. OECD Trade and Environment Working Papers. https://www.oecd-ilibrary.org/docserver/5jz0v4q45g6h-en.pdf?expires=1609147770&id=id&accname=guest&checksum=B6EA9682903B8ACEC6EC2D8EB364425F.

Global Open Data for Agriculture and Nutrition (2021). Harnessing open data to achieve global food security. https://www.godan.info. Accessed 26 January 2021.

Grand, D. and Vasi, I. (2017). Civil society in an age of environmental accountability: How local environmental nongovernmental organizations reduce U.S. power plants' carbon dioxide emissions. *Sociological Forum* 32(1), 94-115. https://onlinelibrary.wiley.com/doi/abs/10.1111/socf.12318.

Gu, B., Ju, X., Wu, Y., Erisman, J., Bleeker, A., Reis, S. *et al*. (2018). Cleaning up nitrogen pollution may reduce future carbon sinks. *Global Environmental Change* 48, 56-66. http://nora.nerc.ac.uk/id/eprint/519609/1/N519609PP.pdf.

Haddaway, N. (2018). Open synthesis: On the need for evidence synthesis to embrace Open Science. *Environmental Evidence* 7, 26. https://environmentalevidencejournal.biomedcentral.com/articles/10.1186/s13750-018-0140-4. Accessed 26 January 2021.

Ingram, J. and Mills, J. (2019). Are advisory services "fit for purpose" to support sustainable soil management? An assessment of advice in Europe. *Soil Use and Management* 35, 21-31. https://onlinelibrary.wiley.com/doi/full/10.1111/sum.12452.

International Labour Organization (2021). GOALI: Global Online Access to Legal Information. https://www.ilo.org/goali/WCMS_588687/lang--en/index.htm. Accessed 26 January 2021.

International Nitrogen Management System (2019). *Fourth Meeting of the International Nitrogen Management System (INMS-4). Summary of the High-Level Segment. 29-30 April 2019*. https://www.inms.international/sites/inms.international/files/INMS-4%20Summary%20Report%20of%20high-level%20segment%20(30%20July%202019)%20(002).pdf.

International Nitrogen Management System (2021). Towards the establishment of an International Nitrogen Management System. https://www.inms.international. Accessed 26 January 2021.

Ji, Y., Liu, H. and Shi, Y. (2020). Will China's fertilizer use continue to decline? Evidence from the LMDI analysis based on crops, regions and fertilizer types. fertilizer types. *PLoS ONE* 15(8), e0237234. https://doi.org/10.1371/journal.pone.0237234.

Jin, S. and Zhou, F. (2018). Zero growth of chemical fertilizer and pesticide use: China's objectives, progress and challenges. *Journal of Resources and Ecology* 9(1), 50-58. http://www.jorae.cn/EN/abstract/abstract41733.shtml.

Jin, S., Zhang, B., Wu, B., Han, D., Hu, Y., Ren, C. *et al*. (2020). Decoupling livestock and crop production at the household level in China. *Nature Sustainability* 4, 48-55. https://doi.org/10.1038/s41893-020-00596-0.

Kanter, D. and Brownlie, W. (2019). Joint nitrogen and phosphorus management for sustainable development and climate goals. *Environmental Science and Policy* 92, 1-8. https://www.sciencedirect.com/science/article/abs/pii/S1462901118306798?via%3Dihub.

Kirchmann, H., Börjesson, G., Kätterer, T. and Cohen, Y. (2017). From agricultural use of sewage sludge to nutrient extraction: A soil science outlook. *Ambio* 46, 143-154. https://link.springer.com/article/10.1007/s13280-016-0816-3.

Ma, M., Hou, Y. and Bai, Z. (2020). Nutrient cycling in agriculture in China. In *Biorefinery of Inorganics: Recovering Mineral Nutrients from Biomass and Organic Waste*. Meers, E., Velthof, G., Michels, E. and Rietra, R (eds.). Wiley Online Library. Chapter 2.4. https://onlinelibrary.wiley.com/doi/pdf/10.1002/9781118921487.ch2-4.

Macdiarmid, J., Douglas, F., and Campbell, J. (2016). Eating like there's no tomorrow: Public awareness of the environmental impact of food and reluctance to eat less meat as part of a sustainable diet. *Appetite* 96, 487-493. https://pubmed.ncbi.nlm.nih.gov/26476397/.

Mamun, A., Martin, W. and Tokgoz, S. (2019). *Reforming Agricultural Subsidies for Improved Environmental Outcomes*. International Food Policy Research Institute. https://www.foodandlandusecoalition.org/wp-content/uploads/2019/08/Reforming-Agricultural-Subsidies-for-Improved-Environmental-Outcomes-2019_09_06-.pdf.

