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More people, more food, worse water?

a global review of water pollution from agriculture

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Abbreviations

BOD	Biological Oxygen Demand
CAP	Common Agricultural Policy
CGIAR	Consortium of International Agricultural Research Centers
CSR	Corporate Social Responsibility
DPSIR	Drivers, Pressures, State, Impact, Response model of intervention
EC	European Commission
EEA	European Environment Agency
FAO	Food and Agriculture Organization of the United Nations
GAP	Good Agricultural Practice
ICOLD	International Commission of Large Dams
IGRAC	International Groundwater Resources Assessment Centre
IPM	Integrated Pest Management
IWMI	International Water Management Institute
Mha	Million hectares
MIC	Measured Insecticide Concentration
MONERIS	Modelling Nutrient Emission in RIver Systems
N	Nitrogen
NAWQA	National Water-Quality Assessment
NORMAN	Network of Reference Laboratories, Research Centres and related Organisations for Monitoring of Emerging Environmental Substances
NPS	Non-Point Source
OECD	Organisation for Economic Co-operation and Development
OMZ	Oxygen Minimum Zones
P	Phosphorus
PS	Point Source
RTL	Retention Time Locking
SDG	Sustainable Development Goal
SWAT	Soil and Water Assessment Tool
TMDL	Total maximum daily load
UNCCD	United Nations Convention to Combat Desertification
UNDESA	United Nations Department of Economic and Social Affairs
UNEP	United Nations Environment Programme
UNESCO	United Nations Educational, Scientific and Cultural Organization
UNU-EHS	United Nations University Institute for Environment and Human Security

UN-Water	United Nations Water
US EPA	United States Environmental Protection Agency
USDA	United Sates Department of Agriculture
USGS	United States Geological Service
WFD	Water Framework Directive
WHO	World Health Organization
WLE	Water Land and Ecosystems
WRI	World Resources Institute
WWAP	World Water Assessment Programme

Foreword

Water is essential for sustaining economic growth and meeting increased demand for food. However, the availability of water resources globally is on the decline, and in many parts of the world, poor water quality makes it unsafe to use. Water pollution threatens both human and environmental health, affecting billions of people. Serious efforts are needed to prevent the deterioration of water quality in our lakes, aquifers, rivers and seas. The 2030 Agenda for Sustainable Development acknowledges the importance of conserving water resources and abating water pollution.

Nevertheless, while water pollution is slowly starting to receive the attention it deserves, the contribution of agriculture to this problem has not yet received sufficient consideration, particularly in the developing world. Agriculture is the single largest producer of wastewater, by volume, and livestock generates far more excreta than do humans. As land use has intensified, countries have greatly increased the use of synthetic pesticides, fertilizers and other inputs. Moreover, in many countries, livestock production has expanded and intensified even faster than crop production, introducing new types of pollutants, such as antibiotics and animal growth hormones, which pose risks as they move through aquatic ecosystems and water bodies. While these inputs have helped boost food production, they have also given rise to environmental threats, including increased pollution of aquatic ecosystems, as well as to potential human health concerns.

In the current literature, information on water pollution from agriculture is dispersed. We have lacked a comprehensive review, which is what this publication intends to provide. The report covers different rural and agricultural sectors (including crops, livestock and aquaculture), and examines the drivers of water pollution in these sectors as well as the resulting pressures and changes in water bodies, the associated impacts on human health and the environment, and the responses needed to prevent pollution and mitigate its risks.

We need a better understanding of the causes and effects of agricultural water pollution as well as effective means to prevent and remedy the problem. This book contributes to this growing imperative.

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The publication has relied on inputs from a large number of people from different institutions in various forms. Most of these scientists and experts are listed in the chapters of this publication, either as authors or contributors. We are thankful to all of them for their highly valuable inputs.

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Without this collective effort, this outcome could have not been achieved.

PART I: INTRODUCTION AND BACKGROUND


Jim Holmes (IWMI)

CHAPTER 1. SETTING THE SCENE

Javier Mateo-Sagasta and Sara Marjani Zadeh

With contributions from Olcay Unver, Marlos De Souza, Hugh Turral and Jake Burke

1.1 A global water quality crisis and the role of agriculture

Water pollution is a global challenge that has increased in both developed and developing countries, undermining economic growth as well as the socio-environmental sustainability and health of billions of people.

Although global attention has focused primarily on water quantity, water-use efficiency and allocation issues, the poor management of wastewater and agricultural drainage has created serious water quality problems in many parts of the world, worsening the water crisis (Biswas *et al.*, 2012). Water scarcity is caused not only by the physical scarcity of the resource but also by the progressive deterioration of water quality in many basins, reducing the quantity of water that is safe to use.[1]

As a response, the 2030 Agenda for Sustainable Development acknowledges the importance of water and water quality and includes three water quality targets, one specific to pollution,

[1] The Food and Agriculture Organization of the United Nations (FAO) (www.fao.org/land-water/overview/global-framework/global-framework) and the International Water Management Institute (IWMI) (www.iwmi.cgiar.org) are leading agencies in combating global water scarcity by promoting state-of-the-art sustainable water management scenarios.

in Sustainable Development Goal (SDG) 6[2]. The 2030 Agenda for Sustainable Development is expected to strongly influence future policies and strategies and to ensure that the control of water pollution is elevated in international and national priorities.

While human settlements, industries and agriculture[3] are all key sources of water pollution, in many countries, agriculture is the biggest polluter. Of the 3 928 km^3 of freshwater that is withdrawn every year, it is estimated that only 44% is consumed, mainly through evapotranspiration by irrigated agriculture. The remaining 56% (2 212 km^3 per year) is released into the environment as urban wastewater (approximately 330 km^3), industrial wastewater – including cooling water – (approximately 660 km^3) or agricultural drainage (approximately 1 260 km^3) (AQUASTAT, n.d.b; Mateo-Sagasta *et al.*, 2015).

The composition and level of treatment of these 'wastewaters,' and therefore the risks for human and environmental health, vary. Globally, 80 percent of municipal wastewater is discharged into the environment untreated, and industry is responsible for dumping millions of tonnes of heavy metals, solvents, toxic sludge and other wastes into water bodies every year (Sato *et al.*, 2013; Mateo-Sagasta *et al.*, 2015; WWAP, 2017). Yet irrigation is the largest producer in volume of wastewater (also called agricultural drainage) and livestock produces far more excreta than do humans (FAO, 2006). As a consequence, agriculture remains a key global polluter and is responsible for the discharge of large quantities of agrochemicals, organic matter, drug residues, sediments and saline drainage into water bodies (Doetterl *et al.*, 2012; Boxall, 2012; Cañedo-Argüelles *et al.*, 2013; Sutton *et al.*, 2013; Wen *et al.*, 2017). The resultant water pollution poses demonstrated risks to aquatic ecosystems, human health and productive activities (UNEP, 2016).

Industrial agriculture is among the leading causes of water pollution, especially in most high-income countries and many emerging economies, where it has overtaken contamination from settlements and industries as the major factor in the degradation of inland and coastal waters (e.g. eutrophication). Nitrate from agriculture is the most common chemical contaminant in the world's groundwater aquifers (WWAP, 2013). In the European Union, 38 percent of water bodies are under significant pressure from agricultural pollution (WWAP, 2015). In the United States of America, agriculture is the main source of pollution in rivers and streams, the second main source in wetlands and the third main source in lakes (US EPA, 2016). In China, agriculture is responsible

2 SDG Target 6.3: "By 2030, improve water quality by reducing pollution, eliminating dumping and minimizing release of hazardous chemicals and materials, halving the proportion of untreated wastewater and substantially increasing recycling and safe reuse globally" (United Nations, 2016).

3 Agriculture refers to crops, livestock and aquaculture.

for a large share of surface water pollution and is almost exclusively responsible for groundwater pollution by nitrogen (FAO, 2013). In many low-income countries and emerging economies, while the large loads of untreated municipal and industrial wastewater are major concerns, the role of cropping systems, livestock and aquaculture in water quality degradation is still unclear but suspected to be increasingly relevant.

Crops and livestock are the main agricultural pollution sources but aquaculture is emerging. Agricultural pressure on water quality come from cropping (including agroforestry) and livestock systems and aquaculture, all of which have expanded and intensified to meet increasing food demand related to population growth and mobility, and changes in dietary patterns. The area equipped for irrigation has more than doubled in recent decades from 139 million hectares – Mha – in 1961 to 320 Mha in 2012 (FAO, 2014). The total number of livestock has more than tripled from 7.3 billion units in 1970 to 24.2 billion units in 2011 (FAO, 2016a). Aquaculture has grown more than twenty-fold since the 1980s, especially inland-fed aquaculture and particularly in Asia (FAO, 2016b).

The global growth of crop production has mainly been achieved through the intensive use of inputs such as pesticides and chemical fertilizers. The trend has been amplified by the expansion of agricultural land, with irrigation playing a strategic role in improving productivity and rural livelihoods, while also transferring agricultural pollution to water bodies.

The livestock sector is growing and intensifying faster than crop production in almost all countries. The associated waste, including manure, has serious implications for water quality (FAO, 2006). In the last 20 years, a new class of agricultural pollutants has emerged in the form of veterinary medicines (antibiotics, vaccines and growth promoters such as hormones), which travel from farms through water to ecosystems and drinking-water sources (Boxall, 2012). Zoonotic water-borne pathogens are another major concern (WHO, 2012).

There has been a dramatic and rapid increase worldwide in aquaculture in marine, brackish-water and freshwater environments (FAO, 2016b). Fish excreta and uneaten feeds from fed aquaculture diminish water quality. Increased production has combined with a greater use of antibiotics, fungicides and anti-fouling agents, which in turn may contribute to polluting downstream ecosystems (Li and Shen, 2013).

The annual costs of water pollution from agriculture exceed billions of dollars.
The costs of agricultural pollution are generally non-market externalities, which are borne by society as a whole. Water pollution from agriculture has direct negative impacts on human health, for example, the well-known blue baby syndrome in which high levels of nitrates in water can cause methaemoglobinemia – a potentially fatal illness

– in infants. Pesticide accumulation in water and the food chain, with demonstrated ill effects on humans, led to the widespread banning of certain broad-spectrum and persistent pesticides (such as DDT and many organophosphates); however, some of these pesticides are still used in poorer countries, causing acute and likely chronic health effects. Aquatic ecosystems are also affected by agricultural pollution. For example, eutrophication caused by the accumulation of nutrients in lakes and coastal waters has impacts on biodiversity and fisheries (Rabalais *et al.*, 2009). Water-quality degradation may also have severe direct impacts on productive activities, including agriculture itself. For example, dam siltation caused by the mobilization of sediment due to erosion is an increasing challenge (Basson, 2008), which has cost many millions of dollars. Irrigation using saline or brackish water has limited agricultural production on hundreds of thousands of hectares worldwide (Mateo-Sagasta, 2010).

A nationwide study in the United States estimated that farm nitrogen pollution costs Americans in the range of US$59–US$340 billion a year (Sobota *et al.*, 2015). In the European Union, van Grinsven *et al.* (2013) estimated the annual cost of pollution by agricultural nitrogen to be in the range of €35–€230 billion per year. Many of these costs are associated with damages to aquatic ecosystems, deteriorating water quality and the associated human health impacts. Despite data gaps, methodological challenges and limited assessments, the Organisation for Economic Co-operation and Development (OECD) estimated that, in its member countries alone, the environmental and social costs of water pollution caused by agriculture probably exceed billions of dollars annually (OECD, 2012). This is particularly apparent when impacts from other agricultural pollutants (see Chapter 3), beyond nitrogen, are accounted for.

Diagnosis, prediction and monitoring are key requirements for the management of aquatic ecosystems and the mitigation of harmful impacts on them. If planners and lawmakers are to design cost-effective measures for preventing pollution and mitigating risks, they need to know the state of aquatic ecosystems, the nature and dynamics of the drivers and pressures that lead to water-quality degradation, and the impacts of such degradation on human health, economics and the environment.

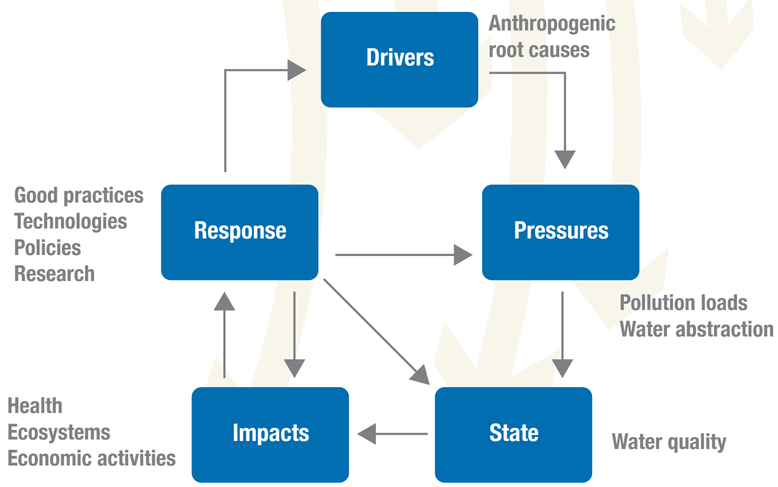
Nevertheless, because of their diffuse nature, it is difficult to identify and quantify agricultural polluters and their relative contribution to the degradation of water quality. The specific processes linking agricultural activities to pollutant concentrations in water are imperfectly understood. Improved baseline and monitoring data on management practices and water quality, together with modelling, are essential to understand the causes and effects of water pollution from agriculture and to identify and plan the right responses.

1.2 What this publication is about

Existing literature provides scattered information on water pollution from agriculture, but does not comprise a comprehensive review, which is what this publication aims to provide. The report seeks to compile and integrate the best available information and data. It covers different rural and agricultural sectors, including crops, livestock and aquaculture, and examines the drivers of water pollution from these sectors, the resulting pressures and changes in water bodies, the associated impacts on human health and the environment, and the responses needed to prevent pollution and mitigate risks.

This publication provides an analysis of problems and options for improvement. It is structured using the Drivers, Pressures, State, Impact, Response (DPSIR) model. DPSIR is a causal framework for describing the interactions between society and the environment (OECD, 1993; European Commission, 2002). The framework has been used to formulate a number of relevant policies for pollution control, including the European Water Framework Directive, and has been used by several UN organizations to produce different global public goods, such as the United Nations University Institute for Environment and Human Security (UNU-EHS)/United Nations Environment Programme (UNEP) international water quality guidelines for ecosystems (UNU-EHS/UNEP, 2016). The DPSIR framework provides a structure within which to present indicators needed to enable feedback to policy-makers on environmental quality and the impact of certain policy choices. Each of the DPSIR components is connected to another (cause-effect) (see Figure. 1.1), but can also be defined individually (Table 1.1).

FIGURE 1.1 **DPSIR framework and water quality**



Source: Adapted from OECD, 1993; European Commission, 2002.

Table 1.1 | The DPSIR framework, definitions and examples from agriculture

Term	Definition	Examples from agricultural water pollution
Driver	An anthropogenic activity that may have an environmental effect	Primary drivers: population growth and mobility, and change in consumption patterns Secondary drivers: expansion and intensification of irrigated agriculture, rain-fed agriculture, livestock production and inland aquaculture
Pressure	The direct effect of the driver	Loads of nitrogen, phosphorus, pesticides, biochemical oxygen demand, sediments, salts, organic matter, pathogens or emerging pollutants generated on-farm (at source) and reaching water bodies (e.g. rivers, lakes, aquifers, coastal waters, marine waters)
State	The condition of the water body resulting from both natural and anthropogenic factors (i.e. physical, chemical and biological characteristics of the water body)	Concentration of ammonia, nitrate phosphate, persistent organic pollutants, suspended solids and other agricultural pollutants in water bodies (e.g. rivers, lakes, aquifers, coastal waters, marine waters)
Impact	The effects of the pressure on the environment, health and the economy	ENVIRONMENT: e.g. fish killed, ecosystems modified-eutrophication HEALTH: e.g. increased human mortality or morbidity resulting from water pollution by agriculture ECONOMY: e.g. as a result of unsafe agricultural products irrigated with polluted waters or a decrease in productivity due to toxicity or salinity/sodicity
Response	The measures taken to improve the state of the water body or to mitigate the impacts of water quality degradation	Responses on drivers (including change in diets and consumption habits), pressures (including pollution prevention on-farm), state (including remediation or restoration of ecosystems) and impacts (including the control of human exposure to polluted waters)

Note: The distinction made here between state and impact separates effects that are sometimes combined, or confused. One reason for this is that because many of the impacts are not easily measurable, state is often used as an indicator of, or surrogate for, impact.

Source: adapted from the European Commission, 2002.

Although there are other important externalities resulting from agriculture expansion and intensification (e.g. greenhouse gas emissions or loss of habitat and biodiversity), the principal focus of the following chapters is water pollution induced by agriculture. Issues such as water resources depletion or soil erosion by agriculture will be only discussed as contributors to water quality degradation.

The report aims to provide:

- A GLOBAL DIAGNOSIS: When data is available, the report shows where major water quality problems are, what role agriculture plays in these problems and what are the driving forces behind them.
- RESPONSES: The report lists and describes major mitigation and remediation options at the policy level (e.g. strategies, regulations, economic instruments, cooperative agreements, education and awareness), at the farm level (e.g. best practices for agricultural inputs or for erosion control) and off-farm (e.g. vegetated buffers zones or constructed wetlands).
- A SYSTEMATIC METHODOLOGY: The report provides policymakers and practitioners with the definitions and examples they can follow to make a DPSIR analysis for agricultural water pollution. This methodology is applicable at country, river basin or watershed levels

The DPSIR analytical and response framework can include the concept of 'adaptive management,' which involves periodically assessing the results and benefits of remedial activities, and enhancing or modifying them to achieve more effective outcomes (Pahl-Wostl, 2006). Adaptive management underpins sustainable natural resource management strategies in countries such as Australia and New Zealand. Adaptive management recognizes that there may be unforeseen outcomes, synergies and impacts of responses to problems, and that achieving coherence in policy, strategy, planning and practical activities is an iterative and often cyclical process.

Much of the science, routine monitoring and regulatory and institutional development for the better management of water quality have already occurred in the developed world. There is thus a bias in both literature and experience towards the OECD nations, whereas the major emerging challenges lie in the rest of the world, where the extent and severity of the problems are not yet evident or well understood. This publication will reflect this asymmetry in information and experience.

Crucially, public and private resources are stretched in many other directions. Despite differences in context, there is much that transitional and developing countries can learn from the expensive consequences of environmental degradation in industrialized countries, giving them the potential to avoid such consequences themselves. In general, however, this publication contends that cost-effective and targeted management of agricultural non-point source pollution requires a good understanding of the context and detailed processes involved.

1.3 How to use this publication

The report is divided into three different sections, which sequentially introduce the report and review the key drivers of agricultural water pollution (Part 1: chapters 1-2); analyze the related pollution loads, state change in water bodies and resulting impacts on human health and ecosystems (Part 2: chapters 3-9); and explore different approaches to controlling water pollution from agriculture, including policies and institutional arrangements, and on-farm and off-farm responses (Part 3: Chapters 11-12). Examples will be drawn from developed and developing countries.

Chapter 2 examines the driving forces that result in the use and abuse of agricultural inputs, which in turn cause undesirable effects on the receiving waters. The chapter reviews trends in population growth and changes in diet and food demand, and examines how such changes have driven agricultural expansion and intensification, with the increased use of agriculture inputs (fertilizers, pesticides, animal feed, medicines, etc.) per unit of land.

Determining the agricultural pressures on receiving waters is complex, and the multiple factors that govern the emergence of state changes and impacts require an understanding of process and the quantity of pollutant loads. Chapters 3-8 seek to describe the processes linking pollution loads to state change of receiving waters (e.g. rivers, lakes, reservoirs, groundwater or coastal zones) and to the resulting impacts on human health and ecosystems. When available, data and information on pressures, state and impacts are presented by pollutant type, including nutrients, pesticides, salts, sediments, organic matter, pathogens and emerging pollutants. Achieving a full understanding of the relationship between pressures, state change and impacts typically requires modelling and Chapter 9 reviews existing models and their potential role, scope and application.

Pollution management requires that the sources of pollutant loads are identified so that appropriate mitigating measures can be applied. There is a broad range of approaches to managing pollutant export from farms, through broad legislative and financial measures that restrict input use, encourage greater efficiency, or actively limit the export of pollutants. Landscapes can also be managed to reduce the movement or accumulation of some pollutants and thus reduce pressure on receiving waters. In the absence of a precise understanding of cause and effect, broadly targeted regulations and controls may be applied. Practical approaches to mitigating the generation and transmission of agricultural pollutants are presented at river basin and catchment scales, down to farm and field level. All of this is covered in Chapters 10 and 11. The main messages and conclusion of the report are summarized in Chapter 12.

If you belong to the international development community, this publication will help you to identify agricultural pollution hotspots worldwide and will provide guidance on how to decouple agriculture development from water pollution though sustainable intensification.

If you are a national water policy-maker, we hope this publication will a) encourage you to adopt and apply the DPSIR approach to water quality and b) offer you a selection of water pollution prevention and remediation actions that can be undertaken at local and landscape levels. National public policy at large and health and economic sectors that rely on water of adequate quality may benefit from this publication as well.

If you are an agricultural practitioner, this report will help you understand how crop production, livestock breeding or aquaculture can impact water bodies, with serious consequences for society. It can also guide you on how to minimize your sector's footprint on water quality.

If you are a researcher, this publication will help you to identify the main knowledge gaps and research needs related to agricultural water pollution analysis and control.

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CHAPTER 2. GLOBAL DRIVERS OF WATER POLLUTION FROM AGRICULTURE

Javier Mateo-Sagasta and Hugh Turral

With contributions from Jake Burke

Population growth and changes in consumption patterns, including new dietary preferences, are changing and increasing the demand for food and, consequently, driving the transformation of our food and agricultural systems. Irrigated and rain-fed agriculture, livestock production and aquaculture are expanding and intensifying, and bringing new environmental externalities, including those on water quality. Land clearing for agriculture frequently results in land degradation and increased erosion and sediment loads on waterways. Unsustainable agricultural intensification is associated with greater water abstractions, reduced stream flows and depleted aquifers, all of which increase pollutant concentrations. Agricultural intensification has also increased the export of nutrients, pesticides and other pollutants from farms to water bodies.

While there are other relevant drivers of change in global agriculture (Hazell and Wood, 2008), this chapter focuses on the most important drivers for agricultural pollution. Section 2.1 analyses the influence of the growing population and changing diets on food demand and production patterns. Section 2.2 reviews how the expansion and intensification of agriculture are influencing the use of agricultural inputs, including land and agrochemicals. In this section, the trends of cropping systems are discussed,

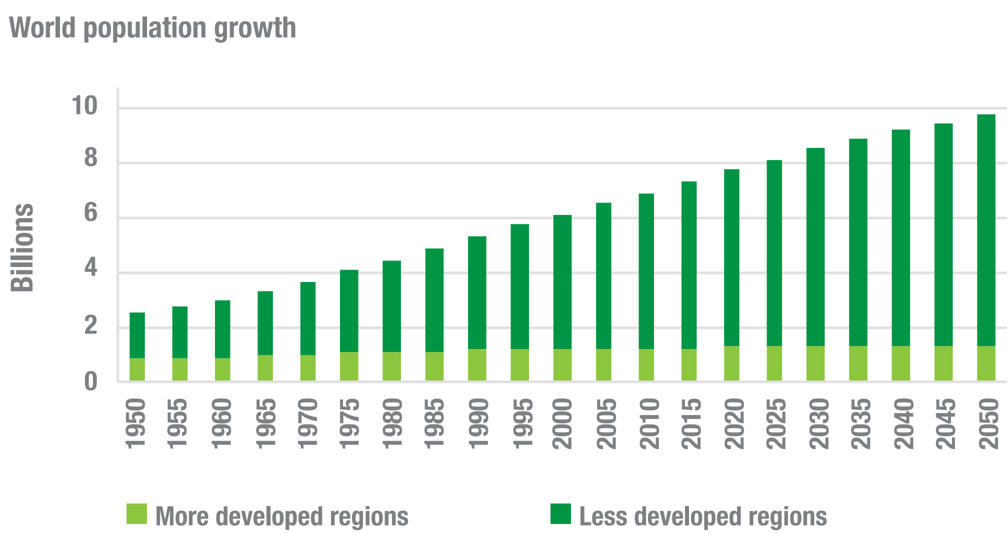
with a special focus on irrigation and the use of fertilizers and pesticides (Section 2.2.1), followed by a consideration of trends in livestock production (Section 2.2.2) and the expansion and intensification of aquaculture (Section 2.2.3).

2.1 Trends in population, diet and food demand

We are currently 7.6. billion people on earth with more than 50% of the world's population concentrated in India, China, the USA and Europe. Globally, population is projected to reach 9.8 billion people by 2050 (UNDESA, 2017) and most of this growth is forecast to take place in developing countries in Asia and Africa, while the population of OECD countries is expected to remain steady or decline (UNDESA, 2017) (Figure 2.1).

Based on economic growth over the last 35 years, forecasts predict that the global economy will be richer by 2050, with future global GDP 2.4 times greater than at present in real terms (Figure 2.2). Analyses reveal a simple and temporally-consistent global relationship between per capita GDP and per capita demand for crop calories or protein (Tilman *et al.*, 2011). As populations have become richer overall, and despite the continuing large number of people living in absolute poverty, average calorie intake has increased from about 2 000 kcal/capita/day in the 1960s to more than 2 800 kcal/capita/day today, with some countries, such as the USA, Italy, Egypt and Turkey, exceeding 3500 kcal/capita/day on average, with even higher intakes in some locations (FAO, 2017).

FIGURE 2.1 | **Past and expected global population in developed and developing countries**

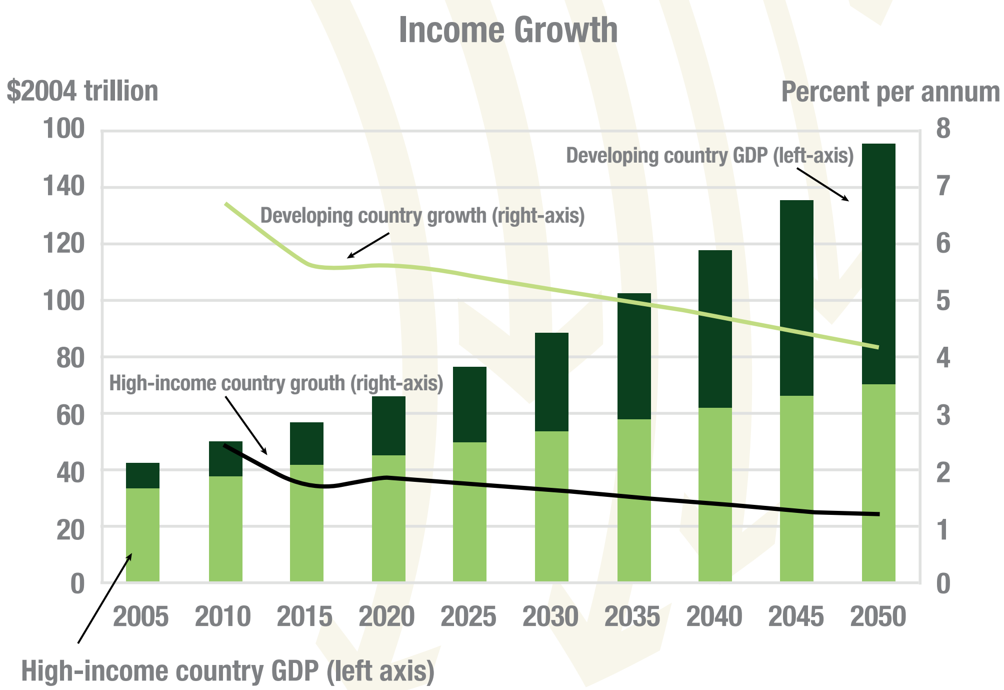


Source: UNDESA, 2017.