Mittal, S., Padmaja, S. and Ajay, A. (2018). Agricultural information and knowledge network in rural India: a case of Bihar. *Journal of Agricultural Education and Extension* 24(5), 393-418. https://www.semanticscholar.org/paper/Agricultural-information-and-knowledge-network-in-a-Mittal-Padmaja/37798844f9452a71eea87517579eea7b04db5907.

Norton, G. and Alwang, J. (2020). Changes in agricultural extension and implications for farmer adoption of new practices. *Applied Economic Perspectives and Policy* 42(1), 8-20. https://onlinelibrary.wiley.com/doi/full/10.1002/aepp.13008.

Oenema, O. (2019). *Principles of Integrated, Sustainable Nitrogen Management Draft Section for a Guidance Document*. *United Nations Economic Commission for Europe Task Force on Reactive Nitrogen*. https://ec.europa.eu/info/sites/info/files/principles_of_nitrogen_management_-_17sep19.pdf.

Organisation for Economic Co-operation and Development (2018). *Agricultural Risk Management and Resilience: A Holistic Approach*. Agriculture Policy Brief. https://issuu.com/oecd.publishing/docs/agricultural_risk_management_and_re.

Plan S (2021). Making full and immediate data access a reality. https://www.coalition-s.org. Accessed 22 January 2021.

ResearchGate (2021). Discover scientific knowledge and stay connected to the world of science. https://www.researchgate.net. Accessed 26 January 2021.

Searchinger, T.D.; Malins, C., Dumas, P., Baldock, D., Glauber, J., Jayne, T. *et al*. (2020). *Revising Public Agricultural Support to Mitigate Climate Change. Development Knowledge and Learning*. Washington, D.C.: World Bank. https://openknowledge.worldbank.org/handle/10986/33677.

Sheehan, J. (2016). Increasing access to the results of federally funded science, 22 February. The White House, President Barack Obama. https://obamawhitehouse.archives.gov/blog/2016/02/22/increasing-access-results-federally-funded-science. Accessed 27 January 2021.

SOHU (2018). The latest policy on organic fertilizer subsidies and rewards in 2018 (announced by the Ministry of Agriculture and Rural Affairs) (in Chinese). https://www.sohu.com/a/255329858_807744. Accessed 26 January 2021.

Somanje, A., Crespo, O. and Zinyengere, N. (2017). Conservation agriculture among farmers in Kalomo, Zambia: Potential for productivity under climate change. In *Beyond Agricultural Impacts: Multiple Perspectives on Climate change and Agriculture in Africa*. Zinyengere, N., Theodory, T., Gebreyes, M. and Speranza, C. (eds.). Academic Press. 77-99. https://www.sciencedirect.com/science/article/pii/B9780128126240000053?via%3Dihub.

Sud, M. 2020. *Managing the Biodiversity Impacts of Fertiliser and Pesticide Use: Overview and Insights from Trends and Policies across Selected OECD Countries*. OECD Environment Working Papers No. 155. https://www.oecd-ilibrary.org/environment/managing-the-biodiversity-impacts-of-fertiliser-and-pesticide-use_63942249-en.

Sustainable Agriculture Initiative Platform (2021). Sustainable agriculture for a better world. https://saiplatform.org. Accessed 26 January 2021.

Sutton, M., Raghuram, N. and Adhya, T. K. (lead authors). (2019). The nitrogen fix: From nitrogen cycle pollution to nitrogen circular economy. In *Frontiers 2018/19: Emerging Issues of Environmental Concern*. Nairobi: United Nations Environment Programme. 52-64. https://wedocs.unep.org/bitstream/handle/20.500.11822/27543/Frontiers1819_ch4.pdf?sequence=1.

Suvedi, M., and Kaplowitz, M. (2016). *What Every Extension Worker Should Know: Core Competency Handbook*. *MEAS Handbook*. Global Forum for Rural Advisory Services (GFRAS). https://www.canr.msu.edu/csus/uploads/457/48841/meas-2016extensionhandbook.pdf.

United Nations Environment Programme (2017). *Towards a Pollution-free Planet: Background Report*. https://wedocs.unep.org/bitstream/handle/20.500.11822/21800/UNEA_towardspollution_long%20version_Web.pdf?sequence=1&isAllowed=y.

United Nations Environment Programme (2021). Online access to research in the environment (OARE). https://www.unenvironment.org/explore-topics/environment-under-review/what-we-do/information-management/online-access-research. Accessed 27 January 2021.

United Nations Environment Programme (2022). *Synthesis Report on Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them*. https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

United Nations Environment Programme (n.d.). Global Partnership on Nutrient Management. https://www.unenvironment.org/explore-topics/oceans-seas/what-we-do/addressing-land-based-pollution/global-partnership-nutrient. Accessed 27 January 2021

United States Congress (2008). NIH public access policy details. https://publicaccess.nih.gov/policy.htm. Accessed 27 January 2021.