Dietary preferences are changing from mostly grains and carbohydrates to a greater consumption of meat, eggs, dairy, oil, fish, vegetables and fruit (Figure 2.3 and Figure 2.4) (FAO, 2009). Trends in meat consumption are a case in point (Delgado, 2003). Between 1961 and 2013, the average annual worldwide meat consumption rose from an average of around 23 kilograms per person to more than 43 kilograms per person (FAO, 2017). Meat consumption is growing rapidly in China, India, South East Asia and Latin America, and is changing the least in sub-Saharan Africa and South Asia. It has been noted that meat-based diets require greater resources per person than vegetarian diets (Cassidy *et al.*, 2013). For example, a meat-based diet requires up to three times more phosphorus per year and person compared to a vegetarian diet (Metson *et al.*, 2012).

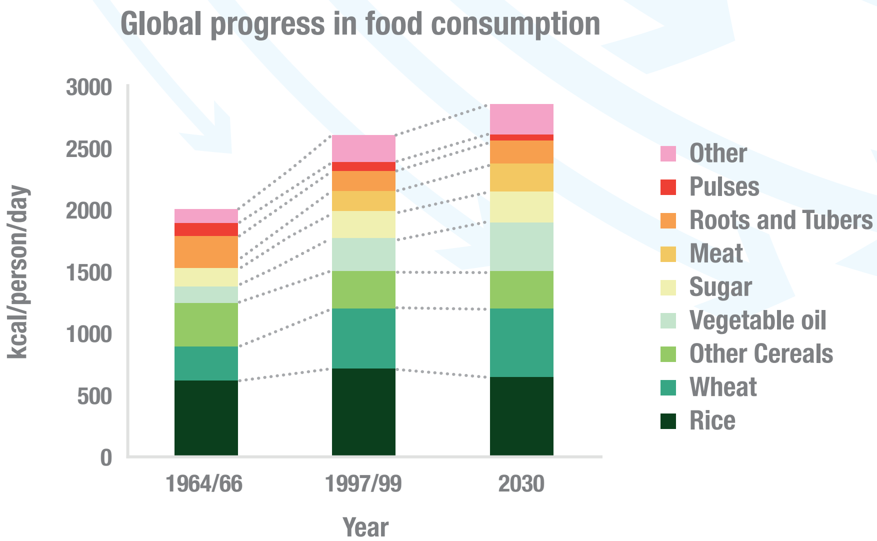
Despite the fact that the demand for cereals in human diets is not expected to increase substantially over the next three decades, the total demand for cereals is anticipated to grow to satisfy the demands for meat production, adding to the expected rise in demand for other crops. As a result, global crop demand will increase by an estimated 70% to 110% by 2050 (Alexandratos, 2009; Tilman *et al.*, 2011).

FIGURE 2.2 Recent trends and forecast growth in income in low and high-income countries



Source: van der Menssbrugghe, 2009.

FIGURE 2.3 Global food consumption patterns 1964-2030



Source: FAO, 2009.

The growth in food demand will impose clear challenges on agroecosystems, although there is little evidence that this will change dietary preferences and trends in general. Despite the lack of impact of public awareness campaigns or environmental labeling of foods on diets (Grunert, Hieke and Wills, 2014), it is apparent that targeted policies are needed to make food consumption patterns more sustainable and resource-efficient.

Another key issue relates to food supply and the response of food systems to the projected growth in food demand. Food losses and waste must be reduced as much as possible to bring food-production closer to actual demand, and to minimize the waste of resources and associated environmental impacts. About one-quarter of food production is lost along the food supply chain, which accounts for 24 percent of the freshwater resources used in crop production, 23 percent of total global cropland area and 23 percent of total global fertilizer use (Kummu *et al.*, 2012).

Nitrogen pollution is particularly important for water quality: Grizzetti *et al.* (2013) calculated that the nitrogen delivered to the global environment from food waste amounts to 6.3 teragrams per year, and that, in the European Union, 12 percent of diffuse nitrogen water pollution in agriculture is linked to food waste.

FIGURE 2.4 **Weekly food consumption of an Australian family**

Sources: Menzel and D'Aluisio, 2007.

The need to produce more food will very likely result in the additional clearing of land for food production, as well as requiring an increase of productivity on existing lands. Business as usual cannot be sustained. The necessary increases in agriculture productivity cannot be achieved at the expense of the environment, as has been the case over the last 50 years.

2.2 Expansion and intensification of agricultural systems

The implications of further intensification of crop, livestock and aquaculture production are worrying: there is potential for this to cause great and widespread harm to ecosystems and human health. Although lower profile than climate change, the challenge of developing a sustainable but highly productive approach to agriculture is beginning to tease the consciousness of both the public and policy-makers. The following sections review the often unsustainable trajectory that agrifood systems have followed over time and identify the ways in which crop production, livestock and aquaculture may have been key contributors to water quality degradation.

2.2.1 Cropping systems

The world's population doubled between 1970 and 2015. During that time, the global production of cereals almost tripled, the production of vegetables increased fourfold, tomato production increased fivefold and soybean production increased eightfold (FAO, 2017). This huge increase in production was achieved through the expansion of agricultural land (and irrigation in particular), the introduction of new crop varieties and more intensive use of agrochemicals and agrotechnologies.

In the future, FAO expects that 90 percent of the growth in global crop production will come from intensification. In developing countries, 80% of the necessary production increase will come from increases in yield and cropping intensity, and 20% from expanding arable land. By 2050, the area of arable land will be expanded by 70 million hectares, about 5% of the current area. This includes an expansion of 120 million hectares in developing countries and, in developed countries, a contraction of 50 million hectares in favour of other uses (Alexandratos, 2009).

In many OECD countries, agriculture is generally large-scale, mechanized and often specialized. Under post-war policies, the intensification of agriculture in Europe and the US proceeded apace to provide sufficient, varied, high quality and affordable food for everyone. Increasing economic efficiency, economies of scale and subsidies favoured intensification. The intensity of fertilizer use per hectare in Europe and the US probably peaked in the 1980s, but the use of organic fertilizers (farmyard manure and slurries) are on the rise and require proper management.

At the other end of the spectrum, in many low-income countries, rainfed agriculture is simply too risky to justify widespread and even replacement rates of fertilizer application. The result is declining soil health and fertility, which contributes to land degradation on a broad scale. The intensive use of agricultural chemicals is often associated with irrigation and horticulture. In low income countries, water quality degradation is more often linked to untreated wastewater from urban areas, sediment loads from soil erosion, salinization and water scarcity (which aggravates pollution) (UNEP, 2016). The stockpiling and use of obsolete pesticides is also a growing concern.

Transitional economies are witnessing increasingly intense input use and rising agricultural pollution loads, which exacerbate the environmental impacts from as-yet largely untreated industrial and municipal effluents. China's use of nutrients per hectare – mostly on irrigated (and horticultural) lands – is thought to be among the highest in the world, for example.

2.2.2 Expansion of irrigation

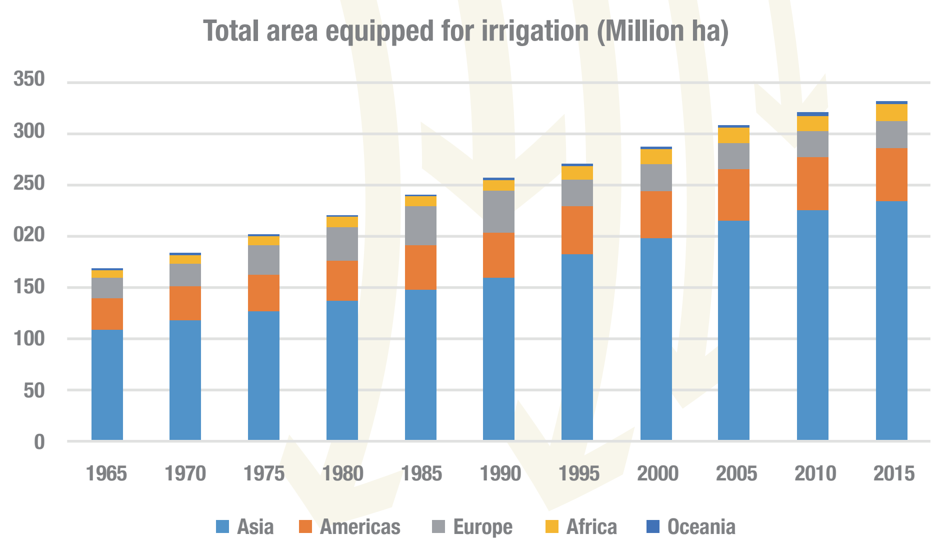
Irrigation is a major factor in agricultural intensification. Irrigation projects have helped to increase food security around the world, particularly in developing countries. Nevertheless, irrigation and drainage have often been associated with a loss of water quality caused by salt, pesticide and fertilizer runoff, and leaching.

Between 1965 and 2015, the total arable and permanent cropped area across the globe increased by 15% from 1 380 million hectares to 1 594 million hectares due to a net increase in the area equipped for irrigation (from 170 million hectares in 1961 to 333 million in 2015), particularly in Asia and the Americas (Figure 2.5) (FAO, 2017).

Currently, more than 50% of the world's irrigated land is located in India, China, the United States and the European Union (Figure 2.6.). Pakistan, India, Japan, Malta and Israel have the highest irrigation intensity, with more than 30% of their agricultural land under irrigation (World Bank, 2013).

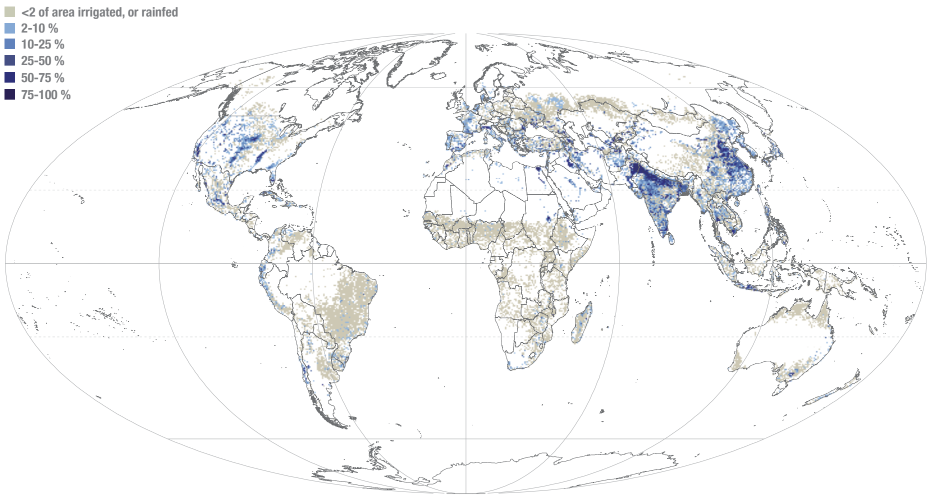
The growth of irrigated agriculture continues but at a changing pace. First, the expansion is likely to slow down in the future, limited by water availability. Also, new developments in irrigated agriculture are making a more efficient and productive use of water in a number of countries, minimizing the leaching of nitrates and other pollutants.

FIGURE 2.5 **Area equipped for irrigation (1961-2015)**



Source: FAO, 2017.

FIGURE 2.6 Global map of agricultural irrigated land



Source: Siebert *et al.*, 2007.

The value of irrigated produce has a high and growing share in agricultural production value (in excess of 50%) and in the value of exports (more than 60%), particularly in OECD countries such as Italy, Mexico, Spain and the United States. Farming increasingly uses groundwater, and the share of total use is over 30% in some countries. Groundwater overdraft is now evident in parts of Australia (e.g. the Namoi Valley) Greece, Italy, Mexico and the United States (OECD, 2008).

Irrigated agriculture will continue to be a focal point for intensification. Since irrigation is already a key cause of water scarcity and degradation in many river basins and some groundwater systems, any further intensification needs to be closely observed and managed carefully to avoid further damage to aquatic ecosystems.

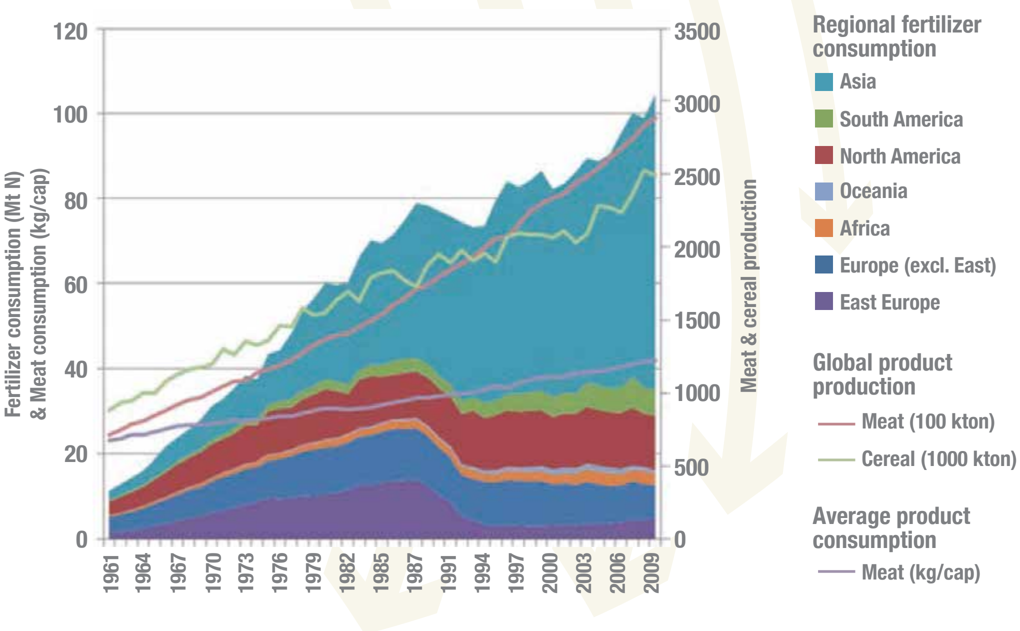
2.2.3 Trends in fertilizer use

Nutrients, especially nitrogen (N) and phosphorus (P) are essential for raising crops and animals to feed an increasing world population. Although mineral fertilizers have been used since the nineteenth century to supplement natural nutrient sources, the use of such fertilizers has increased dramatically in recent decades (Figure 2.7). Today, the world consumes ten times more mineral fertilizer than it did in the 1960s (FAO, 2017). Rockström *et al.* (2009) suggested that the mobilization of nutrients may already have exceeded thresholds that will trigger abrupt environmental change in continental-to-planetary-scale systems, including the pollution of ground and surface waters.

Global fertilizer use increased faster than crop production (particularly cereals) until the late 80's. Fertilizer use has grown coupled to crop production from then (Figure 2.7). However, fertilizer is not used on an equal basis around the world, with some countries using too many nutrients and others not enough. North America, Europe, and parts of South and South-East Asia and Latin America tend to overuse fertilizer with risks on water quality, while Africa, Central America and parts of Asia are unable to mobilize adequate nutrients to meet crop demand and food security needs (Sutton *et al.*, 2013). Fertilizer consumption has particularly boomed in East and South Asia over the last 50 years; in northern America and Europe if has been fairly stable or in decline.

The contribution of manure to total fertilizer use has declined over the last 50 to 60 years. Global manure N inputs decreased from 56% to 40% of total N inputs (from manure and fertilizers) from the 1960s to 2014 (FAO, 2018). Nevertheless, manure remains the main nutrient input to agricultural lands in many developing countries. The biggest contribution rates of manure to fertilization can be seen in Africa (84%) and Latin America (73%) (FAO, 2018).

FIGURE 2.7 **Total mineral fertilizer consumption in major world regions compared with global cereal and meat production and per capita meat consumption**



Source: Sutton *et al.*, 2013.

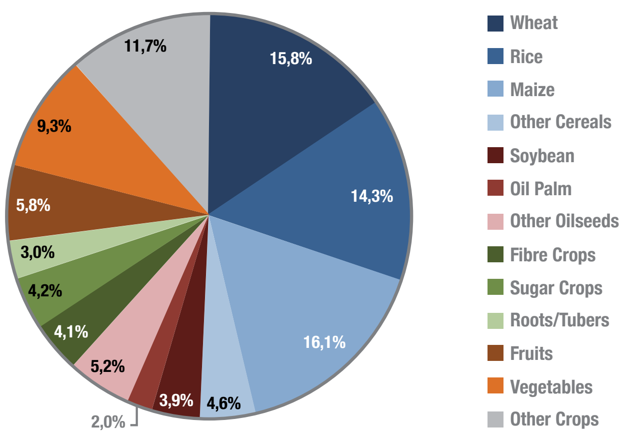
BOX 2.1 Fertilizer use in China

In transitional economies, of which China is the most powerful example, farms remain relatively small and landholdings per person continue to decline. Nevertheless, farming is becoming very intensive to keep pace with accelerating local food demand. Fertilizer use – sometimes promoted by perverse incentives – can be extreme and far from cost-effective (FAO, 2013). Although China's agricultural land accounts for only 7% of the global total, its fertilizer use is more than 30% of the fertilizer used around the world (Yan *et al.*, 2008).

On average, in 2015 fertilizer application in China was approximately 446 kg/ha of cropland (229 kg N/ha; 116 kg P_2O_5/ha; 101 kg K_2O/ha) (FAO, 2017); this is much higher than the recommended upper limit and greatly exceeds fertilizer use in many developed countries. On the North China Plain, the use of N and P fertilizer is reported to be 588 and 92 kg/ha/year, which is 66 and 135 percent more than the crops can assimilate (Vitousek *et al.*, 2009). This excessive use of fertilizer directly endangers soil resources and causes environmental pollution (Sun *et al.*, 2012). Severe environmental degradation is already evident in many of China's rivers and lakes and is causing real concern at all levels of society.

Not surprisingly, the largest portion of fertilizer is used for the production of globally important crops such as wheat, rice and maize (Figure 2.8). However, horticulture (fruit, vegetables and flowers) is generally the most intensive user of fertilizers (and also pesticides), although it accounts for only small proportion of cropped area – typically less than 1% in most countries – with China a notable exception with close to 10%.

FIGURE 2.8 Distribution of fertilizer use by crop at the global level: 2010-2011/11



Source: IFA, 2013.

FAO (Alexandratos and Bruinsma, 2012) predicts a 58% increase in total fertilizer use from 2002/2007 to 2050 in a business-as-usual scenario. At the same time, it estimates that growth could be reduced to 17% with 'efficiencies' in nutrient use derived from better fertilizer technologies and application practices. However, the projected patterns of fertilizer use are markedly different in OECD countries, transitional economies such as China and Brazil and developing countries.

In OECD countries, it is expected that: i) increasing efficiencies will further lower farm chemical inputs and exports, due to higher input prices that reflect increasing oil prices; ii) policies and incentives will encourage a greater use of biowaste and bioenergy feedstock on farms; iii) improvements in precision farming will reduce the demand for chemical inputs; iv) there will be greater public pressure to reduce the health risks arising from agricultural pollutants, forcing farmers to adopt better practices; v) a move to decouple subsidies from agricultural production will occur; vi) farmers' behavior will change to comply with national water quality policies and as a result of education and the provision of information.

A simplistic forecast is that transitional and developing countries will follow the same path that OECD countries have charted in the past: intensification and increased (and inefficient) input use to maximize crop and livestock production. This will be moderated by increasing costs of energy and inputs, notably for N-fertilizer (which is highly dependent on oil prices) and for inorganic (rock) phosphate, of which there is a finite supply. Estimates of the time of peak phosphorous have recently been revised to 2035, after which prices can be expected to rise rapidly as global stocks fall (Cordell, Drangert and White, 2009).

2.2.4 Trends in pesticide use

Pesticides include insecticides, herbicides, fungicides and plant regulators. Humans have sought to control crop pests since the Ancient Greeks used sulphur as a fungicide. Today, pesticide production is a multi-billion-dollar industry and production is steadily moving from the OECD to transitional and developing countries. When improperly selected and managed, pesticides can pollute water with toxic substances that can affect humans. Pesticides may also affect biodiversity by killing weeds and insects, with negative impacts up the food chain.

Inorganic compounds were commonly used, at relatively low levels of intensity, to control agricultural pests (insects, plant diseases and weeds) until 1945. Since World War II, farmers have widely used organic chemical compounds as insecticides, starting with

the highly toxic and persistent organochlorines. DDT and most other organochlorine compounds were banned in the USA in 1972, leading to a slow ripple of regulation through the rest of the world. Organochlorines were banned in China in 1983. Organochlorine compounds were swiftly and progressively replaced by shorter-lived organophosphate products that, in general, do not accumulate in the food chain. However, as knowledge of ecology improves, it has become clear that organophosphate compounds can cause considerable unintended harm. As a result, many of the more toxic compounds have been progressively removed from the market in industrialized countries.

Although considerable use of older pesticides persists, the trend in the developed world is toward using newer pesticides that are more selective, less toxic to humans and the environment, and require less applications per hectare to be effective. A small but growing percentage of these are biopesticides, which are derived from natural materials such as animals, plants, bacteria and certain minerals.

BOX 2.2 | Examples of chemical pesticides

Organochlorine insecticides were commonly used in the past, but many have been removed from the market due to their health and environmental effects and their persistence in the environment (e.g. DDT and chlordane).

Organophosphate pesticides affect the nervous system by disrupting the enzyme that regulates acetylcholine, a neurotransmitter. Most organophosphates are insecticides. They were developed during the early 20th century, but their effects on insects, which are similar to their effects on humans, were only discovered in 1932. Some are very poisonous (they were developed in World War II as nerve agents). However, they are usually not persistent in the environment.

Carbamate pesticides affect the nervous system by disrupting the enzyme that regulates acetylcholine, a neurotransmitter. The enzyme effects are usually reversible. There are several subgroups within the carbamates.

Pyrethroid pesticides are a synthetic version of the naturally-occurring pesticide pyrethrin, which is found in chrysanthemums. They have been modified to increase their stability in the environment. Some synthetic pyrethroids are toxic to the nervous system.

Neonicotinoids affect the central nervous system of insects. They have been associated with some bee kill incidents. Neonicotinoid pesticide products are applied to leaves and are used to treat seeds. They can accumulate in the pollen and nectar of treated plants, which may be a source of exposure to pollinators.

BOX 2.3 Biopesticides

Biopesticides include micirobials, botanicals and semi-chemicals. Microbial pesticides consist of micro-organisms (bacteria, viruses, fungi) and their intermediate metabolites as the active agent. The most widely used microbial pesticides are subspecies and strains of Bacillus thuringiensis, or Bt. Each strain of this bacterium produces a different mix of proteins, and specifically kills one or a few related species of insect larvae. Other biopesticides are naturally occurring biochemical substances that control pests by non-toxic mechanisms, such as insect sex pheromones that interfere with mating as well as various scented plant extracts that attract insect pests to traps (Zhang and Pang, 2009).

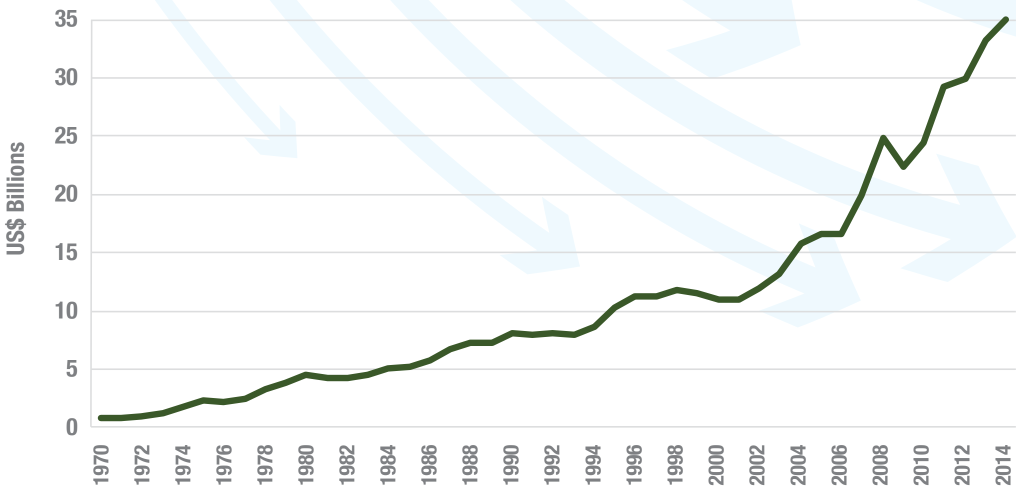
Some advantages of biopesticides, as compared to conventional chemicals, include good control of target pests, with very limited (or unknown) dangers for humans and non- target species. In addition, pest resistance appears to be slow to develop (Yang, 2001).

The biopesticide industry has been developing rapidly in China since the 1990s, with a growth rate of 10% to 20% per year. Hundreds of biopesticides have been registered worldwide, of which more than 30 are manufactured commercially (Xu, 2008). Mexico, the United States and Canada are the biggest users: their consumption of biopesticides accounts for 44% of the world total. Consumption of Europe, Asia, Oceania, Latin America, the Caribbean and Africa accounts for 20%, 13%, 11%, 9% and 3% of world consumption respectively (Qin and Kong, 2006).

Global sales of pesticides have increased dramatically over the past 50 years, with a global market worth more than USD 35 billion per year at current prices (Figure 2.9) (FAO, 2016a). The proportion of herbicides has increased, while the relative proportion of insecticides has declined significantly over recent decades, stabilizing more recently. The proportion of fungicides in use (18-24%) seems to fluctuate from year to year, reflecting variability in climatic conditions and market prices. China, the United States, France, Brazil and Japan are the largest pesticide producers, consumers or traders in the world (FAO, 2016a).

Synthetic pesticides are typically manufactured from petrochemical or inorganic raw materials and thus pesticide prices also track the oil price. It is difficult to obtain current information on pesticide pricing, but it is likely that rising sales exaggerate the growth in the volume of use in recent years as oil prices have risen over the same period. Prices for the herbicide glyphosate across the globe have increased by anywhere between 100% and nearly 500% since 2006, reflecting a number of other factors than the oil price.

FIGURE 2.9 Value of global pesticide trade, 1970–2014

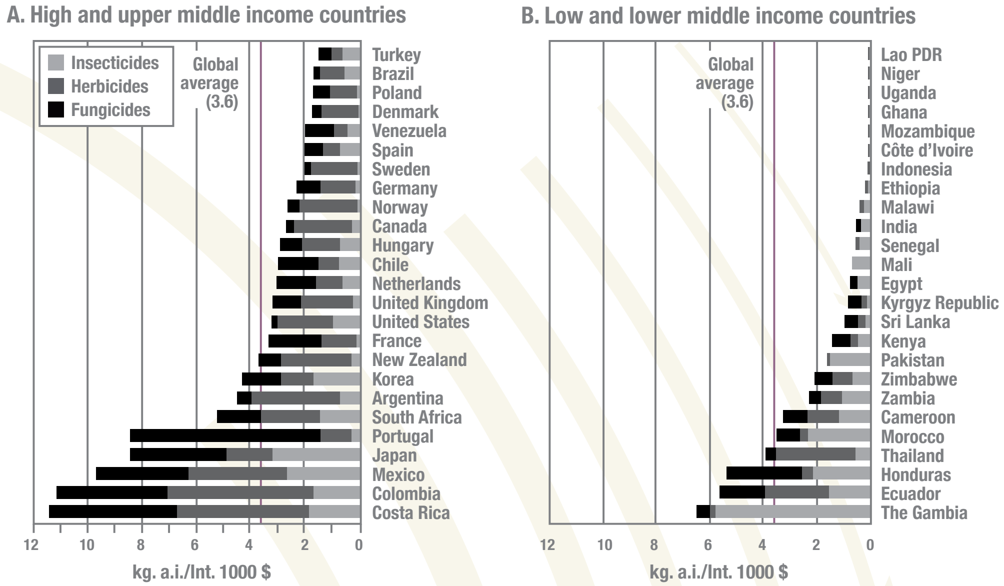


Source: FAO, 2016a.

Recent estimates of pesticide consumption based on FAO data can be seen in Figure 2.10. The figure compares, for a selection of 50 countries, the average intensity of pesticide use in terms of kilograms (kg) of active ingredients (a.i.) per 1 000 international dollars (unit) of crop output. The average country used 3.6 kg per unit of crop output and 3.2 kg of active ingredients per hectare of cropland, but levels vary widely between countries. High and upper middle-income countries (Figure 2.10.A) used much greater quantities of pesticides to produce the same quantity of crop output than did low and lower middle-income countries (Figure 2.10.B). Higher income countries thus had, on average, much lower pesticide productivity. Furthermore, crop output per hectare has generally increased less than pesticide use per hectare, on average a 1.8% increase in pesticide use per hectare has only translated into a 1% increase in crop output per hectare (Schreinemachers and Tipraqsa, 2012).