United States Department of Agriculture (2019). Agricultural research funding in the public and private sectors. https://www.ers.usda.gov/data-products/agricultural-research-funding-in-the-public-and-private-sectors/. Accessed 27 January 2021.

United States Department of Agriculture (2021). National Agricultural Library. https://pubag.nal.usda.gov. Accessed 26 January 2021.

United States Geological Survey (2021). USGS Publications Warehouse. https://pubs.er.usgs.gov. Accessed 26 January 2021.

United States Government Accountability Office (2019). *Federal Research: Additional Actions Needed to Improve Public Access to Research Results*. https://www.gao.gov/assets/710/702847.pdf.

United States Library of Medicine (2021). PubMed. https://pubmed.ncbi.nlm.nih.gov. Accessed 26 January 2021.

United States Office of Science and Technology Policy (2013). *Increasing Access to the Results of Federally Funded Scientific Research*. https://obamawhitehouse.archives.gov/sites/default/files/microsites/ostp/ostp_public_access_memo_2013.pdf.

United States Office of Science and Technology Policy (2020). *Request for Information: Public Access to Peer-Reviewed Scholarly Publications, Data and Code Resulting from Federally Funded Research. Posted to the Federal Register on February 19, 2020*. https://www.govinfo.gov/content/pkg/FR-2020-02-19/pdf/2020-03189.pdf.

United States Senate (2017). S.1701 – Fair Access to Science and Technology Research Act of 2017. https://www.congress.gov/bill/115th-congress/senate-bill/1701?q=%7B%22search%22%3A%22S.+1701%22%7Dands=1&r=2. Accessed 26 January 2021.

Wamatu, J. and Ephrem, N. (2020). *Bridging the Gap between Strategy and Delivery of Improved Sheep Fattening in Ethiopia*. Monitoring, Evaluation and Learning. https://repo.mel.cgiar.org/handle/20.500.11766/10759.

Wolkovich, E., Regetz, J. and O'Connor, M. (2012). Advances in global change research require open science by individual researchers. *Global Change Biology* 18, 2102-2110. https://onlinelibrary.wiley.com/doi/abs/10.1111/j.1365-2486.2012.02693.x.

World Bank (2021a). Enabling the Business of Agriculture (EBA). Country scores. https://eba.worldbank.org/en/data/exploretopics/all-topics. Accessed 26 January 2021.

World Bank (2021b). World Bank Open Data. https://data.worldbank.org. Accessed 26 January 2021.

World Health Organization (2021). Hinari Access to Research for Health programme. https://www.who.int/hinari/en/. Accessed 26 January 2021.

World Intellectual Property Organization (2021). ARDI – Research for Innovation. https://www.wipo.int/ardi/en/. Accessed 26 January 2021.

Zhang, X., Fang, Q., Zhang, T., Ma, W., Velthof, G.L., Hou, Y. *et al*. (2020). Benefits and trade-offs of replacing synthetic fertilizers by animal manures in crop production in China: A meta-analysis. *Global Change Biology* 26(2), 888-900. https://onlinelibrary.wiley.com/doi/abs/10.1111/gcb.14826.

Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them

Envisioning A Chemical-Safe World

Chapter 12 of 12

Transformative actions to minimize the adverse impacts of pesticide and fertilizers

Contents

1 Global drivers, actors and policies affecting pesticides and fertilizer use

2 Status and trends of pesticide use

3 The regulatory and policy environment for pesticide management

4 Environmental and health effects of pesticide use

5. The environmental, human health and economic impacts of pesticides

6 Current pesticide risk reduction and risk management

7 Status and trends of fertilizer use

8 The regulatory and policy environment of fertilizer management and use

9 Environmental and health effects of fertilizer use

10 The impact of fertilizer use

11 Current fertilizer risk reduction and risk management

About ii

12 Transformative actions to minimize the adverse impacts of pesticide and fertilizers 1

12.1 Transformative actions are needed and within reach 1

12.2 Priority transformative actions 4

12.3 Ambitious collaborative action by all stakeholders is needed 8

References 10

List of Tables

Table 12.1 The interface between the 2030 Agenda and sustainable management of pesticides and fertilizers 2

About

In December 2017, Resolution 4 of the 3rd Session of the United Nations Environment Assembly (UNEA 3) requested "the Executive Director to present a report on the environmental and health impacts of pesticides and fertilizers and ways of minimizing them, given the lack of data in that regard, in collaboration with the World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO) and other relevant organizations by the fifth session of the United Nations Environment Assembly". In response to this request, UNEP published a *Synthesis Report on the Environmental and Health Impacts of Pesticides and Fertilizers and Ways to Minimize Them*[1] in February 2022 (United Nations Environment Programme [UNEP] 2022).