Future prospects are worrying as climate change is likely to enhance favourable conditions for pests and diseases of agricultural crops with higher temperatures, higher humidity, more variable rainfall and much more variable runoff in the semi-arid and humid tropics (FAO, 2011). Those conditions may further increase the demand for pesticides.

FIGURE 2.10 Agricultural pesticide use per unit of crop output for a selection of 25 high and upper middle and 25 low and lower middle-income countries



Note: The figure is based on the three most recent years of available data.
Source: Schreinemachers and Tipraqsa, 2012.

In the process of land use intensification, developing countries have increasingly adopted a pest management approach that centres on the use of synthetic pesticides. As a result, several developing countries have undergone double-digit growth in terms of the intensity of pesticide use, though sometimes from a low base level (Schreinemachers and Tipraqsa, 2012). The fast rate of this growth and the reliance on broad-spectrum pesticides in the context of a weak institutional framework, weak rule enforcement and a limited awareness among farmers regarding the use of hazardous chemicals, pose enormous challenges to managing pesticides in a safe and sustainable manner. Indeed, currently millions of tonnes of pesticide's active ingredients are used in agriculture, 25% of which are used in developing countries where 99% of deaths due to pesticides occur (WHO, 2010).

2.2.5 Livestock production

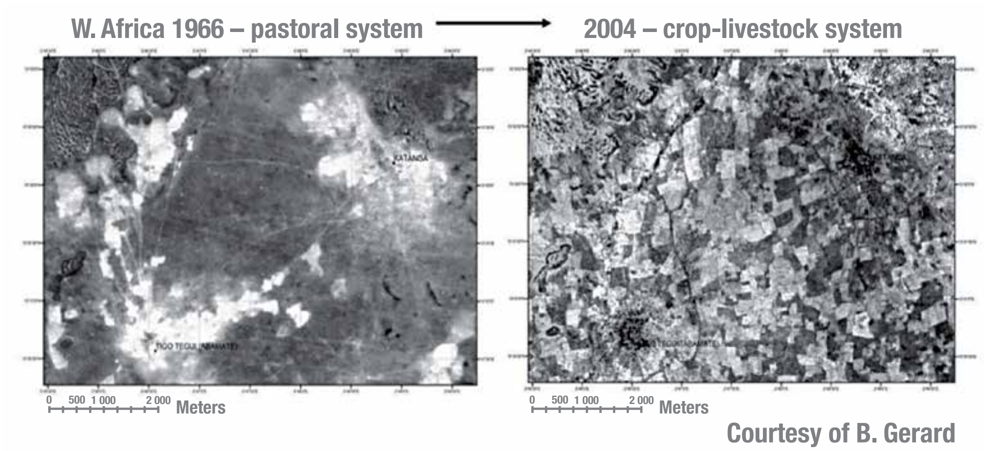
The livestock sector is one of the top three contributors to the most serious environmental problems, including water pollution, at every scale from local to global. Livestock production accounts for 70 percent of all agricultural land and 30 percent of the land surface of the planet (FAO, 2006).

Growing population and urbanization, together with higher incomes and changing diets, are rapidly increasing the demand for meat and dairy products. In response, the global production of meat is projected to more than double from 229 million tonnes in 1999/01 to 465 million tonnes in 2050 (FAO, 2006). Since 1981, global milk production has doubled to 700 million tonnes per year, with 75% of the increase generated in developing countries and 80% generated by smallholders, (McDermott, 2010). Global production is projected to grow to 1 043 million tonnes/year in 2050.

Animal production is changing locations to be closer to urban consumption hubs and to the sources of feedstuff for livestock, be it feed-crop areas, or transport and trade centers where feed is imported. There has also been a shift in species, with production of monogastric species (pigs and poultry, mostly produced in industrial units) growing rapidly, while the growth of ruminant production (cattle, sheep and goats, often raised extensively) is slowing down (FAO, 2006). As a result of these shifts, the livestock sector is entering into more and direct competition for scarce land, water and other natural resources.

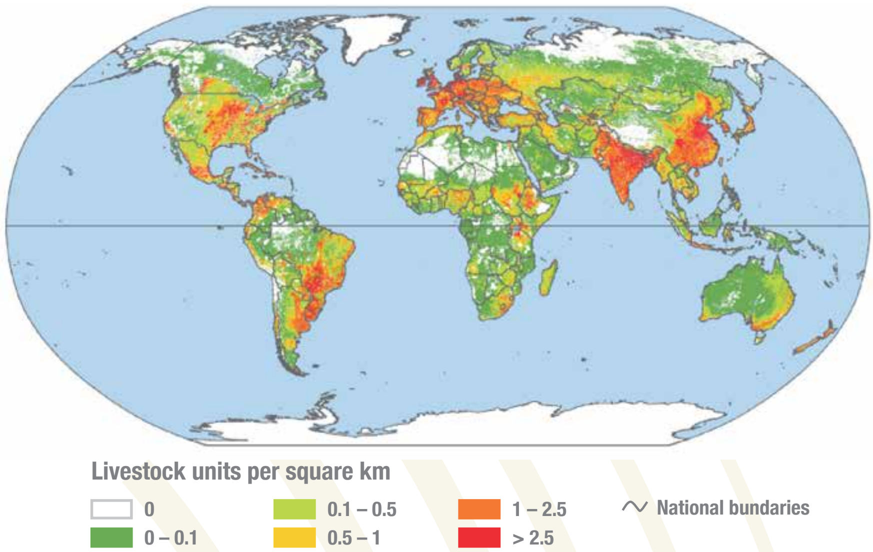
The intensity with which the sector uses land is extremely variable. Of the 3.9 billion hectares used for livestock production, 0.5 are cropped, generally intensively managed; 1.4 are pasture with relatively high productivity; and the remaining 2.0 billion hectares are extensive pastures with relatively low productivity. The trend is to transform extensive pastoral systems into intensive crop-livestock management (Figure 2.11) or industrial livestock production, with a high concentration of animals fed with feed concentrates that are not produced locally (FAO, 2006).

FIGURE 2.11 | An example of the changing nature of livestock systems in West Africa



Source: McDermott, 2010.

FIGURE 2.12 Estimated aggregated distribution of pigs, poultry, cattle and small ruminants



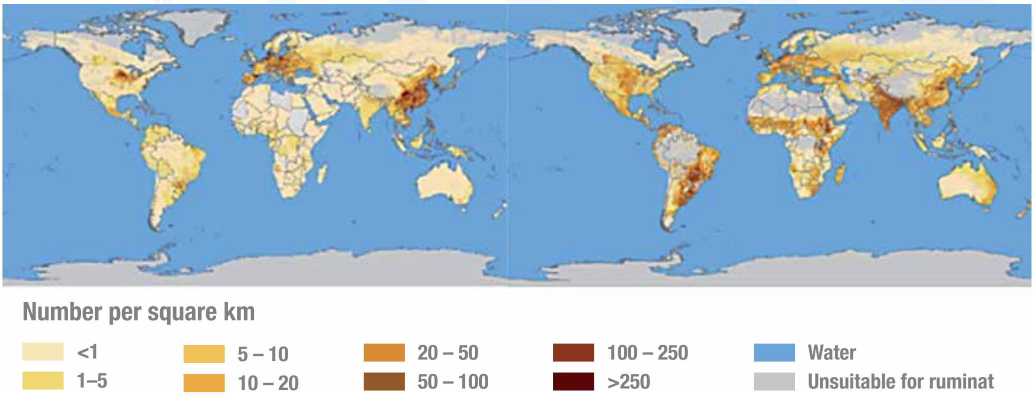
Source: FAO, 2006.

The global geographical distribution of aggregated livestock (pigs, poultry, cattle and small ruminants) is shown in Figure 2.12. There are currently five major areas of livestock concentration: i) central and eastern United States, ii) southern Brazil, Uruguay and northern Argentina, iii) Europe, iv) India and v) China. Other areas with substantial livestock concentration include eastern Africa (e.g. some parts of Ethiopia, Tanzania and Kenya), the eastern part of South Africa, south-eastern Australia and New Zealand (FAO, 2006). Different livestock species have different geographical concentration patterns (Figure 2.13). Pig production is very intense in eastern China, Europe and north-eastern USA, particularly around Iowa and Minnesota, while cattle production concentrates in India, Brazil, Argentina and Uruguay, and to a lesser extent in Europe, USA, China and Central Africa.

There is a strong and well-established relationship between meat consumption and per capita income (Figure 2.14). However, current levels of meat consumption place countries such as China, Brazil, Argentina, Russia or Mexico well above the consumption expected on the basis of per capita income, and other countries like India or Turkey fall well below the expected consumption levels.

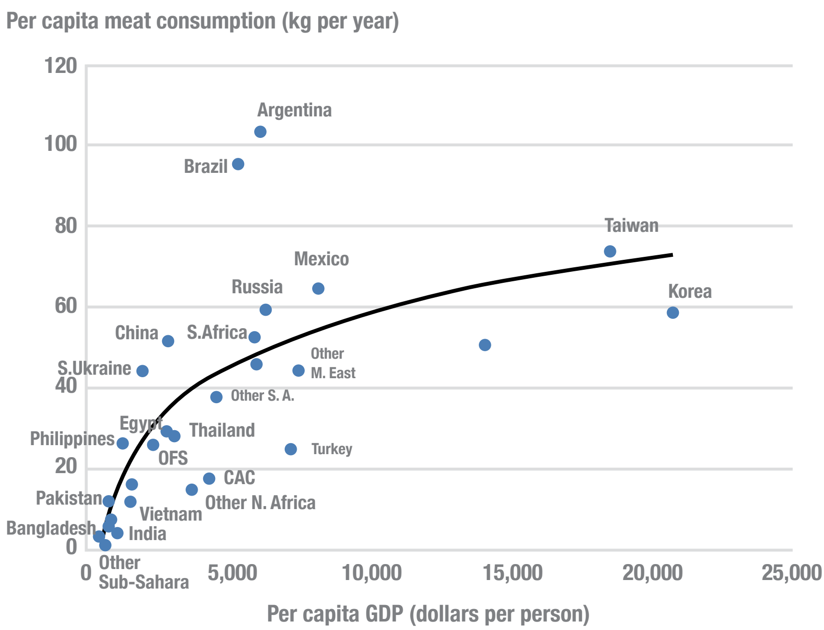
The major structural changes occurring in the livestock sector today are associated with the development of industrial and intensive livestock production systems. These systems often involve concentrating large numbers of animals in relatively small areas.

FIGURE 2.13 Distribution of pigs (left) and cattle (right) density across the world



Source: FAO, 2007.

FIGURE 2.14 Per capita meat consumption and income, in selected developing countries, 3-year average centered on 2010



Source: USDA, 2013.

Traditionally, livestock production was based on locally-available feed resources, such as crop wastes and browse, which had no value as human food. However, as livestock production has grown and intensified, it has come to depend less on locally-available

feed resources, and more on feed concentrates that are traded domestically and internationally. It is estimated that around one-third of the global cereal harvest is used to feed livestock (FAO, 2006).

FAO has estimated that 20 to 25 percent of mineral fertilizer use can be ascribed to feed production for the livestock sector. For example, 4.7 million tonnes of N fertilizer is used for feed and pastures in the USA, almost 3 million in China and more than 1.2 million in France and Germany (FAO, 2006).

There is a great deal of variation in the extent and character of livestock sector growth. China and East Asia have experienced the most impressive growth in the consumption and production of livestock. India's livestock sector continues to be dairy-oriented, using traditional feed resources and crop residues. In contrast, Argentina, Brazil and other Latin American countries have successfully expanded their domestic feed base taking advantage of low production costs and an abundance of land.

In summary, the trends in the global livestock sector can be described as follows:

- The demand and production of livestock products are increasing rapidly in developing countries and have outpaced developed countries. A few large countries such as China are taking center stage.
- This increasing demand is associated with important structural changes in livestock sectors, such as the intensification of production, geographic concentration and up-scaling of production units.
- Despite increasing grain prices, there are concomitant shifts towards poultry and pig meat relative to ruminant meat, and towards grain- or concentrate-based diets relative to low-value feed.

These trends indicate a growing pressure on the environment, and particularly on water quality as more solid and liquid excreta (manure) from livestock, nutrients, BOD, feed additives, hormones, antibiotics and heavy metals, agrochemicals and sediments flow into water as a result of the increased production of livestock and animal feed.

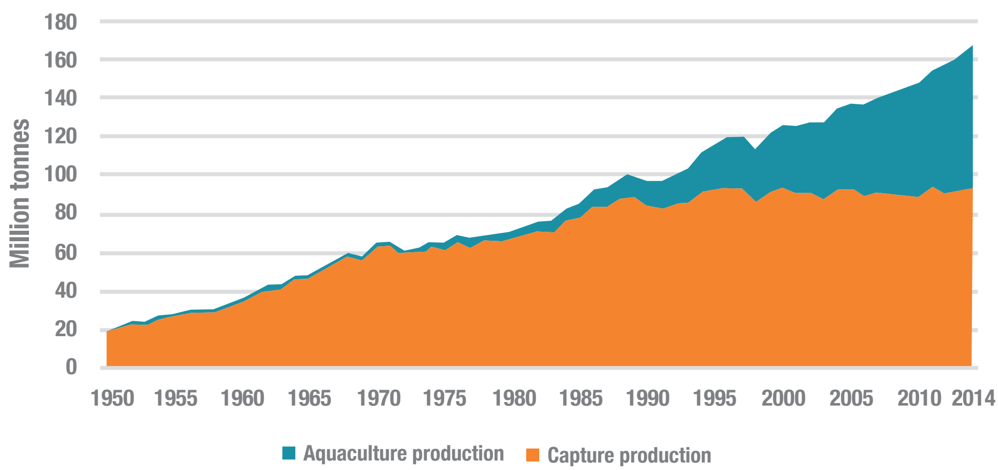
2.2.6 Aquaculture production

Over the last several decades, demand for fish and shellfish for food, feed, and other products has risen dramatically. Simultaneously, wild fish catches have plateaued since the 1990s, and the increased demand has been supplied by aquaculture, which

has expanded dramatically and is producing now nearly half of all fish consumed (Figure 2.15). Total global aquatic animal production reached 167 million tonnes in 2014 (FAO, 2016b), of which an estimated 146 million tonnes was consumed directly by humans. In the meantime, the global harvest from capture fisheries has remained constant at approximately 90 million tonnes, while aquaculture output rose from 47 to 74 mT between 2006 and 2014. Growth has occurred in marine, brackish water and freshwater environments, and the proportion of output from each sector have remained more or less consistent over this period – with between 50 and 60% of production from freshwater, 30-40% from marine conditions and 10% from brackish water environments. There has been a slight increase in the proportion of freshwater species (heading towards 60%) since 2000, while marine production has tracked down to around 30%.

At the same time, there has been a steady increase in the proportion of fed species that require externally-produced foods, with non-fed species now accounting for 30% of production as compared to 50% in 1980 (Figure 2.16). Fed and intensive aquaculture can result in an excess of faeces, uneaten feed and drugs released into water bodies. Carnivorous species, which have high value, require high inputs of fishmeal and other pelleted feeds. Many types of non-fed aquaculture (e.g. mussel farming) can filter and clean waters, but other types (e.g. intensive caged crab culture) may disrupt natural nutrient cycles and result in water quality degradation.

FIGURE 2.15 World fish production (million tonnes)



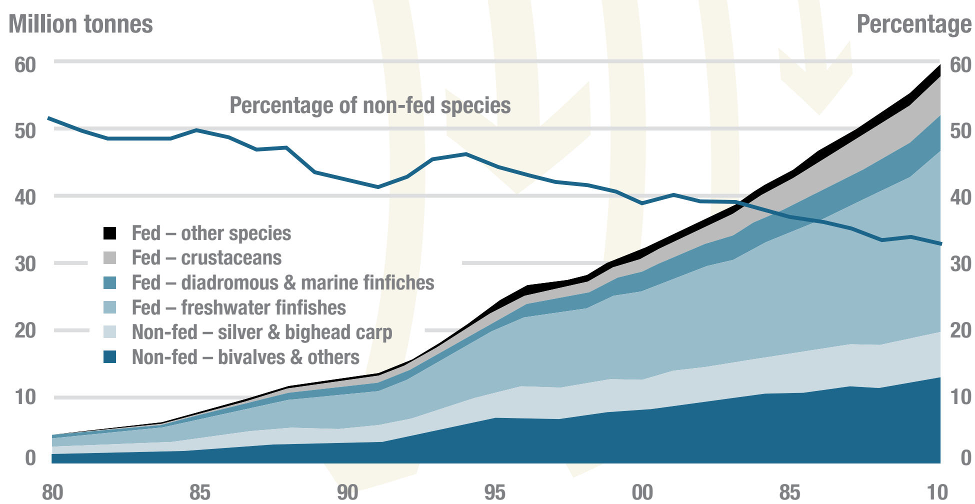
Source: FAO, 2016b.

The growth in aquaculture has overwhelmingly taken place in developing countries, which produce 91% of global output with the greatest concentration in the low-income nations. Globally there is great diversity of fish species. Freshwater species are predominantly cyprinids, tilapia and catfish, whereas diadromous fish (which can live in freshwater, brackish water and saltwater) are predominantly salmonids, milkfish and eels.

Asia generates the highest aquaculture output, representing almost 90 percent of world production, with output from China dominating at 45.5 million tonnes per year (FAO, 2016b).

Economically successful aquaculture demands a high level of water quality. Fish excreta are high in nitrate, nitrite and ammonia and flow rates through production units must be sufficient to control toxic levels and maintain dissolved oxygen levels at 6-9 mg/l. There is thus considerable natural synergy between aquaculture management and the maintenance of good water quality. Market pressure and differentiation are beginning to increase the intensity of production with an increasing concentration on single species. This has resulted in an increase in the use of medicines (antibiotics, fungicides and anti-fouling agents), which in turn pollute downstream ecosystems. Environmental impacts from aquaculture arise from the export and concentration of organic wastes (fish excreta and uneaten feeds) and from medicines. For example, large-scale shrimp culture has resulted in the physical degradation of coastal habitats through, among other factors, the conversion of mangrove forests and destruction of wetlands, salinization of agricultural and drinking water supplies and land subsidence due to groundwater

FIGURE 2.16 **Growth in aquaculture from 1980 to 2010**



Source: FAO, 2012.

abstraction. The discharge of untreated organic waste from salmon production is another concern in countries such as Norway, Scotland, Chile and Canada. However, the impact of this discharge is very localized and is limited to a few hundred metres of sea-bed. The dilution and dispersal of marine and brackish aquaculture effluents are governed by local ocean current patterns, and the nature and extent of impacts on eutrophication and fish stocks are quite variable.

This chapter has provided a summary analysis of how agricultural production systems have responded to growing demands for food over the past decades. Crops, livestock and aquaculture use much more land today than they did fifty years ago (frequently at the expense of forests or grasslands), and the use of land, water and other agricultural inputs is more intense than ever before. Large-scale monocultures in fertilized soils, intensive livestock production and fed aquaculture are becoming the rule rather than the exception in many parts of the world. As will be seen in subsequent chapters, the expansion and intensification of agricultural has contributed through different pathways (Chapter 3) to increased loads of nutrients (Chapter 4), pesticides (Chapter 5), sediments (Chapter 6), salts (Chapter 7), organic matter pathogens and other pollutants of emerging concern to water bodies (Chapter 8), with unprecedented impacts on human health and ecosystems. Nevertheless, as shown in Chapters 10 and 11, there are validated and emerging solutions that can pave the way towards a more sustainable intensification of agriculture to feed the world without further compromising the productivity and safety of agro-ecosystems in the long term.

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PART II:
PRESSURE, STATE AND IMPACTS


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CHAPTER 3. AGRICULTURAL POLLUTION SOURCES AND PATHWAYS

Javier Mateo-Sagasta

With contributions from Hugh Turral

Historically, the analysis of negative impacts on aquatic ecosystems (e.g. dead zones) or on human health (e.g. blue baby syndrome) has focused attention on the nature and probable sources of the causative pollutants (e.g. nitrate from agriculture). Further investigation, often through formal research projects, helps to determine the correlations between likely contributing agents, loads and load dynamics (timing, frequency and duration) and their concentrations in water. Finally, remedial actions can be identified and implemented, although perhaps not in time to forestall substantial negative impacts. This approach can be improved if the links between pollution loads and their impacts are better understood, so that decision-makers can take preventive actions before pollution loads are sufficient to threaten human health and ecosystems. The particular challenge of agricultural pollution is to determine the source(s) of pollutants (which are frequently diffuse) and their actual contributions to the loads experienced in a lake, river, estuary or coastal zone.

This chapter introduces key concepts related to agricultural water pollution and describes the main types of pollutants arising from agricultural sources and their pathways to water. Subsequent chapters (Chapters 4-8) review specific pollutants more thoroughly and will provide global data on loads and impacts when available. Chapter 9 discusses how modelling can help to link the causes of water pollution from agriculture

to their effects (i.e. links between drivers, pressures, state change and impact) and how models can be used to plan and inform policies.

3.1 Clarifying terms and concepts

Agricultural pressure on water quality is defined here as a direct effect of agriculture expansion and intensification that can cause a change in the physicochemical characteristics of water. Pressures include the increased load of chemicals, sediments or pathogens that enter water bodies through runoff or percolation. Loads are determined as the product of concentration and flow rate, and are calculated in terms of mass per time unit (e.g. kg/day). Because loads are determined by flow and concentration over time, both components must be measured or modelled. Water abstractions and the consumptive use of water by agriculture can also affect water quality, as they reduce water quantity in rivers, lakes or aquifers, therefore increasing the concentration of existing pollutants.

The state of a water body is normally defined in physical (e.g. temperature) and chemical (e.g. concentration of nitrate) terms. The metrics of state document pollutant concentrations, pH, turbidity, temperature and similar parameters. Other metrics record the state of an ecosystem, usually through indexes that rely on indicator species. Indicator species are chosen for their sensitivity to environmental conditions. For example, frogs and toads are good indicator species because the skin of the adults is moist and permeable, allowing numerous pollutants entry into their bodies. If the chosen indicator species declines in numbers or health, it is a sign to look for detrimental influences. Nevertheless, some authors consider these indicator species to be indicators of the impact of a water quality change on ecosystems (Ferreira *et al.*, 2011; Sebastian *et al.*, 2012; UNEP, 2016). The European Commission suggests different steps to analyze the link between pressures and state change in water bodies (see box 3.1).

This book defines impact as the effect water pollution on the environment, human health and economic activities. Examples of water pollution impacts are given in Table 3.1. State and impact are separate concepts that are often combined or confused. One reason for this is that many of the impacts are not easily measurable, thus state is often used as an indicator of impact. While it is possible to determine the state of receiving waters (such as lakes, wetlands, etc.) by measuring certain indicators, it is harder to quantify the actual loads of pollution to water bodies and to link them to state and impact. Despite different attempts to quantify them through measuring or modelling (e.g. Schwarzenbach *et al.*, 2010; UNEP, 2016), the links between cause and effect are sometimes elusive, or counter intuitive.

The quantification of agricultural pollution loads, while seemingly simple, poses considerable practical challenges. National statistics (e.g. on the use of N fertilizers) can be employed to estimate pressures in broad terms. However, they would need to be sufficiently disaggregated across land use type (e.g. cropping, horticulture, pasture) to avoid presenting a misleading picture. Even when we can identify an extreme use

BOX 3.1 Pressures and state change analysis for water pollution: the European Union case

A European Commission guidance document on pressures and impacts for the Water Framework Directive (European Commission, 2003) focused on how to define and implement a work program to identify pressures, monitor their behavior and determine how to mitigate them. The document requires the integration of different sources of water pollution, in addition to agricultural sources. It notes that only significant pressures should be considered and monitored, but recognizes that investigation and research are needed to identify which pressures are actually significant, and that development of indicators and some modelling (even with inadequate data) may be required in conjunction with selective and improved monitoring.

The recommended work programme involves the following steps:

1. Screen all available information on pollution sources:
   a. collate information;
   b. produce a short list of likely key pollutants responsible for observed impacts.
2. Test for relevance:
   a. estimate concentrations in water bodies by monitoring or modelling;
   b. compare measured concentrations with benchmarks, including Environmental Quality Standards, where they exist.
3. Safety net: check if a small number of pollution sources can have a significant combined effect, or whether trends are increasing, even if a standard has not been breached.
4. Prepare a final list of relevant pollutants and their actual and target loads and concentration levels in specific river reaches, lakes and estuaries.

Table 3.1 I Examples of potential negative impacts on human health, the environment and economic activities due to water pollution from agriculture (crops, livestock and aquaculture)

Impacts on:	Examples of impacts
Health	Increased burden of disease due to reduced drinking water quality
	Increased burden of disease due to reduced bathing water quality
	Increased burden of disease due to unsafe food (contaminated fish, vegetables, etc.)
Environment	Decreased biodiversity (e.g. as a result of pesticide toxicity)
	Eutrophication and dead zones
	Visual impacts such as landscape degradation
	Bad odors (e.g. from manure)
	Diminished recreational opportunities
	Increased greenhouse gas emissions

➤

Impacts on:	Examples of impacts
Productive activities	Reduced agricultural productivity (e.g. by the use of saline drainage water)
	Reduced market value of harvested crops, if pollution acknowledged
	Reduced number of tourists in polluted areas
	Reduced fish and shellfish catches

Source: Adapted from Hernandez-Sancho *et al.*, 2015.

of a potential pollutant, say nitrogen fertilizer, it does not necessarily follow that it is a significant pressure. For example, in some water-scarce areas, due to necessarily high levels of water control, the export of soluble nitrate to groundwater or surface water is limited despite the intensive use of fertilizers because deep percolation and runoff volumes are small (Duncan *et al.*, 2008; Molden *et al.*, 2010).

3.2 Types of pollutants and agricultural sources

The chief agricultural contributors to water pollution (and the main targets for water pollution control) are nutrients, pesticides, salts, sediments, organic carbon, pathogens, heavy metals and drug residues. The relative contribution of different types of agriculture pollutants to water quality degradation is presented in Table 3.2. The importance of different types of agricultural pollution can vary, depending on the circumstances. Negative impacts, such as eutrophication, arise from combinations of stressors, which can include sediments, nutrients and organic matter. Significant portions of nutrient load may be carried by sediments into surface waters, whereas groundwater pollution results mostly from dissolved pollutants (with perhaps the exception of pathogens).

Water quality monitoring programs typically record key pollution indicators, such as concentrations of ammonium, nitrate, phosphate, oxygen (or oxygen demand), salts, pathogens (e.g. E. coli) and suspended solids, and sometimes also pesticides and heavy metals, although these are expensive to measure. Other indicators, such as temperature or pH, are recorded because they impact biota directly, or because they mediate the impacts of pollutant loading. For example, temperature plays an important role in the occurrence of algal blooms in conjunction with nitrate and phosphate loadings; it also affects the solubility of oxygen in water and high temperatures can induce anoxic waters. Many indicators need to be measured in groundwater as well as in surface water; these have historically received greater attention when that water is used by humans for drinking.