The overall goal of the synthesis report is to provide the information base to enable other advocacy actions to be taken by stakeholders to minimize the adverse impacts of pesticides and fertilizers. Specific objectives of the synthesis report are to:

- Update understanding of current pesticide and fertilizer use practices;
- Present major environmental and health effects of pesticides and fertilizers, during their life cycle, and identify key knowledge gaps;
- Review current management practices, legislation and policies aimed at reducing risks in the context of the global chemicals, environmental and health agenda;
- Identify opportunities to minimize environmental and health impacts, including proven and innovative approaches.

This chapter on "Transformative actions to minimize the adverse impacts of pesticide and fertilizers" is the 12th in a series of 12 chapters that make up a comprehensive compilation of scientific information. The chapters were developed to both inform and further elaborate on the information provided in the synthesis report. Please note that the disclaimers and copyright from the synthesis report apply

1 The Synthesis report is available at https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

Transformative actions to minimize the adverse impacts of pesticide and fertilizers

12

"Business-as-usual is not an option" also applies to scaling up incremental changes

The previous parts of this report have proposed various types of incremental change to strengthen pesticide and fertilizer management schemes, methods and practices. The proposed interventions focus on strengthening and building upon existing policy and management frameworks. They are often technical in nature. Options for incremental changes are described in Chapter 6 for pesticides, and in Chapter 11 for fertilizers. Such options are mainly addressed to the technical and policy communities working on pesticide and fertilizer management.

12.1 Transformative actions are needed and within reach

Current and projected patterns of pesticide and fertilizer use are not sustainable

This report assesses the current situation with regard to global use patterns of pesticides and fertilizers, as well as existing policy and regulatory frameworks that address their manufacture, distribution and use. The report also evaluates the prevailing scientific understanding of the environmental and human health impacts of pesticides and fertilizers and identifies important knowledge gaps.

Despite the international, regional and national management approaches used to minimize the adverse impacts of pesticides and fertilizers, evidence presented in Parts II and III of the report suggests that reducing these impacts presents many challenges. Moreover, as the use of pesticides and fertilizers increases their impacts also increase. For a summary of these impacts, see Chapters 4, 5, 9, and 10.

The projected growth of pesticide and fertilizer markets, together with prevailing deficiencies in management systems, will result in greater adverse impacts and more inefficient nutrient use in coming years unless a fundamental change takes place. Business-as-usual is not an option.

There is a need for transformative actions and for scaling up incremental changes

This chapter, which is the final chapter of the report, emphasizes that both transformative actions and the scaling up of incremental changes are needed to achieve a chemical-safe world with regard to pesticides and fertilizers. To ensure

that the transition to this chemical-safe world is fair and leaves no one behind, transformative actions should be coupled with supporting ones. Joint commitment by all stakeholders is required in order to achieve sustainability scenarios for pesticide and fertilizer use, in contrast to a business-as-usual scenario that perpetuates an unsustainable status quo.

Transformative actions seek to increase demand for more sustainable agricultural products that minimize unsustainable pesticide and fertilizer use and fundamentally change a range of management practices. This chapter describes such transformative actions. Many of them target actors in the value chain beyond those working directly on pesticide and fertilizer management.

The proposed transformative actions may not be applicable globally. In addition, specific conditions in countries and regions need to be considered. These include, but are not limited to, differences in economic situations, agroecological conditions, past histories of sustainable pest and nutrient management, cultural and gender aspects, existing policies and legislation, and the effectiveness of their implementation.

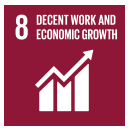
Using the 2030 Sustainable Development Agenda to engage new actors and foster transformative action

The 2030 Sustainable Development Agenda is a universal global framework that encourages action by all stakeholders to achieve sustainable development by the year 2030 (United Nations 2020). The 17 Sustainable Development Goals (SDGs) and their specific targets serve as a detailed reference framework and global compass to inspire and shape action by all relevant actors, particularly policymakers (Moyo 2016). Major global agreements and policies relevant to pesticides and fertilizers operationalize the SDGs in various ways with regard to the specific sectors or aspects to which they apply, creating synergies for implementation.