Cultivation practices can impact both surface and groundwater quality. For surface water, two types of impacts are of most concern: the loss of topsoil as a result of erosion,

Table 3.2 I Categories of major water pollutants from agriculture and the relative contribution from different agricultural production systems

Pollutant category	Indicators/examples	Relative contribution by:		
		Crops	Livestock	Aquaculture
Nutrients	Primarily nitrogen and phosphorus present in chemical and organic fertilizers as well as animal excreta and normally found in water as nitrate, ammonia or phosphate	***	***	*
Pesticides	Herbicides, insecticides, fungicides and bactericides, including organophosphates, carbamates, pyrethroids, organochlorine pesticides and others. Many, such as DDT, are banned in most countries but are still being used illegally and persistently	***	–	–
Salts	Ions of sodium, chloride, potassium, magnesium, sulphate, calcium and bicarbonate. These are measured in water, either directly as total dissolved solids or indirectly as electric conductivity	***	*	*
Sediment	Measured in water as total suspended solids or nephelometric turbidity units – especially from pond drainage during harvesting	***	***	*
Organic matter	Chemical or biochemical oxygen-demanding substances (e.g. organic materials such as plant matter and livestock excreta), which use up dissolved oxygen in water when they degrade	***	***	*
Pathogens	Bacteria and pathogen indicators, e.g. *Escherichia coli*, total coliforms, faecal coliforms and enterococci	*	***	*
Metals	E.g. selenium, lead, copper, mercury, arsenic and manganese	*	*	*
Emerging pollutants	E.g. drug residues, hormones and feed additives	–	***	**

Source: Author's descriptions.

and its subsequent deposit in water courses and lakes; and runoff of nutrients (N and P) from an excessive use of fertilizer. Pesticide runoff can be locally very relevant when pesticides are applied incorrectly or when rain washes them away. Another local water quality problem occurs when farmers attempt to desalinize irrigated fields by applying large amounts of leaching water (Cañedo-Argüelles *et al.*, 2013). In addition, pumping

of groundwater can induce saline intrusion in coastal aquifers or the migration of low quality water from underlying aquifers (IGRAC, 2009).

The livestock sector is probably the largest source of water pollution, if the land used for feed crops is taken into account (FAO, 2006). The major sources of pollution are animal waste and uneaten feed, land used for feed crops, and tanneries. Animal manure and slurries contain large amounts of pathogens, ammonia and phosphate and have high biological oxygen demand (BOD) (FAO, 2006). The sources of livestock pollution are generally diffuse, but they can be concentrated (e.g. slurry management under zero grazing and feedlots). In many parts of the world, particularly in drylands, overgrazing has caused land degradation and erosion, which has in turn increased sediment loads to water (Doetterl *et al.*, 2012. Heavy metals can concentrate in livestock enterprises (e.g. copper in pig production) resulting in point-source contamination of soils and water.

Pollutants produced by aquaculture, as for livestock, chiefly originate from the use of various inputs, and the excreta and secretions of the aquaculture organism. The major pollutants and pollutant indicators include (Li and Chen, 2013):

- non-ionic toxic ammonia (NH_3), for which the major sources are the faeces of aquatic organisms, feed residues and dead algae;
- nitrite: an intermediate product during the conversion of ammonia into nitrate;
- phosphorus: the major source is phosphorus in feed;
- other chemical residues: bactericides, fungicides and parasite-killing agents, algaecides, herbicides, molluscicides and growth hormones;
- turbidity: suspended particles can be vectors for pathogens and viruses;
- chemical oxygen demand (COD) or biological oxygen demand (BOD): an abundance of organic matter can lead to oxygen deficiency, which can kill fish and cause the release of poisonous or harmful substances, such as ammonia and hydrogen sulfide.

Some important contaminants, such as arsenic (best known in groundwater in Bangladesh, India and Cambodia) or selenium are released from natural sources as a result of extracting large quantities of water (mostly groundwater) for irrigation (Mateo-Sagasta and Burke, 2010).

From a health perspective, the agropollutants of greatest relevance are pathogens from livestock, pesticides and nitrates in groundwater (particularly when the water is used for drinking purposes), trace metallic elements (including arsenic) and emerging pollutants, including antibiotics and antibiotic-resistant genes excreted by livestock. From an environmental perspective, eutrophication due to an excess of nutrients, salinization induced by agriculture and decomposable organic matter (mainly from livestock) in surface waters are probably the most relevant factors (Table 3.3).

3.3 Pollution pathways

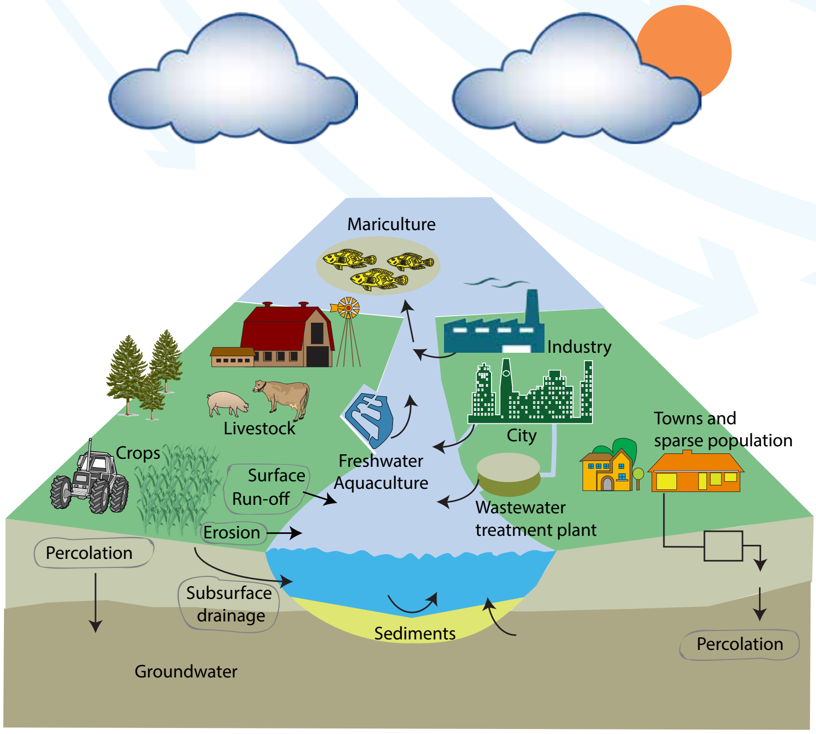
Agricultural and municipal pollution are closely linked to the hydrologic cycle. General sources and pathways for point source (PS) and non-point source (NPS) pollution of water are illustrated in Figure 3.1 below and discussed in box 3.2.

Table 3.3 I Major global water quality issues related to agriculture in inland waters and their relevance to human health and the environment

Issue	Environmental relevance	Health relevance	Water body			
			Rivers	Lakes	Reservoirs	Groundwater
Faecal pathogens	+	+++	•••	•	•	••
Suspended solids	++	+	••	Na	•	Na
Decomposable organic matter	+++	+	•••	•	••	•
Eutrophication	+++	+	•	••	•••	Na
Nitrate	++	+	•	–	–	•••
Salinization	+++	++	•	–	•	•••
Trace metallic elements and arsenic	+	+++	••	••	••	••
Pesticides	+	+++	••	••	••	•••
Acidification	++	++	•	•	••	–

Na: not applicable; Health relevance: + (low) to +++ (high); Occurrence of degradation issues: • (low) to • • • (high); –: (rare occurrence).
Source: Adapted from Meybeck, 2004.

FIGURE 3.1 Water pollution sources and pathways



Source: IWMI.

BOX 3.2 Point and non-point source pollution (adapted from FAO, 2013)

We tend to think of environmentally or health-damaging pollution as wastewater that comes from industries, cities, towns and other sources where polluted water is discharged through a pipe or channel. This type of pollution is known as 'point source' (PS) pollution. Because it is discharged through pipes or channels, it can be easily monitored for quantity and water quality (physical and chemical properties) and can be collected and treated before it is discharged into rivers, lakes or reservoirs. Some agricultural systems, such as big industrial livestock farms (e.g. pigs, poultry), slaughterhouses and intensive aquaculture farms, can be considered point pollution sources.

Other types of land-use activities, such as road construction, mine drainage, rainwater runoff from city streets (which is not collected in storm drains), from agriculture and from many rural villages, produce water pollution that does not come from any specific pipe or channel, but instead tends to be dispersed across the landscape. This type of pollution, which cannot be easily measured because of its diffuse nature, is known as 'non-point source' (NPS) pollution, or diffuse pollution.

Although this publication will concentrate on water quality impacts that arise from agriculture, pollution loadings have many sources and follow pathways through air, surface water and groundwater. It is important to note that aerial pathways are significant, and cause secondary concentration in the hydrologic system (Monteith *et al.*, 2007; Howarth, 2008).

In addition to aerial pathways, agricultural pollutants impact aquatic and marine ecosystems as a result of export from farms, transportation along hydrological pathways and concentration in water bodies. Typical water pollution pathways are: i) from soil solution to deep percolation and groundwater recharge; ii) from runoff, drainage water and floods to streams, rivers and estuaries; iii) from natural or human induced soil erosion to sediment-rich streams.

Some pollutants, such as nitrate or ammonium, are highly soluble and are easily lost from the soil profile through leaching or runoff. Other pollutants (e.g. phosphate or some pesticides and pesticide transformation products) tend to be transported by attaching themselves to suspended soil particles, and can thus concentrate where sediments are deposited (lakes, wetlands, estuaries and coastal zones), providing a reservoir of contaminant that can be re-released in episodic or continuous patterns. Hydrology is therefore the link between the source of pollution and state change in a water body.

Surface drainage increases both soluble and sediment-borne transport of nutrients and pesticides by creating a more direct path to waterways. In contrast, by shifting the major pathway for excess precipitation from surface runoff to subsurface flow, tile drainage has been shown to reduce losses of sediment, phosphorus, and pesticides from agricultural land in the north-western USA (Blann *et al.*, 2009). The hydrology of subsurface drainage in conjunction with surface drainage has more complex implications for sediment and nutrient loading.

Animal waste can enter surface and groundwater from communal farms and feedlots both accidentally and on purpose. Rainfall causes animal waste to run off into surface streams and groundwater. Also, some farmers deliberately clean their animal pens directly into rivers and canals (FAO, 2006).

This chapter has introduced some basic concepts that will help to better understand subsequent chapters where, on the one hand, specific pollutants are analyzed more thoroughly – nutrients (Chapter 4), pesticides (Chapter 5), salts (Chapter 6), sediments (Chapter 7), and organic matter, pathogens and emerging pollutants (Chapter 8) – and, on the other hand, water quality models to analyze sources and effects of water pollution are discussed (Chapter 9).

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CHAPTER 4. NUTRIENTS

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With the growth of intensive and expansive agricultural production in recent years, the use of chemical fertilizers has increased rapidly. Fertilizers are used to supplement soil fertility and to satisfy the demand for high-yielding crops by replacing soil nutrients taken by harvested crops. Agricultural nutrients come in many forms. They can be found in natural sources, such as manures, compost, biological nitrogen fixation (BNF) of legumes, and green manures. They can also be found in chemical or mined sources, such as commercial nitrogen, phosphate and potassium fertilizers. Manure and other organic fertilizers have the added benefit of providing organic matter to the soil, which may improve nutrient cycling, soil structure, aeration, soil moisture-holding capacity and water infiltration.

Nevertheless, the growing use of fertilizers can also lead to the degradation of aquatic resources such as lakes, rivers and marine water resources. The rapid expansion and intensification of livestock production has also contributed to the pollution of water resources. Water pollution from nutrients occurs when fertilizers are applied at a greater rate than they are fixed by soils or taken up by plants; when they leach into groundwater or move via surface runoff into waterways, resulting in costly environmental and health issues. When nutrients drain into rivers, lakes and streams, they can cause eutrophication and accelerate the growth of algae and aquatic plants, which, when decay, reduces the oxygen on which other aquatic life depends. An overgrowth of certain algae species can

also produce high levels of toxins and bacteria that are harmful for humans if they come into contact with contaminated water or consume tainted fish or shellfish.

This chapter focuses on nitrogen (N) and phosphorus (P) nutrient loads that result from agricultural practices, and their impacts on aquatic ecosystems. On the one hand, N and P are essential for proper crop development, high yields and associated social benefits such as income generation and livelihood provision. However, excessive nutrient use can lead to the contamination of soil and water. The next sections review the main trends in fertilizer use and the resulting effects on surface water and groundwater, followed by a discussion of the impacts of nutrient-based water pollution on human health and the environment.

4.1 Use of nutrients (N and P) in agriculture

Agriculture is the single largest user of freshwater globally. One of the root causes of degraded surface water and groundwater is excess nutrients from agricultural production. A major focus of crop breeding in recent years has been to increase yields by improving the growth response to nitrogen. In tandem, other nutrients, notably phosphorous, potassium and sometimes sulphur, have become limiting factors to yield increases, and have thus been added using inorganic products to supplement traditional methods of recycling through manuring, fallowing and crop rotation.

Plant nutrients are typically classified according to the elements that are required or plant growth and development. These include essential and other mineral elements and are typically referred to as macronutrients (e.g. C, H, O, N, P, K) and micronutrients (e.g. Fe, Zn, Mn, Cu, B, Mo) (Mengel, 1982; Frageria *et al.*; 1995). The focus of the chapter will be on P and N compounds because of their demonstrated effects on eutrophication and hypoxia in surface and coastal waters (Rabalais *et al.*, 2009) and ground water pollution by nitrates.

The world currently consumes ten times more mineral fertilizer than it did in the 1960s (FAO, 2017a), and global demand for nitrogen fertilizer is expected to increase from 110 million tonnes in 2015 to 119 million tonnes in 2020 (FAO, 2017a). In contrast, although the use of phosphorus fertilizers was in line with that of nitrogen until the 1980s, it has stalled since 1989. This global growth of fertilizers consumption masks a growing gap in access to fertilizers between developed and developing (mainly tropical) countries. Only 10 percent of the world's croplands are found in developed countries, but they account for 32 percent of the global nitrogen surplus and for 40 percent of the phosphorus surplus (Panuelas *et al.*, 2013).

Global consumption of the three main fertilizer nutrients – nitrogen (N), phosphorus (expressed as phosphate, P_2O_5), and potassium (expressed as potash, K_2O) – was estimated to have reached 186.7 million tonnes in 2016. Table 4.1 shows that consumption for N, P_2O_5, and

Table 4.1 I Global consumption of fertilizer nutrients (2015-2020) (thousand tonnes)

Year	2015	2016	2017	2018	2019	2020
Nitrogen (N)	110 027	111 575	113 607	115 376	117 116	118 763
Phosphate (P_2O_5)	41 151	41 945	43 195	44 120	45 013	45 858
Potash (K_2O)	32 838	33 149	34 048	34 894	35 978	37 042
Total (N+ P_2O_5+ K_2O)	184 017	186 668	190 850	194 390	198 107	201 663

Source: FAO, 2017a.

K_2O is forecast to grown by an average of 1.5, 2.2, and 2.4 percent, respectively, each year from 2015 to 2020, and is expected to reach 201,663 thousand tonnes by the end of 2020 (FAO, 2017b).

The overuse or misuse of mineral fertilizers and manures in agriculture severely affects the quality of water and soil resources. Beyond the farm boundary, nutrients can cause contamination and often eutrophication of water bodies through surface run off and leaching of nutrients from agricultural farmland. In excess amounts, these nutrients overstimulate the growth of weeds and algae in surface waters and can lead to serious algae blooms and oxygen depletion in rivers and lakes, which may create adverse impacts – such as fish kills – for the environment and human livelihoods. Aquaculture (particularly fed aquaculture) can also be a locally relevant source of nutrient pollution, e.g. in Bangladesh (Alam, 2001).

BOX 4.1 | Nutrient leaching and water pollution

Nutrient leaching depends on several factors, including fertilization level, type, and timing of fertilizer application; the method of application; properties of soils (i.e., pH, structure and organic matter content); types of crops and their fertilizer requirements; method of cultivation and agronomic practices; and the level of animal production. Weather conditions and catchment land use can also have a crucial impact on the intensity and quantity of nitrogen leaching.

An insufficient amount of potassium reduces nitrogen uptake by plants and thereby may increase nitrogen leaching from the soil. Insufficient availability of phosphorus leads to decreased plant biomass, even when nitrogen is in an optimal concentration compared to plant requirements. However, the relationship between these elements is not well understood in terms of nutrient leaching in agricultural areas. ➤

➤ Nitrogen, in particular the very soluble nitrate, is easily dissolved in percolating water. Phosphorus is less mobile and reaches surface water due to erosion being bound to soil particles. These different pathways complicate water quality protection, because the elimination of one source may aggravate another. For example, the reduction of fertilization level or one of the elements may not reduce the leaching of nutrients as a result of the unfavorable ratio of nutrients in the soil. A deficiency of phosphorus or potassium limits the uptake of nitrogen by plants, even when the nitrogen level is sufficient. This suggests that, at a low level of fertilization due to a shortage of potassium and phosphorus, there may occur a loss of nitrogen, which results in water and soil pollution. This could affect two-thirds of the world's agricultural land, where potassium deficiency occurs (Lawniczak *et al.*, 2015).

4.1.1 Nitrogen from croplands

In terrestrial ecosystems, nitrogen must be 'fixed' or bound into a reactive form before animals and plants can use it, because in the inert form (N_2) it is chemically unavailable to most living organisms. The reactive forms of nitrogen (Nr) (all forms of N except N_2) are actually far more important to life are (Sutton *et al.*, 2013). These include single or double-bonded nitrogenous compounds such as ammonium and nitrates. Because only a small part of the Earth's biota can convert N_2 to Nr, reactive nitrogen is the limiting nutrient in most natural ecosystems and, almost always, in agricultural systems (Erisman *et al.*, 2015).

Over 90 percent of soil N takes the form of organic N. While there are 13 major nitrogenous fertilizers in use around the world, urea is the dominant compound. Other formulations are used where conditions or crop needs require them. For example, ammonium sulphate is preferred for alkali soils as it slightly acidifies the soil and can bring the pH into a range where more trace elements are available to the plant. Microbial activity determines the transformation of organic and inorganic nitrogen in the soil, and the pathway is principally governed by redox conditions.

In addition to N loss in the form of NO_3-, N can also drain into waterways as soluble NH_4+ or NH_4+ attached to sediments. The pathway and quantity of N loss from agricultural systems can be highly variable and, because it is determined by prevailing conditions, significant changes can occur within just a few hours or days.

The global use of nitrogen fertilizer (both mineral and organic) for agriculture has been increasing in most regions in recent decades (Lu & Tian, 2013). Globally, the application of

mineral nitrogen fertilizers to croplands is currently estimated at around 115 million tonnes N per year. Moreover, the annual human-caused biological fixation of atmospheric N_2 by cultivated leguminous crops and rice is currently estimated at around 65 million tonnes N per year. Approximately 22 percent of human nitrogen inputs end up accumulating in soils and biomass, whereas 35 percent enters the oceans via atmospheric deposition (17 percent) and leaching via river runoff (18 percent) (Panuelas *et al.*, 2013).

Table 4.2 forecasts global and regional nitrogen fertilizer demand against the compound annual growth rate from 2015 to 2020. Sub-Saharan Africa and Latin America and the Caribbean are expected to have a high compound annual growth rate, accounting for 4.83% and 4.09% respectively. In contrast, West Europe and North America have the lowest annual growth, accounting for -0.99% and 0.37% respectively (FAO, 2017b).

Table 4.2 I Global and regional nitrogen fertilizer demand forecasts (thousand tonnes N) and compound annual growth rate (CAGR), 2015 to 2020

	2015	2016	2017	2018	2019	2020	CAGR (%)
WORLD	110 027	111 575	113 607	115 376	117 116	118 763	1.54
AFRICA	3 573	3 641	3 788	3 964	4 126	4 302	3.78
North Africa	*1 835*	*1 870*	*1 929*	*1 984*	*2 042*	*2 102*	*2.75*
Sub-Saharan Africa	*1 738*	*1 772*	*1 860*	*1 980*	*2 084*	*2 201*	*4.83*
AMERICAS	22 506	23 030	23 379	23 768	24 169	24 564	1.77
North America	*14 434*	*14 517*	*14 552*	*14 612*	*14 667*	*14 701*	*0.37*
Latin America & Caribbean	*8 072*	*8 513*	*8 828*	*9 157*	*9 501*	*9 863*	*4.09*
ASIA	66 294	67 082	68 446	69 493	70 525	71 476	1.52
West Asia	*2 982*	*3 048*	*3 127*	*3 213*	*3 302*	*3 395*	*2.63*
South Asia	*22 273*	*22 525*	*23 430*	*24 002*	*24 645*	*25 191*	*2.49*
East Asia	*41 039*	*41 509*	*41 888*	*42 278*	*42 578*	*42 890*	*0.89*

	2015	2016	2017	2018	2019	2020	CAGR (%)
EUROPE	15 874	16 016	16 161	16 290	16 407	16 504	0.78
Central Europe	*2 945*	*3 044*	*3 121*	*3 200*	*3 282*	*3 343*	*2.57*
West Europe	*8 448*	*8 370*	*8 315*	*8 236*	*8 139*	*8 038*	*-0.99*
East Europe & Central Asia	*4 481*	*4 602*	*4 725*	*4 854*	*4 986*	*5 123*	*2.71*
Oceania	1 779	1 806	1 833	1 861	1 888	1 917	1.50

Source: FAO, 2017b.

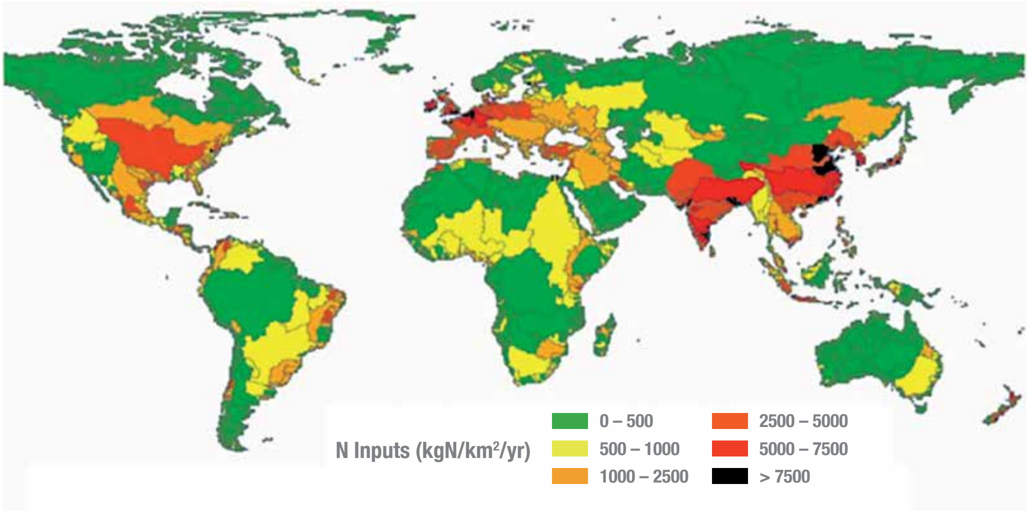
Figure 4.1 below shows the estimated net inputs of diffuse N (runoff or leached from land) in river catchments around the world, with Europe and East Asia accounting for the highest values. Watersheds where anthropogenic inputs are equal to or exceed natural inputs are calculated using data from the Global NEWS[4] database and by statistically relating the total anthropogenic inputs of reactive nitrogen to the output through the hydrosphere (see Sutton *et al.*, 2013 for supplemental material). An estimate for the continental USA in the 1990s (Howarth, 2002) indicated that returns to water were close to 20 percent of total applied agricultural nitrogen, with up to 25 percent lost in gaseous form. When returns from animal and human wastes are considered, as much as 44 percent is estimated to be leached to surface and groundwater, and subsequently transported to lakes, estuaries and the sea.

4.1.2 Phosphorus from croplands

Phosphorus is a key nutrient that stimulates the growth of aquatic organisms in water bodies, but in excessive quantities it has a fertilizing effect that affects both the ecosystem and water quality as whole (EC, 2014). Phosphorus is naturally present in surface water as a result of the mineralization of vegetable and animal residue, or due to anthropogenic pollution, e.g. diffuse sources from agriculture, untreated or insufficiently treated municipal waters and the use of polyphosphate detergents (EC, 2014).

4 Global *NEWS* is an international, interdisciplinary scientific taskforce, focused on understanding the relationship between human activity and coastal nutrient enrichment. It was formed in the spring of 2002 as a workgroup of UNESCO's Intergovernmental Oceanographic Commission (IOC), with co-sponsorship by UNEP, US-NSF, and US-NOAA. Global NEWS is a LOICZ affiliated project.

FIGURE 4.1 **Estimated net anthropogenic nitrogen inputs according to the world's main river catchments**



Source: Billen *et al.*, 2013.

The soil chemistry of phosphorous is complicated. Inorganic P is relatively immobile in the soil and adheres strongly to soil particles and organic material. Although soils often contain high levels of bound mineral P, low concentrations of plant-available P often necessitate fertilization to achieve optimum yields (Hart *et al.*, 1996). Phosphorus can be transported in runoff in the form of soluble P, often called dissolved reactive P (DRP), or attached to sediment and referred to as particulate P (PP).

Phosphorus is primarily obtained from mining finite deposits rich in phosphate. A total of 85 percent of mined phosphate is used for agriculture and only 10 percent for detergent manufacture, with the remainder used in other chemical processes and industry. A number of reports have drawn attention to the finite nature of rock phosphate reserves (see, for example, Keane, 2009; Vaccari, 2009). The majority of global supply currently comes from just a few key countries, posing a potential risk for future demand. Just three countries produce 66 percent of total rock phosphate. Many countries do not have the physical reserves or economic resources to obtain it (Sutton et al, 2013), yet in order to feed the growing world population, global phosphate fertilizer demand is expected to increase from 41 million tonnes in 2015 to 46 million tonnes in 2020. Table 4.3 below shows West Asia and South Asia with 4.4% each, Latin America and the Caribbean with 4.0% and sub-Saharan Africa with 3.6%, which have the highest expected compound annual growth rate. West Europe and North America have the lowest annual growth, accounting for -0.4% and 0.6% respectively (FAO, 2017b).

Table 4.3 | Global and regional phosphate fertilizer demand forecasts (thousand tonnes P_2O_5) and compound annual growth rate (CAGR), 2015 to 2020

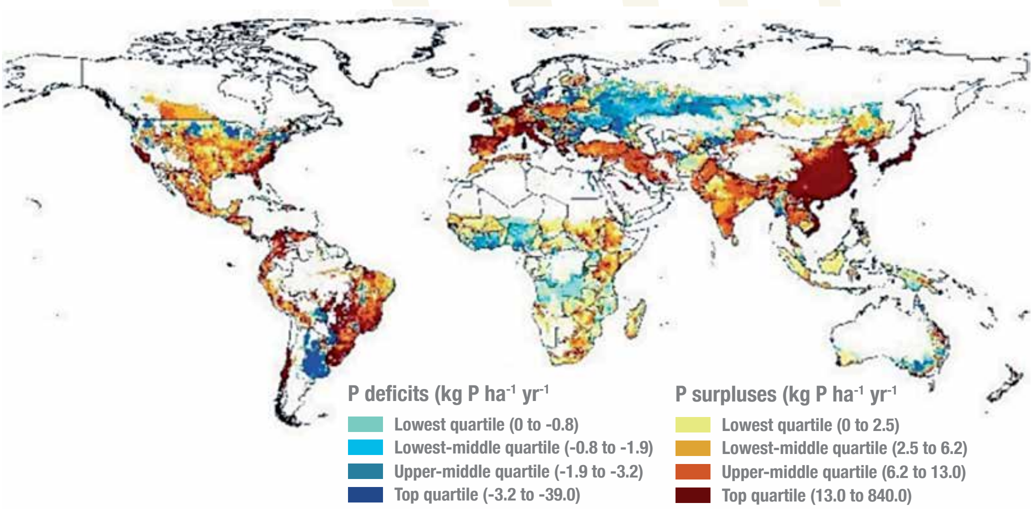
	2015	2016	2017	2018	2019	2020	CAGR (%)
WORLD	41 151	41 945	43 195	44 120	45 013	45 858	2.19
AFRICA	1 448	1 489	1 529	1 571	1 614	1 659	2.8
North Africa	*633*	*642*	*653*	*664*	*675*	*686*	*1.6*
Sub-Saharan Africa	*814*	*847*	*876*	*907*	*939*	*973*	*3.6*
AMERICAS	11 454	11 690	12 060	12 380	12 700	13 009	2.6
North America	*5 035*	*5 070*	*5 085*	*5 123*	*5 160*	*5 187*	*0.6*
Latin America & Caribbean	*6 420*	*6 620*	*6 975*	*7 257*	*7 539*	*7 822*	*4.0*
ASIA	22 918	23 312	24 056	24 544	25 005	25 432	2.1
West Asia	*351*	*367*	*383*	*400*	*417*	*436*	*4.4*
South Asia	*8 165*	*8 435*	*9 025*	*9 383*	*9 760*	*10 107*	*4.4*
East Asia	*14 401*	*14 510*	*14 648*	*14 761*	*14 827*	*14 889*	*0.7*
EUROPE	4 026	4 135	4 217	4 269	4 319	4 368	1.6
Central Europe	*756*	*780*	*807*	*835*	*864*	*889*	*3.3*
West Europe	*1 855*	*1 863*	*1 878*	*1 861*	*1 839*	*1 818*	*-0.4*
East Europe & Central Asia	*1 415*	*1 492*	*1 532*	*1 573*	*1 616*	*1 661*	*3.3*
Oceania	1 305	1 319	1 332	1 356	1 376	1 390	1.3

Source: FAO, 2017a.

The difference between P application and P withdrawal (harvested in plant matter or bound to the soil) is the P surplus. The greatest part is lost to the environment, into the atmosphere or into surface water or groundwater. A combination of data and modelling has been used to map phosphorus surplus using GIS techniques; an example is shown below. Figure 4.2 highlights the interregional variation in estimated phosphorus deficit and surplus in the world, where East Asia, Europe and parts of South America account for the highest surpluses of P. The agronomic P surpluses and deficits for the year 2000 were classified according to global quartiles (see MacDonald *et al.*, 2011 for further details). Such an exercise makes a number of assumptions about the processes, pathways and timing of nutrient application, and measures the potential pressure at source, rather than actual pressure at given points in the water system. The level of disaggregation is still somewhat coarse, but the exercise does provide a clear indication of where to prioritize efforts. Similar work at the catchment scale, mapped in greater detail and linked to more intensive modelling, can identify particular source areas in a catchment.

Phosphorus is of serious concern because it is often transported by sediment in rivers. Phosphate is not as soluble as nitrate and ammonia and tends to get adsorbed into soil particles and enter water bodies thorough soil erosion. In general, net fluxes of phosphate to surface water from soluble reactive phosphorous can be expected to be low, but where there is significant soil erosion (in surface irrigated conditions, or on soils with significant slopes that experience high rainfall), exports off-farm can be significant. The fluxes of phosphate

FIGURE 4.2 Estimated phosphorus deficit and surplus in the world



Source: MacDonald *et al.*, 2011.

in rivers are generally correlated to high stream flow with increased sediment transport, but are subject to considerable variation in both temperate and tropical settings. Short duration flood flows account for the majority of phosphate movement in river systems.

Phosphate is retained in river reaches through the deposition of sediment carrying adsorbed phosphate and due to uptake of phosphate by plant matter, particularly in stiller reaches and during periods of low flow (Demars *et al.*, 2005). Sediment affects water quality physically, chemically and biologically.

4.1.3 Nutrients from livestock systems

Animal manure is a primary source of nitrogen and phosphorus flow into surface and groundwater (US EPA, 2017). Most of the water used for livestock drinking and servicing returns to the environment in the form of liquid manure, slurry, greywater and wastewater.. When livestock is concentrated, the associated production of wastes tends to go beyond the buffering capacity of surrounding ecosystems, thereby polluting surface and groundwaters. Manure is generally collected for use as organic fertilizer, which, if applied in excess, will lead to diffuse water pollution. In many cases, if manure is not stored in contained areas, it can be washed into watercourses via surface runoff during significant rainfall. Feedlots are also often located near watercourses so that (nutrient-rich) animal waste (e.g. urine) can be released directly into the water.

Livestock-related nutrients in water resources contribute to water pollution and can accelerate plant and algae growth, algal blooms and the reduction of oxygen in water. Structural changes

BOX 4.2 | European Union efforts to reduce nutrient loss and water pollution

In recent years, phosphorus losses to water from point sources have decreased due to improved wastewater treatment (EEA, 2005). The 7th Environment Action Programme of the European Union has also confirmed that although nitrogen and phosphorus inputs to the environment in the region have decreased considerably over the past 20 years, excessive nutrient releases continue to affect air and water quality and to have a negative impact on ecosystems, causing significant problems for human health. More attention is now being focused on reducing nutrient loss from diffuse sources, a process that has been accelerated by the Water Framework Directive (WFD, Directive 2000/60/EC), which requires improvement of the quality of surface and groundwaters (EC, 2014). However, further efforts are needed to manage the nutrient cycle in a more cost-effective, sustainable and resource-efficient way and a more holistic approach is needed to address the nutrient cycle (EC, 2014).

taking place in much of the livestock sector, such as the increasing intensity of livestock production systems, higher stocking rates on European dairy farms, and the development of industrial livestock production, are increasing animal density in some areas, and this is linked to excessive volumes of waste and greater water contamination (Carpenter *et al.*, 1998). High levels of nutrient intake by livestock and the release of high concentrations of nutrients into aquatic ecosystems can lead to eutrophication and biological contamination of water resources (e.g. bacterial and viral pathogens) that cause human health problems.

The irrigation of feed crops is one of the largest agricultural sources of water pollution. From manured agricultural lands, N losses in runoff are usually under 5 percent of the applied rate in the case of fertilizer, and P losses to watercourses are typically estimated to be in the range of 3 to 20 percent of the P applied (see Table 4.4) (Carpenter *et al.*, 1998; Hooda *et al.*, 1998 cited in FAO, 2013). Overall N export from agricultural ecosystems to water as a percentage of fertilizer input ranges from 10 percent to 40 percent from loam and clay soils, to 25 to 80 percent for sandy soils (Carpenter *et al.*, 1998). Galloway and colleagues (2004) estimate that 25 percent of the N applied escapes to contaminate water resources (FAO, 2006). Nutrient surpluses resulting from the intensive use of feeds and high rates of fertilizer on pastures and fodder crops have also increased the emissions of nitrate and phosphate to groundwater and surface waters (Bouwman *et al.*, 2013). Feed and forage production induces a loss of N to aquatic sources of some 8 to 10 million tonnes per year, if one assumes such losses to be in line with N-fertilization shares of feed and forage production (some 20-25 percent of the world total) (FAO, 2006). Intensification has mostly occurred on sandy soils, which, although low in fertility, are easy to cultivate and respond well to inorganic fertilizers.

4.2 Nitrate and phosphate concentration in surface water and groundwater

Agriculture is the largest contributor of nitrogen pollution, and excess nitrogen and phosphates can leach into surface runoff into waterways.

Nitrate is a serious threat to many global aquatic ecosystems. It is the most common chemical contaminant in the world's aquifers (Spalding and Exner, 1993; Thorburn *et al.*, 2003; Jalali, 2005; Batlle Aguilar *et al.*, 2007), Standards for nitrate limits in groundwater vary considerably across the globe, although many are more stringent than the WHO guidelines (50mg/l). Nitrate in groundwater has been reported as a major problem in Europe, the United States and South and East Asia. In Europe, even when mean concentrations of nitrate in groundwater have remain relatively stable in the last few decades, nitrate drinking water limit values have been exceeded in around one-third of the groundwater bodies for which information is currently available (Mateo-Sagasta

Table 4.4 | Estimated N and P losses to freshwater ecosystems from manured agricultural lands

Region	N from animal manure		N losses to freshwater courses	P from animal manure		P losses to freshwater courses
	Crop	Pasture		Crop	Pasture	
	(thousand tonnes)					
North America						
Canada	207.0	207.0	104.0	115.3	20.0	16.2
United States	1 583.0	1 583.0	792.0	881.7	84.0	115.9
Central America	351.0	351.0	176.0	192.4	22.0	25.7
South America	1 052.0	1 051.0	526.0	576.8	59.0	76.3
North Africa	36.0	34.0	18.0	18.5	10.0	3.4
West Asia	180.0	137.0	79.0	92.3	48.0	16.8
Western Africa	140.0	148.0	72.0	71.9	26.0	11.7
Eastern Africa	148.0	78.0	57.0	76.0	24.0	12.0
Southern Africa	79.0	3 085.0	791.0	40.6	50.0	10.9
OECD Europe	3 048.0	737.0	1 036.0	1 896.7	18.0	229.8
Eastern Europe	757.0	2 389.0	787.0	413.4	177.0	70.8
Former Soviet Union	2 392.0	167.0	640.0	1 306.2	13.0	158.3
South Asia	3 816.0	425.0	1 060.0	1 920.9	10.0	231.7
East Asia	5 150.0	1 404.0	1 639.0	3 358.3	29.0	406.5
Southeast Asia	941.0	477.0	355.0	512.0	15.0	63.2
Oceania	63.0	52.0	29.0	38.9	20.0	7.1
Japan	361.0	59.0	105.0	223.0	0.0	26.8
World	60 644.0	12 384.0	8 262.0	11 734.7	625.0	1 483.2

Source: FAO, 2006.

and Burke, 2010). Additionally, in India, hundreds of districts in 21 Indian states have reported an occurrence of nitrate in groundwater that is well beyond the national permissible limit (45 mg nitrate/l) (Central Ground Water Board, 2010).

Nitrate levels in groundwater remain above prescribed limits in many OECD countries, on average in roughly 10-15 percent of these cases. Due to the EU Nitrate Directive and national measures, nitrogen pollution from agriculture has been reduced in some areas over the last ten to fifteen years (see Table 4.5) (EEA, 2015). Nitrate Vulnerable Zones, areas designated as being at risk from agricultural nitrate pollution, must be specified to comply with the Nitrate Directive. However, the methodologies for defining at-risk zones vary from country to country and are not governed by regulations in some cases. They are commonly based on an assessment of whether the risk of N leaching to groundwater is high in order to identify areas where reduced fertilizer applications are necessary. Such assessments require modelling or simple assumptions about the partitioning of nutrient balances to determine the proportion of applied nitrogen that is transported to the aquifer.

At the European level, there was a slight increase in average annual mean nitrate concentration in European groundwater from 1992 to 1998. Since 2005, concentrations have declined again and, in 2011, the mean concentration had almost returned to the 1992 level. River nitrate concentrations also declined steadily over the period from 1992 to 2012, when average nitrate concentration in European rivers declined by 0.03 milligrams per liter of nitrogen (mg N/l) (0.8 percent) per year. For example, water body monitoring in Bulgaria, over the periods 2004-2007 and 2008-2011 shows a generally slight and stable improvement of water quality and reduced nitrate concentration. However, this does not mean the problem has been resolved. The status of water bodies in vulnerable areas, mainly in the Danube and East Aegean basins, is dire. The most recent data from 2013 show higher nitrate concentration in the surface waters than in 2012 levels. Similarly, groundwater concentration

Table 4.5 | Average annual mean Nitrate and Phosphate concentration in freshwater in the European Union (1992-2012)

Year	1992	1997	2002	2007	2012
GW nitrate (mg NO_3/l)	17.4	18.8	17.7	18.8	17.6
Rivers nitrate (mg NO_3/l)	2.66	2.49	2.25	2.30	2.10
Rivers phosphate (mg P/l)	0.133	0.120	0.092	0.083	0.055
Lakes phosphorus (mg P/l)	0.039	0.033	0.033	0.026	0.027

Source: EEA, 2015.

remains steadily above 50 mg/l. In some places in the Danube Basin, concentrations are up to 140 mg/l, while in unconfined aquifers in the East Aegean Basin, they reach as high as 230 mg/l. Nitrate concentrations in the lower layers of groundwater are even worse, reaching up to 120 mg/l. These aquifers are the only sources of drinking water and, due to the hazardous situation, emergency measures need to be taken to improve the situation (EC, 2014).

In other hand, during the past few decades, there has been a gradual reduction in phosphorus concentrations in many European lakes. Average lake phosphorus concentration decreased over the period from 1992 to 2012 by 0.0004 mg P/l, or 0.8 percent per year. Phosphorus pollution from point sources is gradually becoming less significant. The treatment of urban wastewater has improved, phosphorus in detergents has been reduced and many wastewater outlets have been diverted away from lakes. However, diffuse runoff from agricultural land continues to be an important source of phosphorus in many European lakes. Moreover, phosphorus stored in sediment can keep lake concentrations high and prevent improvement of water quality despite a reduction in inputs (EEA, 2015).

The average orthophosphate[5] concentration in European rivers decreased markedly over the period from 1992 to 2002 (by 0.003 milligrams per liter of phosphorous [mg P/l], or 2.1 percent per year). In many rivers, this reduction started in the 1980s, but the marked decline is also evident for the time period from 2000 to 2012. Average concentrations are somewhat higher where more river stations are included. The decrease in river orthophosphate can be linked to measures introduced by national and European legislation, in particular the Urban Waste Water Treatment Directive, which calls for the removal of nutrients. Moreover, the switch to phosphate-free detergents has contributed to lower phosphorus concentrations.

4.2.1 Concentration of nutrients resulting in eutrophic lakes, reservoirs and coastal waters

The global distribution of reactive nitrogen is far from uniform, and N pollution in coastal waters is greatest where agricultural activity and urbanization are the highest (Howarth, 2006). The 1970s saw an explosive increase in coastal eutrophication in many parts of the world, which correlates with the increased production of reactive N for agriculture and industry during that period. In some regions, such as the North Sea and the Yellow Sea, human activity probably has increased N fluxes to the coast by 10- to 15- times or more (Howarth, 2006). On average, human activity has likely increased N fluxes to the coast of the USA six-fold.

5 Phosphates are very important in fertilizer production. Orthophosphates are normal phosphates that are composed of one phosphate unit per molecule. The main difference between phosphate and orthophosphate is that phosphate is any compound composed of phosphate units whereas orthophosphate is composed of one phosphate unit.

Coastal zones and estuaries receive nutrient loads from the open ocean as well as from upstream sources, whereas inland lakes only receive nutrient loads from upstream. Despite the knowledge that N is the key factor in the development of hypoxia, there is considerable variation in the susceptibility of coastal zone across a range of N loadings. Oceanic N:P ratios are well below the threshold value for eutrophication (the Redfield Ratio), as denitrification occurs along the continental shelf. Nitrogen, while clearly very significant, is not the only element of concern for coastal systems, even for those in the temperate zone. Phosphorus is probably limiting in some estuaries, (Howarth 1998), for example, the Apalachicola estuary on the Gulf Coast of Florida and in several estuaries on the coast of the Netherlands in the North Sea. Seasonal switching of nutrient limitation has also been observed in the Chesapeake Bay and the hypoxic zone of the Gulf of Mexico.

Estuaries generally have a lower N:P ratio than lakes, therefore nitrogen can often be the limiting factor for eutrophication. Furthermore, conditions for eutrophication are affected by the chemical makeup of sediments (e.g. acidity, carbon contents changes and C:P:N ratios related to sulphate), levels of salinity and water conditions that control the zooplankton-grazing phytoplankton and, therefore affect the equilibrium of trophic webs (Howarth, 1998). At the same time, the reservoir of accumulated P in estuarine sediments is large, tending to increase P concentrations in seawater as compared to those in lakes. The factors controlling increased desorption of P in estuaries are perhaps not yet fully understood. It is thought that the higher sulphate concentrations in estuarine sediments reduce storage and accelerate desorption by sequestering more iron as iron sulphide. However, there is also variation in the response of estuaries to N loading, with some estuaries being far more sensitive to eutrophication than others (NRC, 1993).

The global anthropogenic P load to freshwater systems from both diffuse and point sources is estimated at 1.5 Tg/yr. Asia accounts for more than half of this total load, followed by Europe (19%) and Latin America and the Caribbean (13%). Overall, the domestic sector contributes 54 percent of the total, agriculture 38 percent and industry 8 percent. In agriculture, cereals production makes the largest contribution to the P load (31 percent) followed by fruits, vegetables and oil crops, each of which contributes 15 percent (Mekonnen and Hoekstra, 2017).

4.3 Impacts on health and environment

4.3.1 Human health impacts

Nutrient pollution and harmful algal blooms create toxins and compounds that are dangerous to human health. There are several ways that people, livestock (and pets) can

be exposed to these compounds, including contact with polluted water or consumption of contaminated water or foods (US EPA, 2017). Nitrate poses significant health hazards. It is highly soluble in water and can seep into groundwater from septic tanks, animal waste, fertilizers (manufactured and compost) and sewage sludge. Stormwater runoff also carries nutrients directly into rivers, lakes and reservoirs, which provide drinking water for many people. When disinfectants used to treat drinking water react with toxic algae, harmful chemicals called dioxins can be created. These byproducts have been linked to reproductive and developmental health risks and even cancer (US EPA, 2017).

Nitrate pollution in drinking water is a serious health concern in many developing countries. Nitrate poses a serious threat to the health of infants under six months of age, pregnant women and people with low stomach acid (Hypochlorhydria). The World Health Organization (WHO) thus recommends limiting nitrate-nitrogen in drinking water to 10 mg/l. Infants under six months of age who drink water too high in nitrates can develop methemoglobinemia, the so-called 'blue-baby' syndrome. Infants have bacteria in their stomach that converts nitrate to nitrite. The nitrite enters the baby's bloodstream and reacts with hemoglobin to form methemoglobin, which interferes with the blood's ability to carry oxygen (Wedin and Sorensen, 2013). Infants may show signs of suffocation and blue-tinted skin and can become seriously ill and even die. Although there are no reliable estimates on the current levels of methaemoglobinaemia, according to WHO (2017) the most common cause is a high level of nitrates in drinking water from the use of manures and fertilizers on land.

Direct exposure to toxic algae is another major problem arising from agricultural nutrients and water pollution. Elevated levels of phosphorous can promote the unwanted growth of algae in freshwater, leading to reduced water quality and low levels of oxygen in the water, which can cause fish kills and pose a risk to human health. Species of algae that are common in algal blooms produce neurotoxins (which affect the nervous system) and hepatoxins (which affect the liver). Most algae are generally harmless, but some produce hazardous toxins, which are extremely dangerous when touched or consumed. Drinking, accidentally swallowing or swimming in contaminated water affected by a harmful algal bloom can cause serious health problems including rashes, stomach or liver illness, respiratory problems and neurological affects (US EPA, 2017).

4.3.2 Environmental impacts: eutrophication and hypoxic water

The livestock sector is probably the largest sectoral source of water pollution, contributing to eutrophication, 'dead zones' in coastal areas, degradation of coral reefs, human health problems, emergence of antibiotic resistance and many others. Generally, as the load (or

concentration) of N and P in a lake or river increases, the probability of algae growth also increases. A sudden ‘population explosion’ of naturally-occurring microscopic algae is known as an algal bloom. Algal blooms can be caused by many factors, such as seasonal changes in temperature, abundance of sunlight and/or high nutrient concentration in the water. When the algal species produce toxic organic compounds, they can be harmful or even deadly for humans and biodiversity. After being consumed by small fish and shellfish, these toxins move up the food chain and harm larger animals like sea lions, turtles, dolphins, birds, manatees and fish (US EPA, 2017).

Not all algal blooms produce toxic compounds. But even when they are not toxic, algal biomass and the organic matter they produce can accumulate in dense concentrations near or below the water surface, which can lead to an explosive increase of bacteria present in the water through the degradation of this organic material. Algal blooms can hurt aquatic life by blocking out sunlight, clogging fish gills and causing a sudden drop in dissolved oxygen concentration in the water. This can reduce the ability of fish and other aquatic life to find food and can cause entire populations to leave an area or even die out (Villacorte *et al.*, 2015).

When surface waters become enriched with plant nutrients, eutrophication can also result. The use of fertilizers is associated with eutrophication, which is generally the result of complex interactions between temperature, nutrient loading, flow rate and other biological and geochemical factors. The OECD (2012) defines eutrophication as “the increase in the rate of production and accumulation of organic carbon in excess of what an ecosystem is normally capable of processing”. Similarly, eutrophication is defined by the European Commission as the "accelerated growth of algae and higher forms of vegetation caused by the enrichment of water by nutrients, particularly compounds of nitrogen and/or phosphorus, inducing an undesirable disturbance of the ecological balance in the reservoirs (EC, 2014).”

Eutrophication symptoms may include the following:

- excessive phytoplankton and macroalgal growth at the water surface, which may reduce light penetration and cause the decline of submerged aquatic vegetation;
- an imbalance in nutrient ratios that can lead to a shift in the composition of phytoplankton species, creating favourable conditions for toxic algal blooms;
- changes in the composition of benthic species, leading to reduced diversity and negative impacts on the food web;

- reduction of dissolved oxygen and formation of hypoxic waters (dead zones) in coastal and marine settings.

Despite some data gaps, 415 coastal areas have been identified worldwide as experiencing some form of eutrophication, of which 169 are hypoxic (see the section below), 233 are areas of concern and 13 are systems in recovery (WRI, 2008).

Hypoxic waters (dead zones)

Anthropogenic eutrophication (nutrient over-enrichment) is the main driver behind the expansion, intensity and duration of coastal hypoxic conditions (Rabalais *et al.*, 2009). Hypoxic areas are 'dead zones' where there is insufficient oxygen to support normal marine flora and fauna. The threshold oxygen concentration for hypoxia is not well defined, but a value of 2 mg/l is generally used, based on the observed behavior of a range of marine organisms. This is equivalent to 1.4 ml/L, 63 μmol/L or 30 percent of oxygen saturation (Rabalais *et al.*, 2009). Since it is hard to detect oxygen accurately at such low levels, the exact oxygen threshold for hypoxia is difficult to identify; however, an effective indicator of hypoxia is nitrite content, which is an intermediate of denitrification.

Hypoxia affects an area of 240 000 km^2 globally, comprising 70 000 km^2 of inland waters and 170 000 km^2 of coastal areas. Municipal wastewater is often the main driver, although agricultural nitrogen is also a major factor in some areas, such as the Bay of Mexico. The spatial scales of hypoxic systems range from inshore estuaries to coastal shelves and open ocean areas and span depths of 1-2 m up to 600-700 m. Large areas in the open sea naturally have low oxygen content. These are known as Oxygen Minimum Zones (OMZs) and are the largest hypoxic areas in the world, covering 30 000,000 km^2, which is roughly 8 percent of the total ocean surface area. Methane often builds up in anoxic conditions in seawater and, more notably, in freshwater. Recent studies demonstrate that fluxes of methane (CH_4), a potent greenhouse gas, to the atmosphere from the expanding coastal hypoxic zones are probably insignificant, but coastal upwelling areas with shallow OMZ generate significant quantities of nitrous oxide (N_2O).

Coastal hypoxia kills or impairs fish and other marine life populations, and reduces fisheries catches. Pelagic fishes that are vulnerable to hypoxia and large populations of hypoxia- tolerant gobies now dominate the trophic structure of upwelling areas. Larger mobile predator species are the first to be affected by hypoxic areas. For example, the habitats of Atlantic blue and white marlin and sailfish are reduced and appear to have declined with the shoaling of the OMZ in the Pacific Ocean. They tend to be replaced by

non-pelagic species, such as gelatinous plankton and squid, as observed in the Benguela and California upwelling regions, but exact changes in community composition are difficult to estimate.

However, it is the recent emergence of coastal hypoxia that has put the spotlight on the consequences of intensified agricultural production systems. Until recently, hypoxic areas were found mainly on the coasts and in estuaries of developed countries, but the largest future increases in the number of hypoxic systems are expected in southern and eastern Asia. The best-known examples of hypoxic zones are found in the Baltic Sea, Gulf of Mexico, Black Sea, Mediterranean, Benguela, West Indian Ocean, Sea of Japan, Yellow Sea and South China Sea. The Baltic Sea is the largest hypoxic zone in the world, followed by the Gulf of Mexico. Over the past three decades, oxygen concentrations in both have been declining faster within 30 km of the coast with between 0 and 300 m water depth than in the open ocean.

BOX 4.3 Nutrient impact in the Baltic Sea

The Baltic Sea is the largest single marine area in the world where hypoxia and anoxia are the result of human activity. Hypoxia has occurred intermittently and naturally over the past 8 000 years of its existence, since it is a ‘close’” sea with limited water exchange with the North Sea.

The annual total nitrogen (N) input into the Baltic Sea is estimated to be around one million tons complemented by around 50 000 tonnes of phosphorus (P). The main sources of N within the Baltic Sea catchment are agriculture, municipalities, industry, power plants and traffic (HELCOM, 2002). Although nutrients are mainly carried into the Baltic Sea by rivers, about one quarter the N load is estimated to be airborne, resulting from the burning of fossil fuels. The contribution via groundwater and direct discharges cannot be neglected, but is not well quantified. Denitrification removes about 470 000 tonnes of N per year, and a further 130 000 tonnes/year are fixed in biomass and substrate, resulting in a net export of 150 000 tonnes of N and zero net flux of P to the North Sea.

The surrounding land mass naturally exports high nutrient loads; these have been magnified by agricultural development, which saw a four-fold increase in nitrogen and an eight-fold increase in phosphorous loads during the 20th century. Fish catches rose from a stable level of 0.5 m tonnes per year to 1 m tonnes in 1984, in parallel with increasing nutrient content, then subsequently declined to 0.6m tonnes in part due to overfishing of cod. Baltic cod is particularly sensitive to low oxygen concentrations at early stages of growth and hypoxia has resulted in habitat loss over vast areas, the eradication of benthic fauna, and the severe disruption of the food web.

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CHAPTER 5. PESTICIDES

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With contributions from Cecilia Spottorno, Hugh Turral, Baogen Gu and Carolyn Opio

A pesticide is any substance or mixture of substances intended to repel, destroy or control any pest or prevent plant growth Pesticides may include chemical and biological ingredients and may be further characterized as insecticides, herbicides, fungicides, bactericides, rodenticides, and plant growth regulators, as well as public health insecticides. The use of pesticides can protect crops and prevent post-harvest losses, thus contributing to food security. Pesticides also help to avoid the spread of pests and diseases in global trade and in stored agricultural commodities.

The development of pesticides was fundamental to the Green Revolution and transformation of modern agriculture and in many countries the use of pesticides, especially in monoculture areas, has been common practice for pest control (Ongley, 1996). Despite their importance in plant protection, more recently evidence of the serious impacts on the environment has emerged. Pesticide misuse and pesticides as water pollutants are increasingly serious global challenges resulting in heavy environmental pollution and high health risks for humans (FAO/WHO, 2016). This chapter briefly describes the global usage of pesticides, their characteristics, environmental loads and resulting concentration in water bodies and their impact on human health and the environment.

5.1 Pesticides and water pollution

5.1.1 Pesticide consumption and production

Fruit and vegetable crops account for most of the pesticides used worldwide, although in developed countries, pesticides (mainly herbicides) are used primarily for maize production (Zhang, Jiang and Ou, 2011).

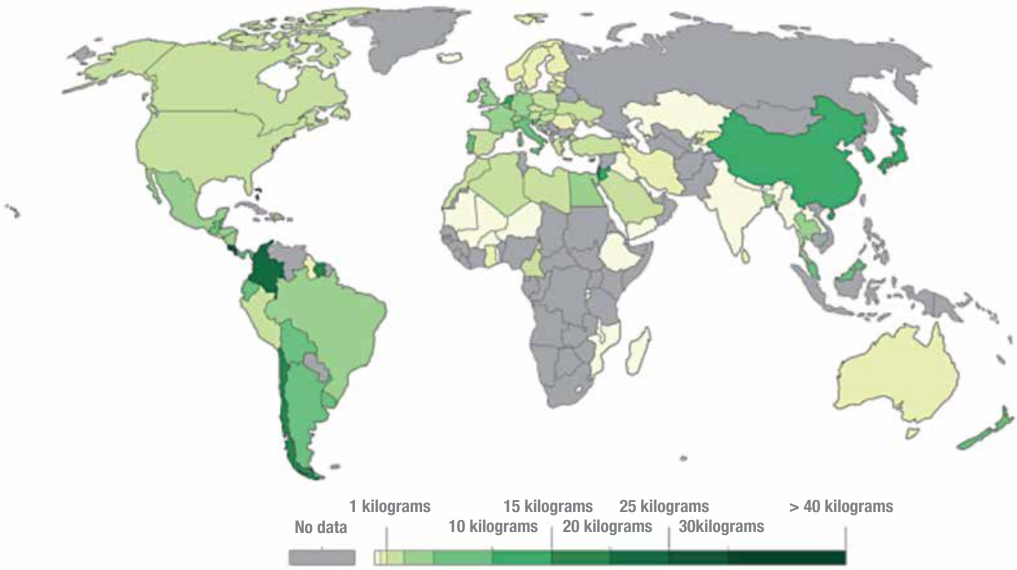
Pesticide production is a global industry worth more than USD 35 billion per year. About 500 pesticides are used for mass application, some of which are highly poisonous to the environment (Zhang, Jiang and Ou, 2011). Globally, 4.6 million tonnes of chemical pesticides are sprayed into the environment every year. Worldwide consumption of pesticides has undergone significant changes since the 1960s. The proportion of herbicides has increased and that of insecticides and fungicides and bactericides has declined (Zhang, Jiang and Ou, 2011). In terms of current global consumption, 47.5 percent of pesticides are herbicides, 29.5 percent are insecticides and 17.5 percent are fungicides, with all others accounting for 5.5 percent (De, 20.14).