The SDGs are central to the identification in this report of options for actions which could be taken by governments and other stakeholders. Effective implementation of these actions can be expected to contribute to meeting several SDG targets (Table 12.1). For pesticide and nutrient management to become sustainable in the long run, the actions proposed in the report need to

Table 12.1 The interface between the 2030 Agenda and sustainable management of pesticides and fertilizers

SDGs	SDG targets	Pesticides and fertilizer linkages
	Eradicate extreme poverty for all people everywhere, currently measured as people living on less than USD 1.25 a day	Increased need for efficient, profitable and sustainable use of pesticides and nutrients
	2.2 End all forms of malnutrition 2.3 Double the agricultural productivity and incomes of small-scale food producers 2.4 Ensure sustainable food production systems and implement resilient agricultural practices	Increased need for effective pest and nutrient management Need to increase sustainable use of fertilizers and pesticides in certain parts of the world Wider adoption of sustainable agricultural production practices
	3.9 Reduce the number of deaths and illnesses from hazardous chemicals	Further reduction of death and illnesses mainly from pesticides Ensuring access to sufficient, safe and nutritious food
	6.3 Improve water quality by reducing pollution, eliminating dumping, and minimizing release of hazardous chemicals and materials	Minimization of water pollution from pesticides and fertilizers

SDGs	SDG targets	Pesticides and fertilizer linkages
	8.8 Promote safe and secure working environments for all workers	Reduction of the occupational risks of handling and using pesticides and fertilizers
	9.5 Enhance scientific research and upgrade the technological capabilities of industrial sectors in all countries, particularly developing countries, including encouraging innovation	Development of innovative and sustainable pest and nutrient management approaches and technologies
	12.2 Achieve the sustainable management and efficient use of natural resources 12.4 Achieve the environmentally sound management of chemicals and all wastes throughout their life cycle 12.6 Encourage companies, especially large and transnational ones, to adopt sustainable practices and to integrate sustainability information into their reporting cycle 12.8 Ensure that people everywhere have the relevant information and awareness for sustainable development and lifestyles	Wider adoption of sustainable pest and nutrient management practices Minimization of the impacts of pesticides and fertilizers on natural resources Further strengthening of the sound management of the entire life cycle of pesticides and fertilizers Further support for and implementation of sustainable pest and nutrient management technologies by the pesticide and fertilizer industries Improvement of information provision about the risks of pesticides and fertilizers and ways to minimize them
	13.2 Integrate climate change measures into national policies, strategies and planning	Minimization of greenhouse gas emissions associated with fertilizers use Saving forests and increasing carbon storage through judicious use of fertilizers Wider adoption of integrated practices in agriculture that enhance farmers' sustainable productivity as well as climate resilience
	14.1 Prevent and significantly reduce marine pollution of all kinds, in particular from land-based activities, including nutrient pollution	Minimization of pollution of marine environments by nutrients and contaminants in fertilizers
	15.5 Take urgent and significant action to reduce the degradation of natural habitats, halt the loss of biodiversity, and protect and prevent the extinction of threatened species 15.8 Prevent the introduction and significantly reduce the impact of invasive alien species on land and water ecosystems, and control or eradicate the priority species 15.9 Integrate ecosystem and biodiversity values into national and local planning, development processes, poverty reduction strategies and accounts	Minimization of the environmental impacts of pesticide and fertilizer use Ensuring the sustainable control of invasive pest species Mainstreaming of ecosystem and biodiversity values into national and regional pest and nutrient management policies
	17.6 Enhance North-South, South-South and triangular regional and international cooperation on and access to science, technology and innovation and enhance knowledge sharing	Improvement of sharing of pest and nutrient management knowledge among relevant stakeholders Enhancing partnering among UN organizations active in sound management of chemicals

be integrated effectively into the implementation of relevant SDGs. Just as the SDGs should be addressed in an integrated, indivisible manner, the actions proposed in this report cannot be implemented in isolation. They require a unified approach.

12.2 Priority transformative actions

The following options for priority transformative actions have been identified through consultations with government experts, independent scientists, experts from intergovernmental organizations, and other stakeholders from the public and private sector. Identification of these transformative actions is based on the impact they are expected to have on the sustainability of pesticide and fertilizer use, the degree to which they would minimize adverse impacts, and their interconnection with the SDGs and associated global policies.

While they are relevant to all regions in the world, implementation of these actions needs to take account of local contexts and situations, as appropriate. In certain cases actions proposed in other chapters may partly overlap with the proposals below, given the scope and complex nature of the transformative actions.

12.2.1 Incentivize healthy and sustainable consumer choices and consumption

Consumer choices play a crucial role in fostering a sustainable food system and value chain. For example, increased household incomes may result in shifts from plant-based to animal-based diets. Animal-based diets are associated with higher demand for resources (e.g., land, pesticides and fertilizers) and, subsequently, more environmental pollution than plant-based diets. Making dietary changes towards more plant-based diets can contribute to minimizing the adverse impacts of pesticides and fertilizers and contribute towards sustainable food systems.

Governments may incentivize healthy and sustainable consumer choices and consumption by enacting policies that promote greater dietary reliance on sustainably produced grains, fruits, vegetables and dairy products, as well as mandatory use of certification systems and labels which address sustainability considerations throughout the value chain by creating full product transparency. These measures may be complemented by innovative communication campaigns and technologies (e.g., mobile phone apps) that synthesize complex information, so that consumers are able to make informed choices.

Options for transformative actions to incentivize *responsible consumption of food, feed and fibres* include:

- Promote consumption of food, feed and fibres which have been produced in a sustainable manner, for example by facilitating market access, labelling schemes and other economic incentives throughout the value chain;
- Make consumers more aware of the pollution and health footprints of agricultural inputs, so that they can make informed choices (e.g., choose diets or clothing with smaller pollution footprints);
- Create attractive environments for the marketing of sustainably produced food, feed and fibres;
- Encourage the food, feed and textile industries to systematically integrate sustainability into their core business strategies and adopt sustainability standards;
- Encourage governments, educators and non-governmental organizations (NGOs) to provide product sustainability information, for example through labelling or consumer campaigns;
- Ensure fair pricing systems for food, feed and fibres produced in a sustainable manner.

12.2.2 Fundamentally change crop management and adopt ecosystem-based approaches

Shifting from a traditional linear to a more holistic approach to address pest pressure and soil fertility issues in agricultural production is essential. This approach uses environmental knowledge and takes into consideration other plant management factors (e.g., germplasm, water)

that influence productivity. Examples of concepts that have proven effective include integrated pest management and integrated vector management, biocontrol, rotation of legumes and non-legumes, and combining cropping with livestock systems. These concepts already exist and have proven effective. However, their implementation should be scaled up.

Providing knowledge, and sharing it widely, will expand the solution base for farmers and allow the adoption of environmentally friendly and sustainable production systems. These fundamental shifts in agricultural production cannot be achieved without active support by the private sector, while governments need to create enabling environments and establish boundaries with regard to practices that will be considered undesirable in the future.

Options for transformative actions to *fundamentally change crop management* and move towards ecosystem-based approaches and solutions to manage pests and nutrients include:

- Make sustainable agricultural intensification the central objective of national policies; promote agroecological approaches for crop, fibre and livestock production; discourage unsustainable agricultural practices;
- Effectively implement integrated pest management (IPM), integrated vector management (IVM), biocontrol and agroecology as the principal approaches to pest management;
- Support the development, availability and affordability of non-chemical alternative pest and vector control products and methods;
- Promote cropping systems that reduce the need for application of fertilizers (e.g., rotation of legumes with non-legumes) and exploit synergies between cropping systems and livestock systems;
- Promote technologies and practices that reduce nutrient losses (e.g., integrated nutrient management, precision farming, enhanced efficiency fertilizers, biochar).

12.2.3 Promote circularity and resource efficiency

A large amount of the fertilizer nutrients used in agriculture are not recycled, creating opportunities to advance circularity through collaborations and partnerships of relevant stakeholders. For example, recycling nutrients in manure is often hampered by spatial separation of crop and livestock production. Taking advantage of synergies between the two systems can make pesticide and nutrient use more sustainable.

Options for transformative action to ensure *circularity and resource efficiency* throughout the life cycle of pesticides and fertilizers include:

- Implement more circular agricultural production systems that reduce the need for pesticides and synthetic fertilizers, based, for example, on crop rotation, nitrogen fixation, organic agriculture, crop varieties adapted to low-input agriculture, agroecology and conservation (zero-till) agriculture;
- Implement upstream quality control of fertilizers, pesticides and other pest management tools before they reach the market;
- Promote the use of modern technologies that contribute to improved pesticide and nutrient use efficiency, such as modelling to forecast pest development, adoption of economic thresholds, lure and kill and targeted application of pesticides, precision agriculture, improved pesticide formulations and application technologies, cloud-based tools for site-specific nutrient management, and enhanced efficiency fertilizers;
- Establish national systems for the collection and recycling of empty pesticide containers and environmentally sound treatment or disposal of other pesticide waste, particularly through public-private partnerships;
- Support recycling nutrients, for example through investing in the development of technology for nutrient recovery and treatment and in farmer training;

- Build consumer awareness of the safety of food grown using recycled nutrients (e.g., from household waste, treated sewage sludge) in countries where these nutrients are properly treated to ensure safety.

12.2.4 Use economic instruments and promote direct finance to create a level playing field for greener products and approaches

The environmental and human health costs associated with agricultural production are often not reflected in input and output pricing. Opportunities exist to progressively internalize the hidden costs of pesticide and fertilizer impacts in order to level the economic playing field for greener/lower-risk products and approaches. This goal may be achieved through the use of taxes or charges or by the elimination of certain subsidies. In some cases, special compensatory measures may need to be put in place to ensure that the transition to a chemical-safe world with regard to pesticides and fertilizers is fair and leaves no one behind.