Overall pesticide consumption in Europe has declined 50 percent compared to the average in the 1980s (Zhang, Jiang and Ou, 2011). As early as 1972, the use of DDT and related organochlorine insecticides, was banned in the United States and many other countries. Subsequently, pesticide consumption in the United States declined by 35 percent without reducing crop production (Zhang, Jiang and Ou, 2011). However, the United States is still the second largest consumer of pesticides in the world, accounting for 410 000 tonnes of active ingredients, followed by Brazil with 396 000 tonnes, Argentina with 208 000 tonnes, Mexico with 99 000 tonnes, Ukraine with 78 000 tonnes, Canada with 73 000 tonnes and France, Italy, Spain and India with around 60 000 tonnes each (FAO, 2017a). China is the world's biggest producer and exporter of pesticides, and annual pesticide use in China is about 1.8 million tonnes of active ingredients (FAO, 2017a), on approximately 300 million hectares of farmland and forests. However, official statistics indicate about seven percent of China's cropland has been polluted due to improper use of pesticides and fertilizers, and the use of high-toxicity pesticides has killed beneficial insects, leading to numerous pest disasters in China in recent years.

Figure 5.1 below shows the average pesticide application per unit of cropland.

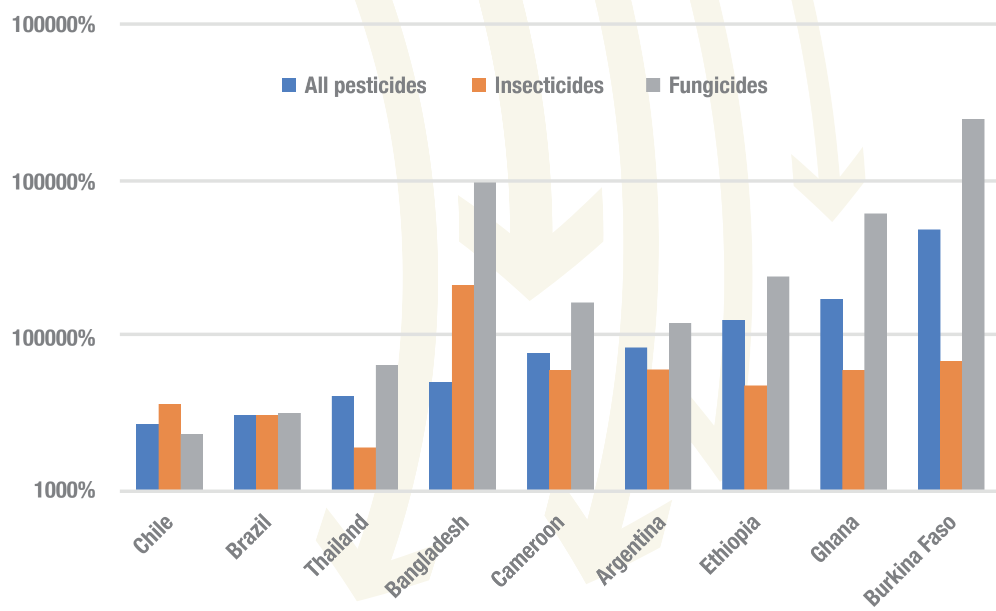
In terms of pesticide consumption in the developing world, sales are growing, even if from generally low levels (Figure 5.2). In Vietnam, pesticide consumption increased from 14 000 tonnes (under 837 trade names) in 1990 to 50 000 tonnes (under more than 3 000 trade names)

FIGURE 5.1 Pesticide use per hectare of cropland, 2007



Source: FAO, 2014.

FIGURE 5.2 Change in agricultural pesticide use (%) over an approximate 20-year period (1990 to latest data: 2007–2012) (countries with the highest increase)



Source: Adapted from Pretty and Bharucha, 2015; FAOStat, 2011; OECD, 2013.

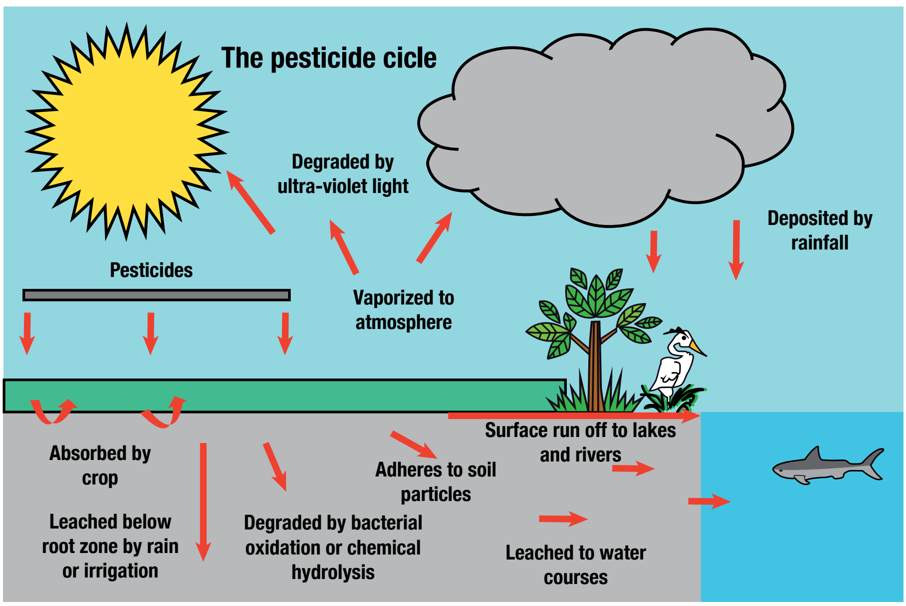
in 2008. Moreover, in the past decade pesticide sales increased even in Africa, which has the lowest consumption of any region. Several upper middle-income countries (e.g. Argentina, Brazil, Malaysia, South Africa and Uruguay) and lower middle-income countries (e.g. Cameroon, Cape Verde, Nicaragua, Pakistan and Ukraine) have experienced double-digit growth in the intensity of pesticide use, albeit from relatively low levels. Costa Rica, Colombia, Japan and Mexico have the highest intensity of pesticide use (kilogram of active ingredient per hectare and per crop output) worldwide (Schreinemachers and Tipraqsa, 2012).

5.1.2 Pesticide pollution of water

In terms of the risk of pesticide water pollution, each pesticide has unique properties and many variable factors affect this risk, such as the active ingredients, contaminants and additives as well as any degradate formed during chemical, microbial or photochemical degradation of active ingredients. The increased use of pesticides has been accompanied by the growing presence in soil of a large number of transformation products (TPs) for a wide variety of pesticides (Aktar *et al*., 2009) These compounds have proven to be highly toxic, persistent and accumulative in the food chain.

Figure 5. 3 below shows the pesticide cycle and how pesticides typically move throughout an ecosystem and may end up in other parts of the environment, such as in water and soil.

FIGURE 5.3 | **The pesticide cycle**



Source: University of Reading, 2018.

The persistence of agricultural toxins in ecosystems presents two primary concerns:

- bioaccumulation: an increase in concentration of a pollutant from the environment to the first organism in a food chain, and
- biomagnification: an increase in concentration of a pollutant from one link in a food chain to another. In order for biomagnification to occur, the pollutant must be long-lived, mobile, soluble in fats and biologically active.

Pesticides and TPs can be grouped into two classes: hydrophobic and polar. Hydrophobic pesticides are persistent, bioaccumulable and bind strongly to soil. They include organochlorines such as DDT, endosulfan, endrin, heptachlor, lindane and their TPs. Most of these have already been banned in agriculture but their residues are still present. Polar pesticides are mostly herbicides, but they also include carbamates, fungicides and some organophosphorus insecticide TPs. They are soluble, which means that runoff and leaching can transported them to surface water and groundwater from soil. The most researched pesticide TPs in soil are derived from herbicides and although most herbicides are not toxic to soil fauna, some are, including triazines (such as atrazine) and bipyridyl herbicides (such as paraquat). Newly developed pesticides, such as carbamate and organophosphate insecticides are considered 'safer' in that they are not persistent, one of the requirements to avoid biomagnification.

Other important properties relevant to pollution include the following:

- Pesticide half-life: The amount of time a pesticide takes to break down is measured in terms of its half-life. The more stable a pesticide, the longer it takes to break down and the higher its persistence. A pesticide's half-life also depends on specific environmental and application factors, but it is unique to each individual product.
- Mobility in soil: All pesticides have unique mobility properties through the soil structure, both vertically and horizontally. Even when pesticides have short half-lives, there are considerable risks from direct contamination of waterways through spray drift, runoff and leakage and seepage from improper storage.
- Solubility in water: Many pesticides are soluble in water so that they can be applied with water and absorbed by the target. The risk of leaching increases with higher solubility. Residual herbicides generally have lower solubility to aid soil binding but their persistence in soil can cause other problems (SDWF, 2017).

External factors are also important in determining the behavior of pesticides in the environment, including:

- Microbial activity, where pesticides in soil are primarily broken down by microbial activity and the greater the microbial activity, the faster the degradation.
- Soil temperature, where soil microbial activity and pesticide breakdown are closely linked to soil temperature.
- Treatment surface, where pesticides, such as residual herbicides, applied to hard surfaces (such as concrete or tarmac in garden pathways and driveways) cannot be absorbed and are particularly vulnerable to movement into water courses and non-target areas, especially after rainfall.
- Application rates, where the length of time that significant concentrations remain in the environment is directly related to the amount of pesticide that is applied (SDWF, 2017).

5.2 Pesticide accumulation in groundwater, surface water, lakes and reservoirs

Pesticides contaminate surface water, groundwater and soil. They can reach surface water by runoff from treated plants and soil and contamination of water by pesticides is widespread. The presence of pesticides as pollutants of water depends on their mobility, solubility and rate of degradation. Many modern pesticides break down quickly in soil or sunlight but are more likely to persist if they reach subsoil or groundwater because of reduced microbial activity, absence of light and lower temperatures (Kerle *et al*., 1994).

Pesticide residues are increasingly present in surface and groundwater (in OECD countries), with a significant number of samples above the legislated limits. In a survey of 3 500 sites in England and Wales, 100 of the 120 pesticides targeted were detected and five herbicides (Atrazine, diuron, bentazone, isoproturon and mecoprop) regularly exceeded EC Drinking Water Directive limits (Packman, 1995). In China, pesticides such as chlordane, Atrazine, carbofuran and many older pesticides, such as DDT, are banned outright but are used illegally and persist in the environment. Pesticides such as DDT, HCH, dieldrin and endrin have been detected in most bodies of water in China. Water bodies near croplands are generally polluted and the pesticide concentration in such water can reach tens of milligrams per litre. The levels of pesticide water pollution can be ranked from highest to lowest concentration as: cropland water > field ditch water > runoff > pond water > groundwater > river water > deep groundwater > sea water (Zhang, Jiang and Ou, 2011).

Most developed countries regularly monitor key pesticides, although high costs for sampling and analysis mean many datasets are not extensive. However, according to a recent meta-analysis, monitoring data is lacking for approximately 90 percent of global cropland (Stehle and Schulz, 2015). This study included 838 peer-reviewed scientific articles on surface water exposure to agricultural insecticides that between them covered more than 2 500 sites in 73 countries. The authors looked at whether measured insecticide concentrations (MICs) exceed regulatory threshold levels, as well as at the relationship between historical insecticide development, the level of environmental regulation and risks of exposure of surface waters to agricultural insecticides. The results showed that no MICs were present in 97.4 percent of the analyses and that newer insecticides, such as pyrethoids, have higher retention time locking (RTL) for surface water or sediment.

Another study, by the United States Geological Service (USGS), reported more than 143 different pesticides and 21 transformation products in groundwater, including pesticides from every major chemical class (Toccalino *et al.*, 2014). Over the past two decades, pesticides and TPs have been detected in the groundwater of more than 43 states. The United States of America Environmental Protection Agency (US EPA, 2017) reports widespread contamination of waterways (rivers, lakes) by Atrazine, the second most commonly used herbicide in the United States. In 1984, 12 kinds of high-concentration pesticides were measured in the groundwater in 18 states of the United States. By 1986, 17 kinds of pesticides had been detected in groundwater in 23 states. In Florida, concentration of dibromoethane in groundwater was 64 times higher than the maximum allowable amount, which resulted in more than 1 000 wells being closed (FAO, 2013). Moreover, the US EPA reports that many rural wells in the country contain at least one of 127 pesticides (Zhang, Jiang and Ou, 2011). Table 5.1 below shows the dominant pesticides used and typical compounds detected in groundwater selected regions.

Research from the USGS National Water-Quality Assessment (NAWQA) programme provides a comprehensive analysis of occurrence and decadal-scale changes in pesticide concentrations in groundwater of the United States for the period 1993-2011. Figure 5.4 below shows the twenty most frequently detected pesticides and their degradation products, as well as how the occurrences of individual pesticides in streams have changed between decades in the United States. The pesticides most frequently detected in the decadal comparisons include 11 herbicides (plus two degradates – deethylatrazine from atrazine and three 4-dichloroaniline from triclocarban, diuron, linuron, and other herbicides), four insecticides (plus two degradates – desulfinylfipronil and fipronil sulfide, both from fipronil), and one fungicide—metalaxyl (Stone, Gilliom and Ryberg, 2014). The results-based

Table 5.1 I Dominant pesticides used and typical compounds detected

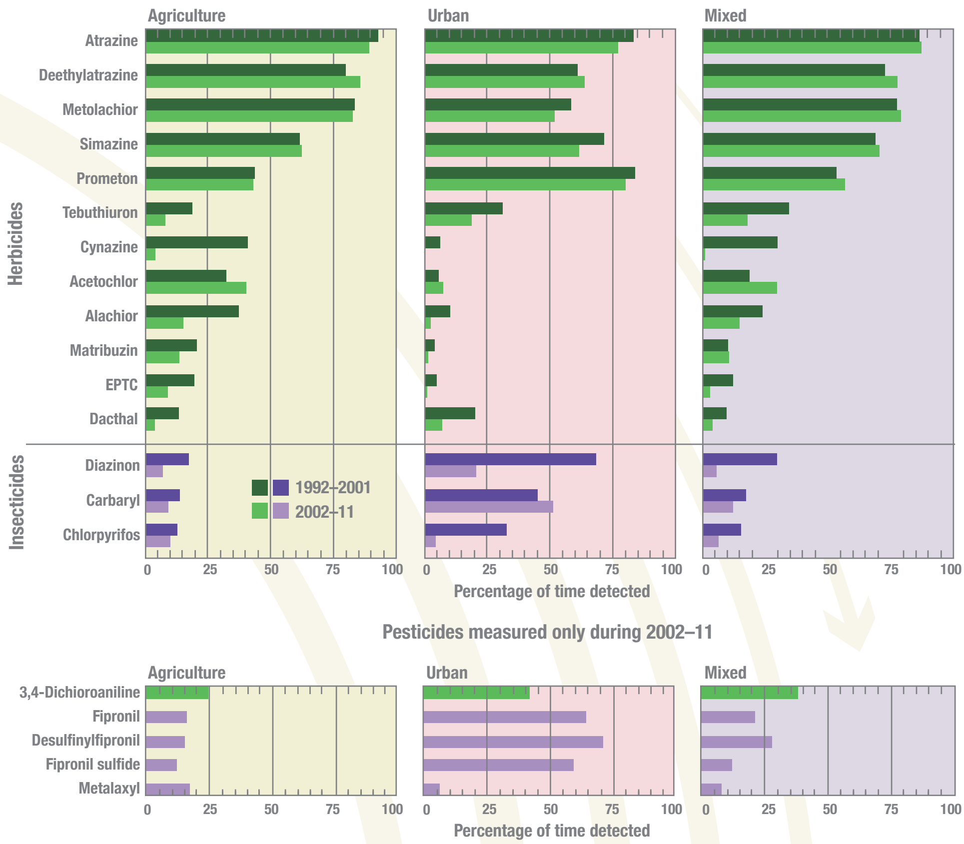
Region	Dominant pesticide use	Typical compounds detected
United Kingdom	Pre-and post-emergent herbicides on cereals, triazine herbicides on maize and in orchards	Isoproturon, mecoprop, atrazine, simazine
Northern Europe	Cereal herbicides and triazines as above	As above
Southern Europe	Carbamate and chloropropane soil insecticides for soft fruit, triazines for maize	Atrazine, alachlor
Northern United States	Triazines on maize and carbamates on vegetables e.g. potatoes	Atrazine, aldicarb, metolachlor, alachlor and their metabolites
Southern & Western United States	On citrus and horticulture, and fumigants for fruit and crop storage	Aldicarb, alachlor and their metabolites, ethylene dibromide
Central America & Caribbean	Fungicides for bananas, triazines for sugarcane, insecticides for cotton, and other plantation crops	Atrazine
South Asia	Organo-phosphorus & organo-chlorine insecticides in wide range of crops	Carbofuran, aldicarb, lindane
Africa	Insect control in houses and for disease vectors	Little monitoring as yet

Source: Morris *et al.*, 2003, as cited in FAO, 2012.

monitoring and assessment confirm previously reported findings: pesticides were frequently detected in groundwater (53 percent of all samples) and although concentrations seldom exceeded human-health benchmarks (1.8 percent of all samples) pesticides were found at concentrations that exceeded aquatic-life benchmarks in many rivers and streams that drain agricultural, urban, and mixed-land use watersheds. The frequent detection (>36 percent) of pesticides in samples from major aquifers used to supply drinking-water indicates the vulnerability of these aquifers to contamination from human activities at the land surface and emphasizes the importance of wellhead protection programmes and other strategies designed to reduce groundwater contamination from human sources (Toccalino *et al.*, 2014). In agricultural land, the proportion of assessed streams with one or more pesticides that exceeded an aquatic life benchmark, were very similar for the previous two decades, with 69 percent for 1992–2001 compared to 61 percent for 2002–2011 (Stone, Gilliom and Ryberg, 2014).

The result outlined in this section highlight the high risk to global water resources and the need for improved global regulation of agricultural pesticide practices. In addition, further research efforts are needed to better understand the presence and effects of

FIGURE 5.4 Percentage of time detected for the periods of 1992-2001 and 2002-2011, for the most frequently detected pesticides in United States streams and rivers



Source: Stone, Gilliom and Ryberg, 2014.

pesticide use, especially in developing countries, which often lack monitoring data and where chemical analyses are often not carried out because of a lack of facilities, low quality reagents or financial constraints (SDWF, 2017).

5. 3 Impacts on health and environment

Despite recent estimates that the economic impact of pesticides on non-target species (including humans) is approximately USD 8 billion annually in developing countries, the use of pesticides is increasing and millions of tonnes of active pesticide ingredients are used in agriculture (Aktar *et al.*, 2009). Understanding the potential effects of the resulting chemical mixtures on humans and the environment is one of the most complex problems facing scientists and regulatory agencies (SDWF, 2017). Pesticide accumulation in groundwater and surface water bodies, especially lakes and wetlands, is thus a growing concern.

Pesticide residues reach the aquatic environment through several non-point and point sources of pollution, including direct run-off, leaching, careless disposal of empty containers or the washing of equipment after pesticide application. All pesticides are designed to be sufficiently toxic and persistent to reduce populations of the pest they are designed to control, but most pesticides also poison fish and wildlife, contaminate food sources, destroy animal habitats and, moreover, are toxic to humans—representing a significant threat to human health when present in the water supply (Entry and Sojka, 2014). Contamination of surface water by pesticides usually depends on the farming season, whereas contamination of groundwater is more persistent and therefor may have continuous toxic effects on human health if used for public consumption (Herrero-Hernández *et al.*, 2013).

Prior to the 1980s, there was relatively little concern that water resources, especially groundwater, could be polluted by pesticides (Morris *et al.*, 2003). While significant advances have been made in controlling point-source pesticide pollution since then, little progress has been made regarding non-point-source pollution because of challenges related to the seasonality, inherent variability and multiplicity of origins of non-point-source pollution. At the same time, regulatory agencies have long agreed on the effects of pesticides in drinking water and limits to their. For instance, WHO water quality guidelines exist for some pesticides used in agriculture and public health—including for some highly hazardous pesticides—where there is a likelihood of drinking-water contamination (WHO, 2010). While there may be some increased vigilance about the probable negative impacts of using toxins to control pests and diseases in agriculture, a lack of understanding of the status of pesticides in ecosystems still poses a significant challenge. Some of the reasons for this knowledge gap includes: low detection limit requirements; costs of routine and area-wide monitoring; and poor understanding of the fate and processing of pesticides and their transformation products.

Pesticide poisoning in humans is a high-profile concern. Pesticide contaminated soil and water resources hamper development efforts in rural communities that are suffering from acute and likely chronic health effects related to pesticide poisoning (FAO, 2016). The health effects of pesticides depend on the type of pesticide. Some, such as the organophosphates and carbamates, affect the nervous system. Others may irritate the skin or eyes, cause cancer, or affect the hormone system. In any case, exposure to a sufficient amount of almost any pesticide can make a person ill, and the toxicity of some pesticides is so high that even very small quantities can kill a person.

Water degraded by pesticide runoff impacts human health in two ways; the consumption of food products contaminated by pesticides and the direct consumption of pesticide-

contaminated water (FAO, 2013). Contaminated food, for example, mostly fruit and vegetables, is believed to be responsible for about 10 per cent of cancer cases in India (Aktar, Sengupta and Chowdhury, 2009). Furthermore, when pesticides come in contact with bodies of water, they can interfere with the food chain and cause that way. For example, if chemicals such as lead or copper from pesticides enter water bodies, fish can take them up and concentrate them. When people eat such contaminated fish, they can suffer damage to multiple systems including kidney disease (Ajia, 2017).

While there are no reliable figures on how many people suffer from pesticide-related health effects annually, acute pesticide poisoning causes significant human morbidity and mortality worldwide, especially in developing countries, where poor farmers often use very hazardous pesticide formulations. The lack of data on pesticide-related health issues is the result of several factors, including a lack of standardized case definitions. Studies in developed countries estimate that the annual incidence rates of acute pesticide poisoning in agricultural workers is around 182 per million, and 7.4 per million for full-time workers, and schoolchildren. In developing countries, where there may be insufficient regulation, lack of surveillance systems and training, poorly maintained or non-existent personal protective equipment and larger agriculturally-based populations, incidences are expected to be higher (Thundiyil *et al.*, 2008).

According to WHO and UNEP, worldwide there were more than 26 million human pesticide poisonings and about 220,000 deaths per year (Richter, 2002). In the United States alone, there are 67 000 human pesticide poisonings per year, compared to 500 000 in China, where such incidents result in 100 000 deaths per year (Zhang, Jiang and Ou, 2011). Pesticides can also induce various diseases. In China, as in India, it is estimated that 10 percent of all cancer cases are related to pesticide poisoning. Chen (2004) found that the incidence of breast cancer was linearly correlated with the frequency of pesticide use and that the organochlorine pesticide DDT, and its derivative DDE, were likely responsible for breast cancer (Zhang, Jiang and Ou, 2011).

Pesticides may also affect biodiversity by killing weeds and insects, with negative impacts up the food chain. Recent studies have shown that some pesticides that mimic natural hormones interfere not only with the normal functioning of the endocrine system, but also the immune, reproductive and nervous systems of non-target animals. The widely used pesticide atrazine causes male frogs to develop female characteristics at very low concentrations in water, which causes problems for frog reproduction. Glyphosate, another of the world's most common herbicides, is especially toxic to amphibians (e.g. frogs) and causes impaired growth and development and mortality. This is a particular

problem for wetlands near farms. Organophosphate and carbamate pesticides are highly toxic to target organisms, but can also seriously impact birds (FAO, 2013).

Pesticide accumulation in the food chain, with the potential to poison humans and livestock, led to the ban of organochlorine pesticides such as DDT. The adverse health impact of pesticides, among other reasons, led pesticide designers to focus on developing products that do not harm non-target organisms and that are rapidly detoxified. In developed countries, although considerable use of older broad-spectrum pesticides persists, the trend is towards newer pesticides that are more selective and less toxic to humans and the environment and that are effective at lower doses. However, as better knowledge and understanding of the complexity of ecosystems is gained, expectations for true specificity and targeting of pesticides seems increasingly challenging and harder to achieve.

Overall, the four main issues that concern pesticide production and application worldwide can be summarized as follows:

- Some countries still produce or use highly toxic pesticides.
- Pesticides are overused on a variety of crops, such as cotton, vegetables and rice.
- The quality of pesticides is sometimes poor; some countries do not regulate pesticides effectively and thus counterfeit and illegal pesticides are in use.
- Pesticide residue standards are not implemented effectively (Vaagt, 2008, as cited in Zhang, Jiang and Ou, 2100).

The continued use of obsolete pesticides[6] also poses a significant risk in the developing world, where many older, non-patented, more toxic, environmentally persistent and inexpensive chemicals are still extensively used, creating serious acute health problems. Additionally, it is estimated that approximately half a million tonnes of toxic chemicals are contaminating soil and water resources as they slowly leak from outdoor containers, and eliminating dangerous stocks of pesticides is a development priority. Few developing countries have a clear policy concerning pesticides. There is a lack of rigorous legislation and regulations to control pesticides and few training programmes for personnel to inspect and monitor use and initiate training programs for pesticide consumption. Furthermore, there is growing concern in these countries that few farmers or consumers are aware of the extent of pesticide residue contamination of local fresh produce purchased daily or the potential, long-term, adverse health effects on consumers (FAO, 2011).

[6] Obsolete pesticides are defined as stocked pesticides that can no longer be used for their original purpose or any other purpose and therefore require disposal (FAO, 2017b).

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CHAPTER 6. SALTS

Javier Mateo-Sagasta and Joost Albers

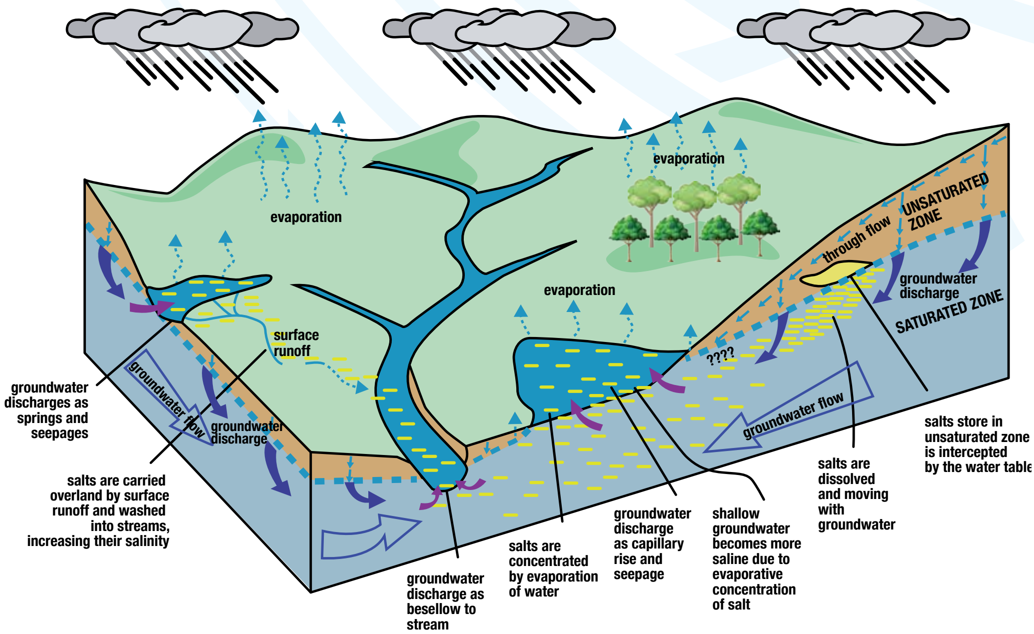
Several decades ago there were few constraints to the disposal of drainage water from agriculture. Now, human-induced salinization of freshwater bodies is a challenge of growing concern with major potential economic impacts, particularly in arid and semiarid areas. The 1260 km^3 of return flows that agriculture is estimated to generate globally every year (FAO AQUASTAT) could result in the mobilization and transport of billions of tonnes of salts to freshwater bodies. The agriculture-induced intrusion of saline groundwater or seawater to freshwaters adds to the problem, especially in coastal areas.