Providing financial support to encourage the adoption of technologies and practices that improve the use efficiencies of pesticides and fertilizers can help achieve a shift to more sustainable agriculture. Funds may be mobilized, for example, through taxes that penalize polluters and reward non-polluters. Where subsidies are used to increase access to fertilizers (e.g., in some low and middle income countries), support for technologies that increase fertilizer efficiency may be more effective. A careful balance is needed between input and technology subsidies, taking regional conditions into consideration.

Options for transformative action to *incentivize and redirect finance* to more sustainable pest and nutrient management include:

- Sensitize policymakers about the importance of basic public funding as a requirement to ensure sustainable pesticide and nutrient management and to minimize indirect environmental, health and economic costs to society;
- Progressively internalize the environmental and human health costs of the use of pesticides and fertilizers in their pricing to level the economic playing field for greener/lower-risk products and approaches;
- Apply smart economic policies (e.g., pricing policies for inputs and outputs) to achieve the right balance between food security and environmental health;
- Establish smart subsidies and taxes to promote sustainable pest and nutrient management, and remove counterproductive subsidies and tax exemptions;
- Redirect revenues from economic instruments towards supporting farmers in shifting to more sustainable practices, as well as towards research and development (R&D) that supports such shifts;
- Facilitate the marketing and use of technology packages that contribute to sustainable use of nutrients, lower-risk pesticides, bioprotectants and pest management instruments through discriminatory smart subsidies.

12.2.5 Adopt integrated and life cycle approaches for sound pesticide and fertilizer management

The current practice of regulating individual pesticides and fertilizers could be transformed by promoting broader sustainable pest and nutrient management solutions. Alternative pest and nutrient management options should be evaluated as part of the decision-making process, together with an evaluation of the economic and environmental impacts of these options. Evidence-based and interdisciplinary decision-making should drive pest and nutrient management choices, while uncertainties and knowledge gaps are taken into account in a precautionary manner.

Options for transformative actions to move towards ***more*** *integrative approaches for the sound management* of pesticides and fertilizers include:

- Transform the current focus on the regulation of pesticides and fertilizers to the promotion of sustainable and holistic approaches in pest and nutrient management solutions;
- Include all relevant sectors of government in the authorization process of pesticides and fertilizers;
- In addition to evaluating biological efficacy and environmental and human health risks, assess local needs for individual pesticide and fertilizers, as well as risks compared with those of other pest and nutrient management approaches, in the registration process;
- Ensure evidence-based decision-making while taking account of the precautionary principle in situations where there are scientific complexity, uncertainty and knowledge gaps, in which case there may be a need to avoid or reduce potentially serious or irreversible threats to the environment and/or to human health;
- Give the results of monitoring the efficacy and environmental and human health risks of pesticides and fertilizers a more explicit place in their (re-)authorization;
- Facilitate registration of biological and other low-risk pesticides, for example by applying specific data requirements, reducing registration costs and fast-tracking their evaluations;
- Consider, in policy decision-making, the complexity of quantifying the contribution of fertilizer use to risks, as pollutants associated with fertilizers can also be from other sources.

12.2.6 Strengthen standards and adopt corporate policies for sustainable supply chain management

Advancing sustainability standards and sustainable supply chain policies by corporate actors in the value chain (e.g., retailers, food companies, textile companies) can be an important driver for sustainable transformation of upstream agricultural practices. Including targets for sourcing organically certified products, and taking a life cycle approach, can enhance the effectiveness of such measures.

Options for transformative action to *advance sustainability standards and sustainable supply chain policies* include:

- In the case of all actors in the supply chain, enhance information sharing to reveal information about upstream use of pesticides and fertilizers and harmonize approaches to sharing such information;
- In the case of all actors in the agro-food system, widely adopt sustainability standards which address the use and impacts of pesticides and fertilizers;
- In the case of all actors in the supply chain, adopt sustainable supply chain policies with attention given to minimizing unsustainable use of pesticides and fertilizers;
- In the case of producers of pesticides and fertilizers, adopt extended producer responsibility policies to minimize their adverse impacts throughout the value chain;
- Promote emerging good practices and initiatives with regard to sustainability standards and sustainable supply chain management where these ideas, practices and initiatives are less well known;
- Adopt proactive corporate measures aimed at making sustainable products and solutions available and becoming market leaders in this respect.

12.3 Ambitious collaborative action by all stakeholders is needed

The time is right to scale up action to advance the sustainability of pesticides and fertilizers

The trends and opportunities presented in this report point in one direction: advancing the sustainability of pesticides and fertilizers offers many benefits – environmental, social and economic. Governments (including regional and local authorities), private sector actors, research institutions and other stakeholders are therefore encouraged to scale up their commitment and actions to advance the sustainable use of pesticides and fertilizers through collaborative action. Putting such collaboration into practice throughout global agri-food and public health systems is essential in order to create the transformative market shifts needed to advance the sustainability of agricultural production, products and services and minimize the adverse environmental and human health impacts of pesticides and fertilizers.