This chapter will briefly review the main processes responsible for salt mobilization with a focus on human-induced salinization of freshwater and with particular attention to salts mobilized by irrigation. The chapter aims to provide a concise review of the extent of salts mobilized by agriculture and consequent effects on human and ecosystem health.

6.1 Agriculture-induced salt loads to water

Salinity is a measure of the quantity of dissolved salts in water, also known as total dissolved solids or total dissolved salts (TDS). Freshwater bodies can receive salt through different pathways, for example through direct surface runoff from saline lands, subsurface drainage of saline waters to fresh water bodies, or the interception of saline stores due to the elevation of the ground water table that also may recharge surface waters (see Figure 6.1). Salts may degrade water quality in fresh water bodies such as wetlands, streams, lakes, reservoirs and estuaries as a result of salt mobilisation and concentration.

FIGURE 6.1 Salinization processes of soil and water bodies and the hydrological cycle



Source: New South Wales Government, 2013.

Table 6.1 provides an overview of categories of salinity and accompanied concentration levels as proposed by Freeze and Cherry (1979). Dissolved salts typically include ions such as sodium (Na^+), chloride (Cl^-), potassium (K^+), magnesium (Mg^+), sulphate (SO_4^{2-}), calcium (Ca^{2+}) and bicarbonate (HCO_3^-). These salts accumulate in the soil profile over time in areas where evaporation levels are higher than precipitation levels and, eventually, may be washed out to water bodies or percolate to groundwater (Mateo-Sagasta and Burke, 2010).

Table 6.1 | Classes of salinity and salt concentration levels

Class name	Class limits (TDS range, in mg/l)		
Fresh water		<	1 000
Brackish water	1 000	-	10 000
Saline water	10 000	-	100 000
Brine		>	100 000

Source: after Freeza and Cherry, 1979.

Salinization of freshwater has many different causes that could be categorized as natural or human-induced. Natural salinity refers to the 'primary' salinity that was present prior to the development of land for agriculture. Human-induced salinity refers to the 'secondary' salinity often caused by changes in land use.

6.1.1 Natural salinization of freshwater

Natural salinization of freshwater occurs when salts enter bodies of water through natural processes:

- natural inflow of salt groundwater into freshwater aquifers;
- weathering of salt containing rocks within the catchment due to precipitation;
- single submergence event of soils under seawater in coastal areas; and
- atmospheric precipitation, both coastal and inland, of rainwater that includes dissolved salts coming from evaporated seawater.

The rates at which these natural processes occur depend on factors such as climate and geology (Ghassemi, Jakeman and Nix, 1995; Herczeg, Dogramaci and Leaney, 2001; Post and Abarca, 2010; Williams, 2001; Williams, 1987).

6.1.2 Human-induced salinization of freshwater

Human-induced freshwater salinization, or 'secondary salinization' (Williams, 2001; Cañedo-Argüelles, 2013), is often attributed to land-use change, poor land management and agricultural activities (including irrigation and drainage). However, salinization can also occur as a result of discharges of municipal or industrial wastewater, salt mining, de-icing of roads and leaking canals and reservoirs (Anning and Flynn, 2014).

Land-use change may involve the clearance of native vegetation to use the land for crops and pastures, which have increased by 460% and 560% respectively in the last 300 years at the expenses of forests and grasslands (Klein Goldewijk, 2001). This change in land use has decreased evapotranspiration and increased aquifer recharge (by two orders of magnitude) and streamflow (by one order of magnitude) but also degraded water quality by mobilization of salts and salinization caused by shallow water tables (Scanlon *et al.*, 2007). In addition, when native vegetation has long root systems, these can take up shallow water and thus prevent a rise in groundwater. When native vegetation is cleared and replaced by shallower-rooted crops and pastures, net evaporation declines, which

results in a rising groundwater table. When the underlying groundwater is saline and rises to the surface, water bodies are recharged with saline water (Mateo-Sagasta, 2010) A well-known example of this is the wetland adjacent to the Murray river in Australia, where salinity increased after the clearance of native vegetation (Walker, Bullen and Williams, 1993).

Human-induced water salinization can be also specifically related to agricultural irrigation and drainage in multiple ways. For example:

i) excessive irrigation can raise water tables from saline aquifers and this can increase seepage of saline groundwater into water courses and increase their salinization;

ii) salts accumulated in soils (particularly in arid and semiarid areas) can be mobilized by irrigation with the application of leaching fractions for soil-clearing. Soil leaching entails allowing an excess portion of the irrigation water to carry salts away through drainage schemes. Drainage water is typically 4-10 times more saline than irrigation water but, when reclaiming already salinized soils, drainage water will be much more saline (e.g. 50 times more than irrigation water). This effluent risks salinization of receiving water bodies (van Hoorn and van Alphen, 2006);

iii) overexploitation of groundwater for agriculture in coastal areas, which results in sea water intrusion into freshwater aquifers;

iv) excessive fertilizer application may increase the concentration of salts in drainage water in irrigated areas and also in run off and percolation in rain fed areas.

In irrigation systems, salt, once mobilized, can be transported and discharged to surface drainage and river systems as a result of groundwater seepage, surface runoff, engineered subsurface drainage and irrigation channel outfalls (Duncan *et al.*, 2008). Furthermore, salt discharge will change over time as a result of both climatic and management influences.

Table 6.2 shows examples of salt loads from irrigated lands in different global locations. Such data are not well documented and are available only for some countries, mostly in arid and semiarid regions, where soil and water salinization are typically of greater concern than in humid areas. As illustrated by the table, salt mobilisation varies widely between regions and irrigation areas, even when these have similar climatic conditions. The load of salt exported per hectare of agricultural land depends on the drainage volumes (which in turn depend on the irrigation management practices and water use efficiency) and the concentration of salt in drainage water (which depends

Table 6.2 | Salt loads from irrigation return flows in different arid and semiarid areas

Location	Salts in irrigation return flows (TDS)	Year	Source
Different irrigation districts, Ebro river basin, Spain	2-16 Mg/ha/year	1990-2004	Causape *et al.*, 2006
La Violada irrigation district, Ebro river basin, Spain	19.3 Mg/ha/year	80'-90'	Barros, Isidoro and Aragüés, 2012
La Violada irrigation district, Ebro river basin, Spain	9.9 Mg/ha/year	2006-2008	Barros, Isidoro and Aragüés, 2012
Lerma watershed, Ebro river basin, Spain	1.3-5.8 Mg/ha/year	2004-2008	Abrahao *et al.*, 2011
Harat Plan, Yazd Province, Iran	1.0-35.4 Mg/ha/year	2009	Jafari *et al.*, 2012
Colorado River Basin, USA	26-41 Mg/ha/year	1975-1984	Duncan *et al.*, 2008
Shepparton irrigation region, Murray-Darling river basin, Australia	0.04-0.66 Mg/ha/year	2003	Duncan *et al.*, 2005
Kerang irrigation region, Murray-Darling river basin, Australia	3.7-10.1 Mg/ha/year	1997-2003	Duncan *et al.*, 2005
Murrumbidgee/ Coleambally irrigation areas, Murray-Darling river basin, Australia	0.1-2.3 Mg/ha/year	1995-1997	Duncan *et al.*, 2005

on factors such as the soil salinity or saline groundwater seepage) (Causape *et al.*, 2006; Duncan *et al.*, 2008). In areas where evapotranspiration is higher than precipitation, salts tend to accumulate naturally in the soil profile and, with irrigation or after heavy rains, salts are mobilized and loads from farms to downstream waters tend to be high (Abrahao *et al.*, 2011). In terms of irrigation practices, efficient irrigation methods, such as drippers and sprinklers, reduce return flows and, therefore, overall loads.

For example, in La Violada irrigation district, in the Ebro river basin (Spain), investments in adequate management of irrigation water reduced by half the salt exported from the irrigation district between the 1980s and 2006-2008 (Barros, Isidoro and Aragüés, 2012). In the Shepparton irrigation region, Murray-Darling River Basin, Australia, the salt loads were kept low thanks to the low volumes of drainage water and the low concentration of salt in drainage. The low concentration of salt in drainage was mainly due to the low salinity of irrigation water (0.06 dS/m) and to the low contribution of groundwater seepage to irrigation return flows, which is due to low connectivity between surface and groundwater systems and relatively good groundwater quality (Duncan *et al.*, 2005). In other regions such as the Colorado River Basin, in the United States of America, high volumes of highly saline drainage water are discharged. This is associated with the inefficient use of water at farm level and substantial losses during water conveyance, and sometimes with the displacement of saline groundwater through deep percolation of irrigation return flows (Duncan *et al.*, 2008).

The contributions to water salinization by aquaculture and livestock (excluding the production of animal feed) are minor compared to irrigated crops, with only localized effects where livestock and aquaculture are more intense.

6.2 Salinization of soils, groundwater and surface waters

6.2.1 Soils

Irrigation causes salinization of soils in many parts of the world (Figure 6.2) and where soils are salinized, water salinization is an accompanying problem. Worldwide, an estimate 24 percent of the area under irrigation is affected by salinization and water logging in the broadest sense. This equates to 65 million ha, of which 34 million ha faces severe salinization (Mateo-Sagasta and Burke, 2010). Asia and the Americas experience the greatest reported area salinized due to irrigation (Table 6.3). At a country level, Pakistan (7 Mha), China (6.7 Mha), United States of America (4.9 Mha), India (3.3 Mha), Jordan (2.3 Mha), Uzbekistan (2.1 Mha), Iran (2.1 Mha), Iraq (1.8 Mha), Turkey (1.5 Mha), and Turkmenistan (1.4 Mha) lead the absolute rankings (FAO-AQUASTAT).

6.2.2 Freshwater

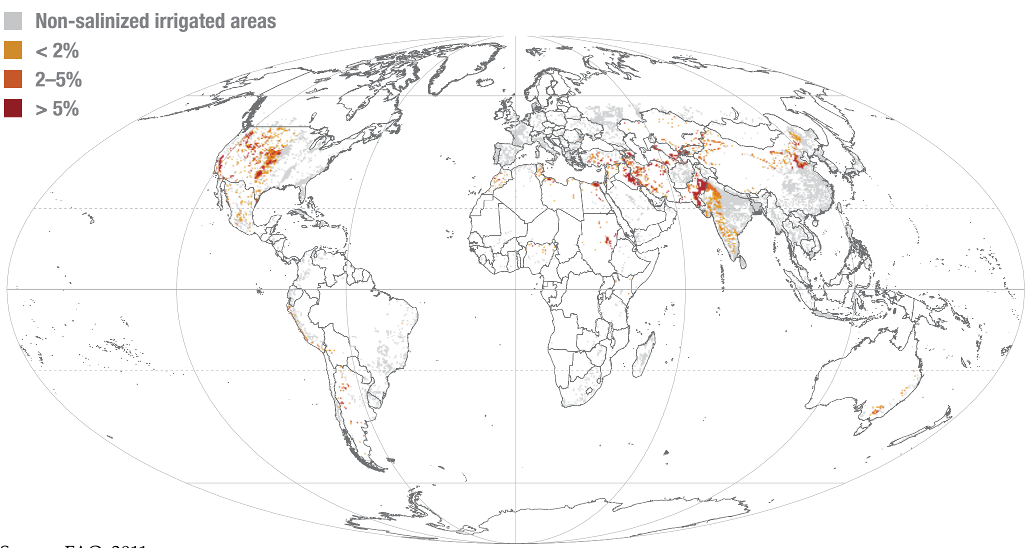
Freshwater salinization is a major environmental problem affecting surface and ground water. Water scarcity is rising, over-abstraction of groundwater occurs in many places, salinization of freshwater bodies is increasing, and aquifers are intruded by seawater in several different coastal areas (FAO, 2011). Salinization of freshwater systems mainly

Table 6.3 I Area salinized by irrigation per region

Region	Million ha
South Asia	10.30
East Asia	6.70
Western Asia	6.12
Northern America	5.34
Central Asia	3.21
Southern America	0.95
Sub-Saharan Africa	0.68
Northern Africa	0.68
Australia and New Zealand	0.20
Total	34.19

Source: AQUASTAT, different years.

FIGURE 6.2 Land salinization due to irrigation. Legend shows the percentage of land salinized by irrigation



Source: FAO, 2011.

affects groundwater (IGRAC, 2009) but also rivers (Cañedo-Argüelles *et al.*, 2013), wetlands (Herbert *et al.*, 2015) and reservoirs (Meybeck, 2004).

Many examples exist of irrigation causing increased salinity of rivers: the Breede River in South Africa (Scherman, Muller and Palmer, 2003), the Amu Darya river in Central Asia[7] (Crosa *et al.*, 2006) and the Murray-Darling River system in Australia. In addition, there are coastal aquifers that have already been permanently salinized, for instance, in Gaza, Gurajat, some coastal areas in Mexico or West Java (Mateo-Sagasta and Burke, 2010). In the Great Menderes river in Turkey, increased salinity has resulted in the extinction of carp (*Cyprinus carpio*) andcatfish (*Silurus glanis*) (Koç, 2008). The broader picture is harder to discern. Despite the existence of many well-documented cases, there is not enough information with the correct geographical and temporal resolution to construct a systematic and quantitative global assessment of surface water salinization. UNEP (2016) tried to address the limitations in the availability of global data with a water quality modelling effort for Latin America, Africa and Asia, using various assumptions and proxies to overcome gaps in the data. This exercise suggests that severe (> 2000mg TDS/l) and moderate (450-2000mg TDS/l) salinity pollution affects around one-tenth of all river stretches in these three regions. The assessment attributed most of the salt loading to irrigation return flows in Africa and Asia, while in Latin America most loadings were attributed to the manufacturing industry.

6.2.3 Groundwater

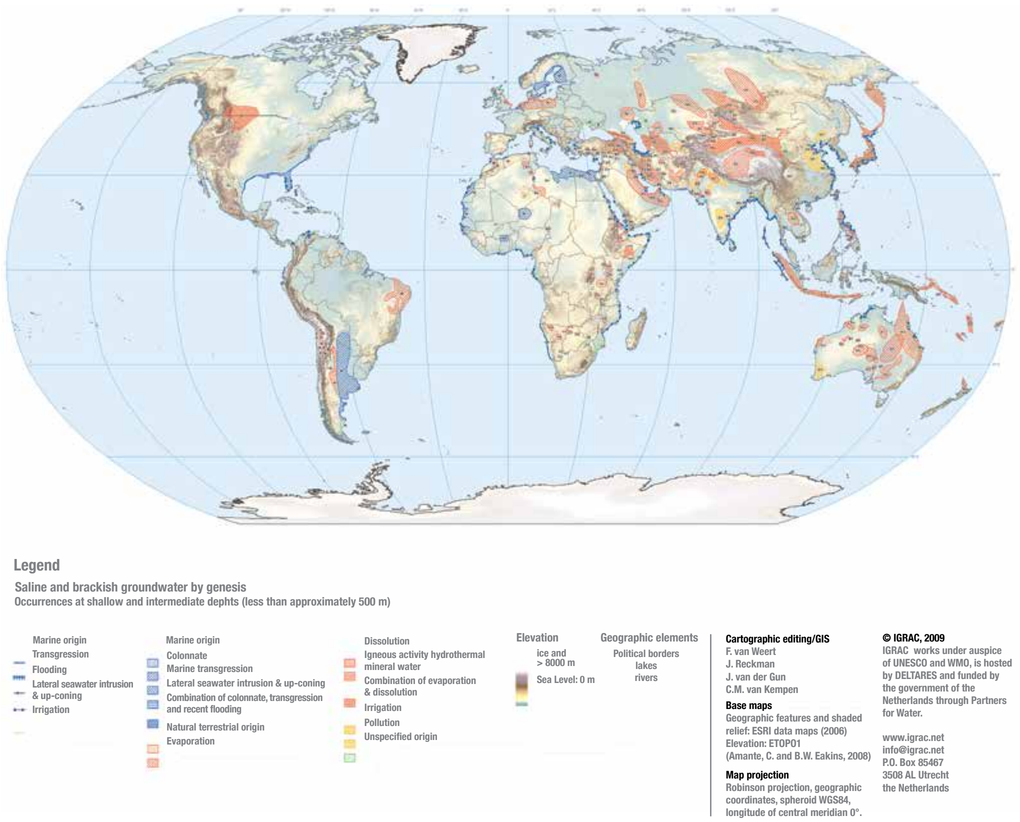
Data are too patchy to allow a quantitative global assessment of groundwater resource status (Foster *et al.*, 2013). Nevertheless, the available data suggest that saline water from irrigation is probably one of the most widespread causes of groundwater quality

BOX 6.1 | Enhanced salinity in the Aral Sea

In 1960, to promote agriculture, the Soviet government decided to establish dams and extensive irrigation programs along the Syr Daria and Amy Daria rivers, which drain into the Aral Sea. These two rivers delivered four-fifths of the water to the Aral Sea, while one-fifth came from rainfall. As a consequence of the dams and irrigation, the level of the Aral Sea dropped by 20 m and the volume shrank from 1060 km^3 to 210 km^3 between 1960 and 1998. Salinity rose from 10 g/l to 100 g/l in the southern part of the Aral Sea, varying over time with precise location (Thompson, 2008). The salinization process has been accelerated by positive feedback arising from stratification of salts and temperature in the Aral Sea.

[7] See also Box 6.1.

FIGURE 6.3 Saline groundwater occurrence and genesis



Source: IGRAC, 2009.

deterioration (Morris, 2003; IGRAC, 2009). In 2009, approximately 1.1 billion people lived in regions that have saline groundwater at shallow and intermediate depths. In these areas, groundwater salinity is mainly caused by irrigation and seawater intrusion and, to a lesser extent, by dissolution and igneous processes (IGRAC, 2009). In Figure 6.3, yellow spots indicate substantial saline groundwater caused by irrigation in North America (USA), the Middle East (Azerbaijan, Iraq, Syria and Turkey), Asia (China, India, Pakistan, Tajikistan, and Uzbekistan) and south-west Australia.

6.3 Impact on the environment, human health and economy

Agriculture-induced salinization of waters can affect environmental health (including biodiversity and ecosystem functions), economic activities (and especially crop production) and human health. Each of these impacts is discussed below.

6.3.1 Environmental health

Highly saline freshwaters alter the geochemical cycles of other major elements, e.g. carbon, nitrogen, phosphorus, sulphur, silica and iron. Salinized water can potentially increase release of nitrogen, phosphorus and silica, which could: enhance eutrophication instream and downstream; disrupt natural processes like denitrification; reduce storage and increase mineralization of carbon; and increase generation of sulphur compounds, which are toxic to plants and animals. However, the extent to which biogeochemical cycles are altered depends on soil and water chemistry, the magnitude and timing of salinization, hydrology, availability of substrate, and reaction of the biota community to higher salinity concentrations. For more information on the effects of salinity on biogeochemical cycles see Herbert *et al.* (2015).

While in some cases, salinized waterbodies (e.g. wetlands) maintain very high levels of biodiversity, in general when salinity rises, biodiversity of all forms—including microorganisms, algae, plants and animals—declines (Pinder *et al.*, 2004; Pinder *et al.*, 2005; Lorenz, 2014). Salinization can affect freshwater biota at three levels: changes within species, changes in community composition, and eventually biodiversity loss and migration. High concentrations of sodium and chloride ions in freshwater causes accumulation of toxic salts in the cells of plant species, which disturbs the uptake of water and important ions and eventually leads to death (Kozlowski, 1997). Other consequences of freshwater salinization are changes in behaviour, food uptake, growth, germination, seedling survival and reproduction (Herbert *et al.*, 2015).

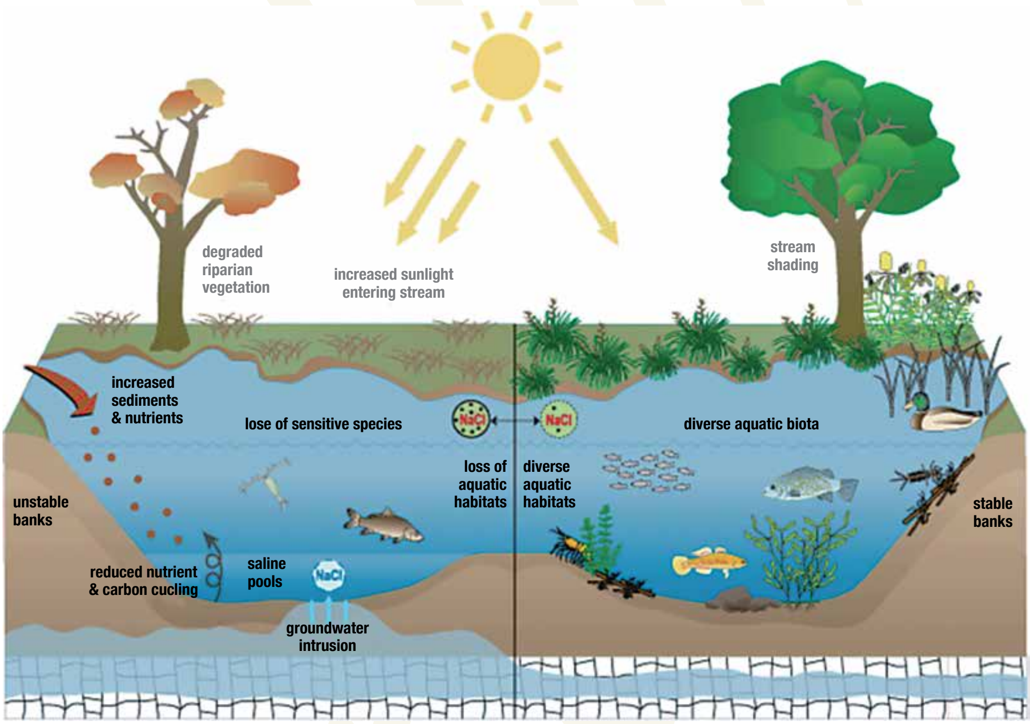
As salinity increases, saline-sensitive species are replaced by more salt tolerant species (Pinder *et al.*, 2004; Nielsen *et al.*, 2003). Increased salinity also provides the opportunity for salt-tolerant invasive species to take hold (Thouvenot, Haury and Thiébaut, 2012). For example, in the Aral Sea freshwater species disappeared as a consequence of increasing salinity from 1960 onwards. By 1990, five fish species were still left, of which only one was indigenous (Kolar and Lodge, 2000). Some freshwater species can be very sensitive to increases in water salinity, even when the water can be acceptable for drinking purposes and irrigation. For example, electrical conductivity (a proxy for salinity) of 2 mS/cm) can displace many freshwater insect species (Cañedo-Argüelles *et al.*, 2016).

One of the first signs of salinization is the disappearance of riparian vegetation and macrophytes, because salts accumulate in the root system and hinder the plants' uptake of water and nutrients (Williams, 2001; Dunlop, McGregor and Horrigan, 2005). Motile species may attempt to avoid increased salinity by migrating to areas with less saline water (Cañedo-Argüelles *et al.*, 2013). For example, fish may move to shallow water where conductivity (salinity) is lower (Dunlop, McGregor and Horrigan, 2005).

Ultimately, salinity negatively affects ecosystem function as a result of positive feedback loops induced by altered geochemical cycles, species healthiness, community composition, or biodiversity loss and migration. Figure 6.4 depicts an overview of salinity impacts on freshwater ecosystems. In the long-run, genetic diversity might be reduced, thereby reducing ecosystem resilience to external shocks and disturbance (Dunlop, McGregor and Horrigan, 2005).

As mentioned above, plant life in riparian zones may be diminished by saline waters and therefore provide less canopy to protect the water from sunlight. More light entering rivers causes a shift from heterotrophic to autotrophic communities (Millán *et al.*, 2011). Another consequence of less abundant riparian zones is higher nutrient flows into river systems, since plants in the riparian zone normally capture nutrients in runoff and groundwater (Dunlop, McGregor and Horrigan, 2005).

FIGURE 6.4 **Overview of salt impacts on aquatic ecosystems**



Source: Dunlop, McGregor and Horrigan, 2005.

6.3.2 Economic activities

Salinization entails economic consequences, since ecosystem services such as provisioning of food and regulation of water quality are impaired. If fish populations decrease or change as a result of water salinization, for example, incomes and food security of fishers may suffer.

High salt concentrations prevent the uptake of water by plants causing reductions in crop yields. Salts accumulate in the root zone to such an extent that, as a result of increased osmotic pressure, the crop is no longer able to extract sufficient water from the salty soil solution. If water uptake is appreciably reduced, the plants rate of growth slows, with symptoms that resemble those of drought. In the early stages, soil salinization reduces plant productivity, but in advanced stages it kills all vegetation and transforms fertile and productive land to barren land. With this in mind, the Food and Agriculture Organization recommends limits to the use of saline water for irrigation (Ayers and Westcot, 1994). Restrictions on use for irrigation start at a concentration of 450 mg/l TDS, a concentration that is not unusual downstream of irrigation areas in semiarid regions.

Good livestock production also requires water of sufficient quality. The effect of water salinity on livestock health and productivity depends on many factors, including the species, breed and age of the animals drinking the water, the water and mineral content of the animals' feed, the temperature of the climate and the water, and which minerals are present in the water (Curran and Robson, 2007). Different species differ in their tolerance of drinking water salinity. While poultry and beef cattle are more sensitive, pigs can tolerate more saline water. When they first encounter saline water, animals may initially be reluctant to drink and may show symptoms of diarrhoea. When water salinity is too high, loss of production and decline in animal health should be expected.

Calculation of the economic impact of salinization of land and freshwater bodies remains under-researched in many parts of the world. A review of previous studies shows a very limited number of highly variable estimates of the costs of salt-induced land degradation (Qadir *et al.*, 2014). This review suggests that the global annual cost of salt-induced land degradation in irrigated areas could be US$ 27.3 billion because of lost crop production. No such global estimate exists for the economic impacts of freshwater salinization, with only a few scattered studies. For example, in the Border Rivers catchment in Australia, Wilson *et al.* (2004) estimated the costs of water salinity associated with infrastructure damage to be almost $700 000 per year.

6.3.3 Human health

Human health may be affected by salinized drinking water. The maximum allowable intake of sodium is 2 g per day, equivalent to 5 g salt per day (WHO, 2012). For chloride in drinking water the limit is 250 mg per litre (WHO, 2003). The most common health issue related to saline water is high blood pressure (hypertension), which may lead to higher risks of heart diseases and stroke. Other adverse health effects include skin diseases, miscarriages, diarrhoea and acute respiratory infection (World Bank, 2013).
Global exposure to salinized drinking water and the global implications for human health have not been comprehensively assessed. Nevertheless, the effects are well documented locally, such as in the coastal areas of Bangladesh, where sea-water intrusion, poor water management and shrimp farming have caused the salinization of ponds, rivers and tube-wells used for obtaining drinking water. Significant associations were seen between salinity increases in drinking water and the incidence of both pre-eclampsia and gestational hypertension (Khan *et al.*, 2011; Khan *et al.*, 2014).

This chapter has examined how agriculture mobilizes and transports large amounts of salts every year to receiving water bodies with potential severe effects on ecosystems and human health. Impacts are potentially stronger in arid and semiarid areas, where soil salinity is more frequent and where receiving water bodies have less dilution capacity. The agriculture-induced intrusion of saline groundwater or seawater to freshwaters adds to the problem and requires increasing attention as the remediation of salinized aquifers can be a very costly and a long-term endeavour, if possible at all.

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CHAPTER 7. SEDIMENT

Javier Mateo-Sagasta and Joost Albers

Soil erosion and sediment transport are natural processes that have been substantially modified by humans. Anthropic activities have simultaneously increased sediment transport by global rivers through soil erosion, and yet reduced the flux of sediment reaching the world's coasts because of retention within reservoirs (Syvitski *et al.*, 2005).