The public sector has an important role to play not only in regulating pesticides and fertilizers, but also in putting in place enabling policies and actions to foster the required transformation. Relevant measures include phasing out pesticides that pose high risks; supporting green and sustainable chemistry research; promoting sustainable agriculture, integrated pest and vector management and agroecologically based approaches; raising awareness of the full cost of unsustainable practices; and providing fiscal incentives to foster market transformation.

Developing road maps for sustainable pesticides and fertilizer management

The road map approach to planning and decision-making has been used for a number of years, including applications to technology innovation (see, for example, Phaal, Farrukh and Probert 2004). Road mapping is a technique which brings actors and stakeholders together to develop a common vision and long-term planning to achieve that vision. Beyond identifying a common vision, road maps help to identify existing resources, describe gaps, define action, and obtain adequate funding to fill gaps. They are often used in the private sector, but are equally relevant for other stakeholder groups, including public bodies.

The road map approach has been used by several actors in the chemical sector and value chain to advance action to achieve the sound management of chemicals and waste (e.g., World Business Council for Sustainable Development [WBCSD]; Figure 12.1). One of these road maps, the WHO Chemical Road Map, adopted in May 2017 by the World Health Assembly, identifies actions in which the health sector has either a lead or an important supporting role to play in advancing the sound management of chemicals and waste (WHO 2017).

Road maps have also been developed taking a value chain approach that brings together different actors across the life cycle of chemicals and chemical products. For example, the European Union (EU) Roadmap for the Chemical Industry in Europe Towards a Bioeconomy (RoadToBio 2019) applies a value chain approach for a variety of products with the goal of increasing the use of bio-based feedstocks. Given the potential benefits of road maps, the 2019 Global Chemicals Outlook (GCO-II) encouraged the development of country- and stakeholder-driven road maps on specific topics and by different stakeholder groups to support the implementation of sound management of chemicals and waste beyond 2020 and to help monitor progress at all levels including the global level (UNEP 2019).

Consistent with the suggestion in the GCO-II, road maps should be developed by diverse stakeholder groups (e.g., producers, corporations, research institutions, the public sector) to advance concerted national and global action to achieve sound management of pesticides and fertilizers.

These road maps may be developed on specific topics or themes, such as advancing adoption of sustainability standards, minimizing the adverse impacts of Highly Hazardous Pesticides (HHPs) in a particular context, or reducing pesticide and fertilizer run-off in a single watershed. They may also be elaborated at the national level to set specific goals and targets that need to be achieved in a country on its path towards sustainable pest and nutrient management.

What is required is leadership within relevant organizations. Such leadership can come from the top through senior management or policymakers, or from the bottom up through interested and committed individuals from all stakeholder groups.

Together we can achieve a world without adverse impacts from pesticides and fertilizers by taking ambitious and urgent action

References

Moyo, D. (2016). Implementing the Sustainable Development Goals is an imperative. In *OECD Yearbook 2016. Paris: Organisation for Economic Co-operation and Development.* http://www.oecd.org/development/implementing-sustainable-development-goals-imperative.htm.

Phaal, R., Farrukh, C. J. P. and Probert, D. R. (2004). Technology road mapping – A planning framework for evolution and revolution. *Technological Forecasting and Social Change* 71(1-2), 5-26. https://doi.org/10.1016/S0040-1625(03)00072-6.

RoadToBio (2019). *Roadmap for the Chemical Industry in Europe Towards a Bioeconomy – Strategy Document.* https://www.roadtobio.eu/uploads/publications/roadmap/RoadToBio_action_plan.pdf.

United Nations (2020). The Sustainable Development Agenda. https://www.un.org/sustainabledevelopment/development-agenda. Accessed 3 February 2021.

United Nations Environment Programme (2019). *Global Chemicals Outlook II – From Legacies to Innovative Solutions: Implementing the 2030 Agenda for Sustainable Development.* https://www.unenvironment.org/resources/report/global-chemicals-outlook-ii-legacies-innovative-solutions.

United Nations Environment Programme (2022). *Synthesis Report on Environmental and Health Impacts of Pesticides and Fertilizers and Ways of Minimizing Them.* https://www.unep.org/resources/report/environmental-and-health-impacts-pesticides-and-fertilizers-and-ways-minimizing.

World Business Council for Sustainable Development (2018). *Chemical Sector SDG Roadmap.* https://www.wbcsd.org/Programs/People/Sustainable-Development-Goals/Resources/Chemical-Sector-SDG-Roadmap.

World Health Organization (2017). *Chemicals Road Map - Road Map to Enhance Health Sector Engagement in the Strategic Approach to International Chemicals Management Towards the 2020 goal and Beyond.* https://www.who.int/publications/i/item/WHO-FWC-PHE-EPE-17.03.