Sediment is made up of solid particles of soil that consist of minerals and organic matter that move from their site of origin by soil erosion. Soil erosion processes are the detachment, transport and deposition of sediments. Solid particles of soil detach from the soil layer when the soil is uncovered and exposed to raindrops, shearing of water and wind, or surface runoff. Deposition of sediments occurs when the erosive force (e.g. wind or water) is no longer able to move the sediments. Sediments may be moved over land into rivers systems and ultimately end up in the oceans, but frequently they are intercepted and deposited at a wide range of sites such as in a riparian zone, at the bottom of a hill slope, a lake, a reservoir or a floodplain. The deposition process of sediments is called 'sedimentation' (National Research Council, 1993).

Under normal conditions, sediment loss from land is balanced by new soil production through weathering of rocks. Human activities, however, have altered the magnitude of land erosion, which increases when soils are exposed to rain or wind and when their structure is degraded. The key drivers responsible for altered sedimentation rates include deforestation, land clearance for agriculture, inappropriate agriculture practices, earth moving in construction works and mining activities. These human activities lead to higher

sediment loads in river systems (Syvitski and Kettner, 2011; Walling, 2009). Effects of climate change, such as intensified and changed patterns of precipitation, also influence sediment loads to the world's rivers (Walling and Fang, 2003; Walling, 2006, 2008).

This section focuses on the agricultural contribution to soil erosion and sediment transport to water, and its consequences for human health and agroecosystems.

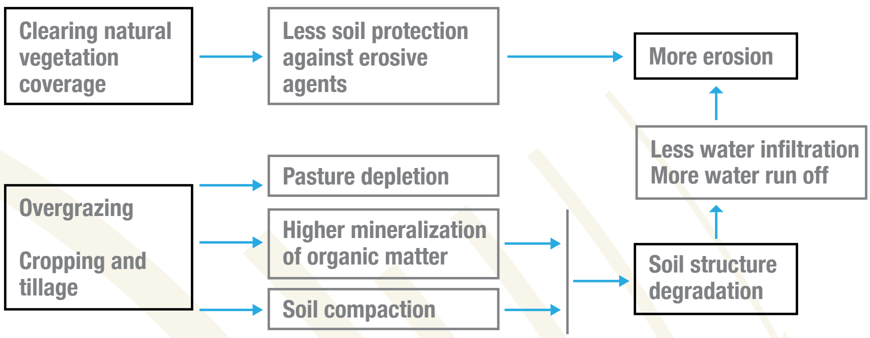
7.1 Agricultural erosion and sediment loads to water

Agricultural activities contribute to increased soil erosion and sediment loads in river systems. The key mechanisms behind agricultural soil erosion are depicted in Figure 7.1. Land clearance, whereby natural vegetation is converted into agricultural cropland or pastures, reduces soil protection against erosive agents and lowers the input of organic matter to soils, which weakens the soil structure. In croplands, excessive tillage can lead to soil compaction and higher organic matter mineralization, which further degrades the soil structure. In addition, inappropriate tillage up and down slopes (as opposed to along the contours) favours soil erosion. In pasturelands, overgrazing reduces vegetation, leaves the soil uncovered and increases soil degradation (e.g. through compaction). Degradation of structure makes soil more vulnerable to erosive agents and can reduce water infiltration and increase run-off, all conductive to more erosion. Uncovered soil also results in faster runoff that exacerbates overland transportation of sediments (Montgomery, 2007; Syvitski and Kettner, 2011). Higher erosion rates decreases soil fertility and reduces biomass productivity, which may lead to additional land clearing or tillage in a negative feedback loop. However, the magnitude of increased sediment loads by land clearance depends on the topography, the extent to which the catchment is affected and the local climate (Benavides and Veenstra, 2005; Walling and Fang, 2003; Walling, 2006, 2008, 2009).

Soil typology also affects erodibility. Most clay-rich soils (e.g. vertisols) have a high resilience because they are resistant to detachment. Sandy soils (e.g. arenosols) are also resilient because of low runoff even though these soils are easily detached. Medium textured soils, such as silt loam soils, are only moderately resistant to erosion because they are moderately susceptible to detachment and they produce moderate runoff. Soils having a high silt content are the most erodible of all soils. They are easily detached, tend to crust and produce high rates of runoff. Organic matter reduces erodibility because it reduces the susceptibility of the soil to detachment (FAO-ITPS, 2015).

Sediments can be physical pollutants but can also be carriers of chemical pollutants and pathogens. Sediments have high ionic exchange rates, which allows them to adsorb contaminants (Ongley, 1996). While being transported over land, sediments can adsorb

FIGURE 7.1 Key mechanisms of agriculture soil erosion



and carry pollutants such as nutrients, pesticides and heavy metals. However, the binding capacity depends on the sediment characteristics (such as the organic matter content). For instance, hydrophobic contaminants (such as some pesticides) bind more easily with organic matter (Dunlop, McGregor and Horrigan, 2005). Pollutants can be released from sediments when the environment (e.g. redox potential or pH) changes. Therefore, in addition to the dissolved pollutants that reach water bodies, sediments can carry more pollutants to aquatic ecosystems, and are effectively the most important pathway for some types of pollutants with low solubility such as phosphates (Coelho *et al.*, 2012), some metals (Peng *et al.*, 2009) and pesticides (Weston *et al.*, 2004). Global estimates suggest that soil erosion by water is responsible for annual fluxes of 23–42 Mt of nitrogen and 14.6–26.4 Mt of phosphorus from agricultural land (FAO–ITPS, 2015), much of which contaminates freshwater ecosystems.

The quantification of agricultural soil erosion at a global scale is difficult because the variability of soil erosion in space and time is extremely high. Nevertheless, there have been a number of attempts to make global estimates, a summary of which is provided in Table 7.1.

Table 7.1 I Global estimates of soil erosion

Soil erosion	Gt/y	Sources
Total soil erosion	50–201	Oldeman *et al.*, 1991; Lal, 2003
Soil erosion by water	20–172	FAO–ITPS 2015; Ito 2007
Soil erosion by wind (on arable land)	<2	FAO–ITPS 2015; Ravi *et al.*, 2011
Soil erosion from agriculture (crops and pasturelands)	20–75	Doetterl, Van Oost and Six, 2012; Wilkinson and McElroy, 2007; Berhe *et al.*, 2007

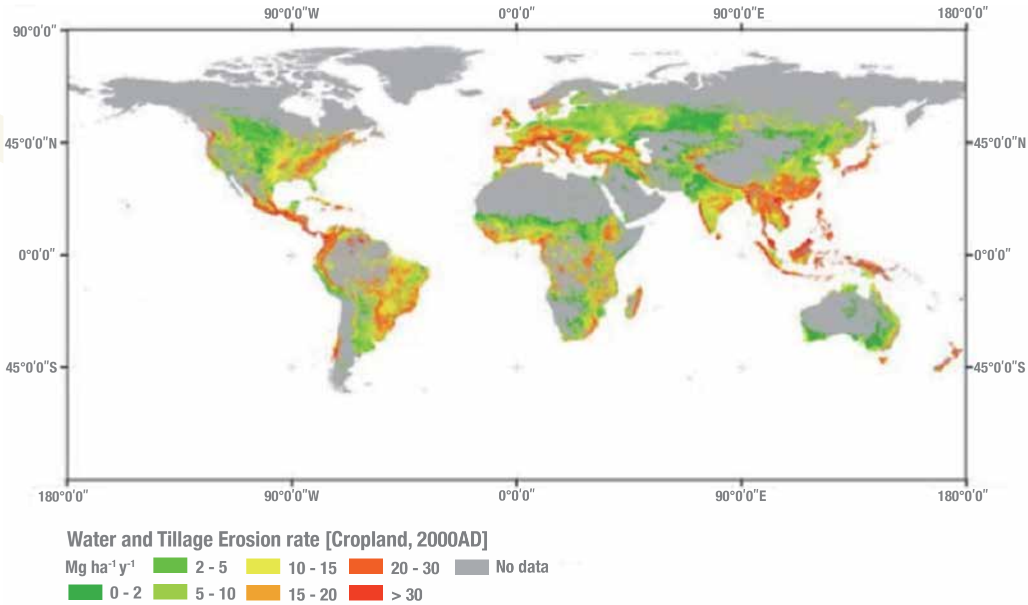
Estimated rates of soil erosion in arable and intensively grazed lands are 100–1 000 times higher than natural erosion rates and far higher than rates of soil formation (Montgomery, 2007). With loss of soils, nutrients are also lost and need to be replaced with fertilizers to maintain fertility, at significant economic cost. For example, using US farm-gate prices for fertilizers, global soil erosion is estimated to cost annually USD 33–60 billion to compensate for nitrogen loss and USD 77–140 billion for phosphorus (FAO-ITPS 2015).

Estimates of global erosion are often based on the extrapolation of results derived from soil erosion experiments at the plot scale. Using these methods, existing estimates of agricultural soil erosion range between 28 and 75 Gt/y (Wilkinson and McElroy, 2007 and Berhe *et al.*, 2007, respectively). Experimental data used in these estimates are derived from a few observations which may be biased towards steep slopes, bare soil and extreme rainfall, and hence may overestimate global rates of erosion. To reduce this uncertainty, a more recent study by Doetterl, Van Oost and Six (2012) parameterized a simplified erosion model driven by coarse global databases, using an empirical database that covers the conterminous United States of America and that represents a wide range of climatic, soil, topographic, cropping and management conditions. The model results showed good agreement with empirical estimates at continental scale and the application of the model globally allowed an estimate of global erosion rates for agriculture (cropland and pastures).

Estimates from Doetterl, Van Oost and Six (2012) show an average annual global erosion rate for cropland and pastures of 10.5 and 1.7 tonnes/ha, respectively, which results in a global annual average of 4.2 tonnes/ha for total agricultural area. Despite these estimated averages, annual erosion rates on hilly croplands and grasslands in tropical and sub-tropical areas may reach 50–100 tonnes/ha (FAO-ITPS, 2015). In terms of total flux, estimates from Doetterl, Van Oost and Six are equivalent to 20.5 Gt of soil per year. This global erosion rate corresponds to an annual rate of 193 and 40.4 kg/ha of soil organic carbon from cropland and pasture respectively. Soil erosion rates are shown in Figure 7.2 for cropland and Figure 7.3 for pastureland. High erosion rates occur particularly in tropical regions where steep slopes and high rainfall coincide. About 43% of the agricultural sediment flux appears to be in Asia (Doetterl, Van Oost, and Six, 2012).

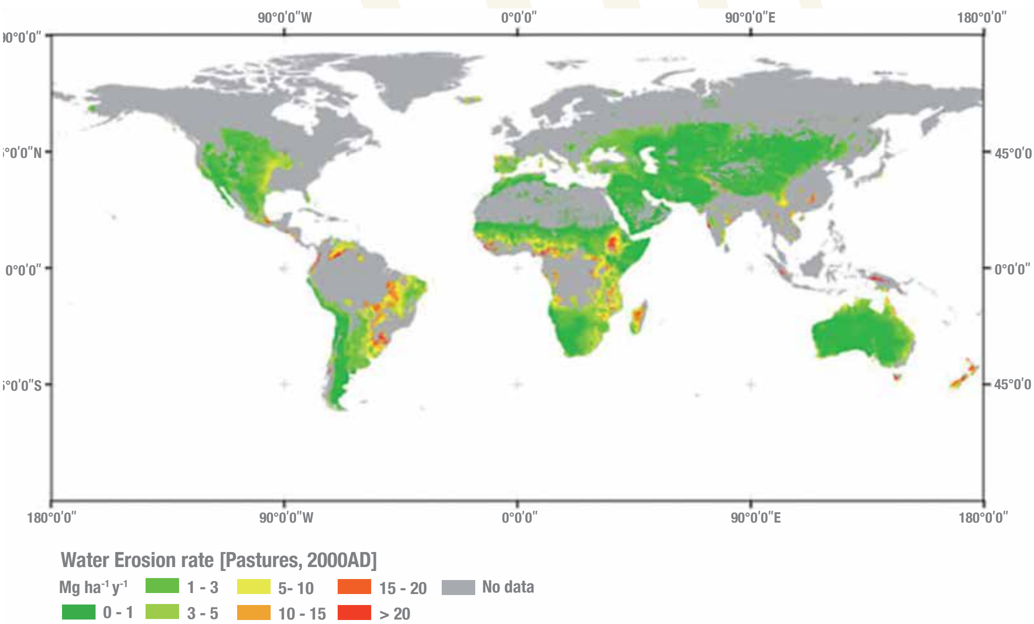
The estimates of soil erosion from agriculture discussed above do not include the soil lost due to erosion from deforestation (unless this deforested land was transformed into agricultural land and captured in global statistics as such). Nevertheless, satellite observations suggest that between 2000 and 2012, 2.3 million km^2 of forest were lost

FIGURE 7.2 Global soil erosion estimates for croplands for the year 2000



Source: Doetterl, Van Oost and Six, 2012.

FIGURE 7.3 Global soil erosion estimates for pastureland for the year 2000



Source: Doetterl, Van Oost and Six, 2012.

globally, while only 0.8 million km^2 were reforested (UNCCD, 2017), with potential effects on soil erosion that are not captured in the preceding estimates.

Within the agricultural sector, the contribution from aquaculture to global erosion is probably small compared to erosion from overgrazing or cropping, but it can be locally important, particularly in coastal areas. One of the largest threats to coastal integrity is the rapid conversion of mangroves into fish and shrimp ponds. Such conversion across the entire intertidal zone sets off cascading effects that contribute to subsidence and erosion of the coastline. Removal of mangrove forests increases the exposure and vulnerability of the coast to waves and jeopardizes sediment trapping and accumulation, all conducive to more coastal erosion (van Wesenbeeck *et al.*, 2015).

The emerging consensus suggests that erosion rates will increase in response to climate change. A model-based study predicts a reduction in average erosion rates for North America and Europe, but a global increase of about 14 percent by 2090, 65 percent of the increase attributed to climate change and 35 percent to population pressure and changes in land use (Yang *et al.*, 2003).

7.2 Sediment concentration, turbidity and sediment yields in surface waters

Globally, while deforestation and agricultural expansion and intensification have increased soil erosion and the sediment loads to rivers, the flux of sediment reaching the world's coasts has been reduced (Syvitski *et al.*, 2005). Sediment eroded from land may be far higher than sediment actually transported by rivers since the sediment might be trapped and stored, for example at the bottom of a slope, before it enters the river system. In addition, yields of sediment at the river basin outlet do not directly reflect sediment loads into rivers because sediments might be deposited within the river system in reservoirs, river banks and the like. By contrast, sediment yields at the river mouth could be higher than on land when river banks function as a source of sediment.

Despite these complex sediment dynamics, rivers are the most important carriers of sediment from land to ocean. Approximately 95 percent of sediments enter the ocean through river systems (Syvitski *et al.*, 2003). Estimates of the global sediment flux to oceans vary but a relatively recent study by Syvitski *et al.* (2005) indicate it to be 12.6 Gt/y (16.2 Gt/y in a hypothetical scenario with no reservoirs).

Different rivers show different patterns of sediment transport. A survey of 145 rivers, mainly in the northern hemisphere, conducted by Walling and Fang (2003) revealed that

BOX 7.1 Research needs on sediment

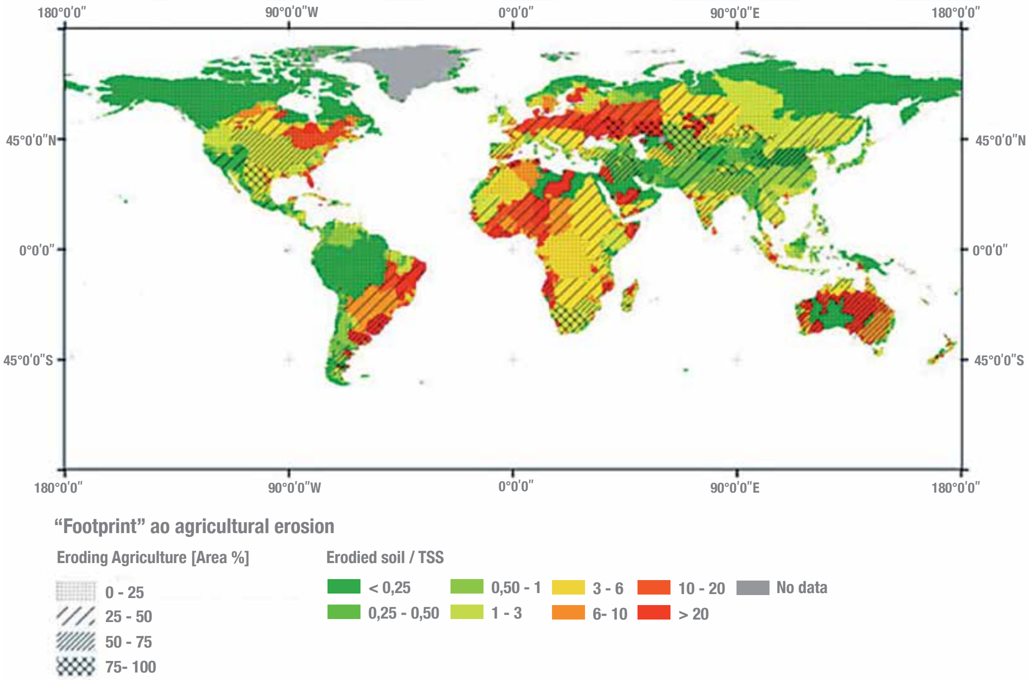
Although rough global estimates exist of agricultural contribution to total sediment fluxes, there is still a need for improved sediment monitoring programmes:

- Monitoring sediments fluxes in smaller rivers instead of only major rivers (Walling, 2009);
- More monitoring of sediment yields for rivers in developing countries. At present, data are mostly available for developed regions such as North America and Europe (Vanmaercke *et al.*, 2011).
- Longer monitoring periods to identify the causes and the extents of sediment fluxes in river systems (Walling, 2009);
- More accurate and reliable research addressing the agricultural contribution of sediments to water bodies – particularly rivers;
- Research on the economic consequences of sediment pollution of water bodies by agriculture.

48 percent of the samples had experienced no change in sediment delivery, 47 percent had decreased and only 5 percent showed an increase. An example of declining sediment fluxes is the Chao Phraya in Thailand where sediment delivery has fallen from 28 Mt/y in 1960 to 6 Mt/y in 1990, without a significant reduction in river outflow. A contrary example is the Rio Magdalena basin in Colombia, where sediment loads at the outlet increased by 40–45 percent between 1975 and 1995 (Walling, 2006).

The relative contribution of agricultural soil erosion to river sediment flux in global major basins is depicted in Figure 7.4. In general, the contribution of agriculture is relatively small compared to natural erosion processes (Doetterl, Van Oost and Six, 2012; Syvitski and Kettner, 2011; Walling, 2009). This is because most of the time sediments from agricultural land are deposited in terrestrial zones such as reservoirs, floodplains and wetlands (Smith *et al.*, 2005). In river basins where the topology consists mainly of mountain uplands, natural erosion processes have a dominant role on river sediment fluxes, for example, the zones in South America, North America and Southern and Eastern Asia shown in green in Figure 7.4. In low lands with intensive agriculture, the relative contribution of human activity on both soil erosion and river sediment fluxes is high, as shown for Central Europe and West Africa in red (Doetterl, Van Oost and Six, 2012).

FIGURE 7.4 Estimated contribution of agriculture to river sediment fluxes for the year 2000



Source: Doetterl, Van Oost and Six, 2012.

7.3 Impact on the aquatic environment and reservoirs

Sediments can affect water bodies physically and chemically with consequences on human health, ecosystems and economic activities.

7.3.1 Physical effects

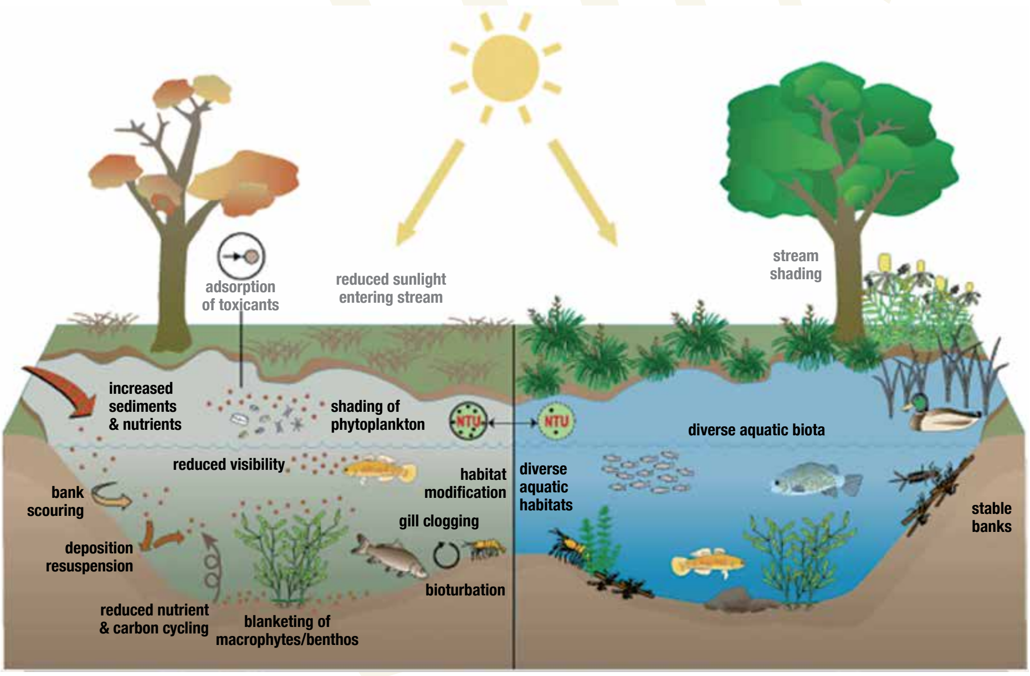
Increased suspended sediments enhance the turbidity of water bodies. Higher turbidity leads to multiple undesirable effects on aquatic plants, algae, invertebrates, and fish (see Figure 7.5), which ultimately results in disturbed functioning of aquatic and terrestrial ecosystems (Dunlop, McGregor and Horrigan, 2005). Less sunlight, the result of increased turbidity, inhibits the photosynthesis and growth of algae and rooted aquatic plants (Li, 2013). Less food is thus available to species higher on the food chain such as fish. Moreover, reduced penetration of sunlight can lower the water temperature, which can alter breeding cues of temperature-sensitive species (Dunlop, McGregor and Horrigan, 2005). In addition, turbidity reduces the visibility of both prey and predators that rely on their sight. Prey are no longer able to avoid predators or find safe places to

live, while for predators reduced visibility hampers their ability to find food (Abrahams and Kattenfeld, 1997). For species that do not rely on sight, high turbidity may be beneficial. Prey and predators with highly developed alternative senses, for example smell, are more successful in turbid conditions (Dunlop, McGregor and Horrigan, 2005).

High sediment concentrations also affect fish species in other ways. Fine sediment particles clog and damage fish gills, leading to respiration problems. Some fish species are able to flush their gills, but this requires a lot of energy. If flushing continues for too long, energy reserves are depleted and death may occur. Sedimentation also destroys fish spawning habitats. Deposited sediment forms a blanket over the spawning beds, which inhibits spawning and results in the loss of biodiversity (Dunlop, McGregor and Horrigan, 2005).

High rates of sedimentation disrupts the hydraulics and transportation capacities of the river channel. Accumulated sediments reduce the depth in rivers and increase risks of floods and inundation. Sedimentation also reduces the storage capacity of reservoirs,

FIGURE 7.5 Visualization of in-stream processes and impacts of turbidity on aquatic biota



Source: Dunlop, McGregor and Horrigan, 2005.

which can affect irrigation schemes, reduce water supplies and make hydroelectric power stations less effective (Walling, 2009). More than 100 Gt of sediment are now sequestered in reservoirs built largely in the past 50 years (Syvitski *et al.*, 2005). Wisser *et al.* (2013) suggested that the world may have gone beyond peak reservoir storage capacity because new dams are not compensating for the declining water storage capacity of large reservoirs. A recent study (Basson, 2008) estimates that by 2050, approximately 64 percent of the world's current reservoir storage capacity will have been filled with sediment. Moreover, increased sediment in water bodies can affect industries such as tourism, which in turn may result in devastating economic losses (Benavides and Veenstra, 2005).

7.3.2 Chemical effects

As previously mentioned, contaminants such as nutrients, pesticides and metals easily attach to the surface of sediment particles and exacerbate the pollution of water bodies that receive the sediments. Sediment transport can thus increase the concentration of other pollutants in water (Dunlop, McGregor and Horrigan, 2005; Ongley, 1996). This reduces the water quality for drinking and irrigation and increases the cost of water treatment. The binding capacity of sediments can also alter global geochemical cycles of key elements, especially the carbon cycle (Walling, 2009).

Contaminated sediments also affect aquatic species. Some toxic substances can kill species that live in the benthic environment at the bottom of water bodies. Benthic species – such as worms, crustaceans, and insect larvae – are important food sources for larger animals. When benthic species die, less food is available to larger species such as fish. In addition, the consumption of toxic substances by benthic organisms leads to bioaccumulation of toxins in the organism. These toxins can also move up the food chain as larger animals eat smaller animals that contain toxic compounds, resulting in biomagnification. A high concentration of toxic compounds can kill species that are not resistant, while species that can tolerate the toxins often experience health problems such as tumours, fin rot and disrupted reproduction. Biomagnification may ultimately pose a threat to the health of humans who eat contaminated fish and other aquatic species (Li, 2013).

This chapter has reviewed how human activities and agriculture have increased and accelerated natural erosion rates, resulting in increased sediment loads entering bodies of water. Crops and pasturelands alone are responsible for the mobilization of huge amounts of sediment every year, much of which ends in water. The global cost to society, including the environment, is not well quantified but a simple extrapolation of the local evidence available (ICOLD, 2009) suggests that it exceeds billions of dollars.

7.4. References

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CHAPTER 8. ORGANIC MATTER, PATHOGENS AND EMERGING POLLUTANTS

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While crop production is the main agricultural activity contributing to global water pollution through the release of nutrients, pesticides, salts and sediments, livestock is the principal source of pollution by organic matter, pathogens, and emerging pollutants such as hormones and antibiotics.

Given the growing trends in livestock production (see Chapter 2), the current and predicted importance of pollutants such as zoonotic water-borne pathogens or organic matter, is probably out of the question. Nevertheless, in recent years, there have been growing concerns about new and emerging pollutants found in the aquatic environment. Such pollutants present a new water quality challenge in both developing and developed countries, due to their potential impacts on human and environmental health and the lack of regulations to monitor and control them.

This chapter provides a very brief analysis of the main agricultural sources of these pollutants and their main effects on water quality.

8.1 Organic matter

The main sources of agricultural water pollution by organic matter include livestock-related wastes such as animal excreta, uneaten animal feed, effluents from animal-processing industries and mismanaged crop residues.

Pollution by organic matter is typically measured by biological oxygen demand (BOD): the amount of oxygen used by microorganisms to decompose waste. Livestock-related wastes have the highest BOD levels (see Table 8.1). For example, the BOD of pig slurry is in the range of 30 000–80 000 milligrams per litre, as compared with the typical BOD of domestic sewage at 200–500 milligrams per litre (FAO, 2006).

As is the case for other pollutants, pollution by organic matter is growing because of increasing municipal and industrial wastewater discharge, the intensification of agriculture (including livestock farming) and reduction in river dilution capacity due to climate change and water extractions. Water pollution by organic matter from intensive livestock farming is now significantly more widespread than organic pollution from urban areas, affecting a larger extent of water bodies (Wen *et al.*, 2017). Global meat consumption nearly doubled between 1980 and 2004, with an annual increase of around 3.6% per year; it is expected to double again by 2030 (FAO, 2011). As a result of expected urbanization, further expansion and intensification of the livestock sector (see Chapter 2), and changes in river discharge, more rivers are predicted to be degraded by organic matter beyond acceptable limits. Figure 8.1 shows that by 2050 the world's worst water quality deterioration is projected to occur in India, sub-Saharan Africa and Mexico, with other smaller regions also facing substantial challenges. Pollution in China was the highest and most widespread in 2000 and will remain among the highest in 2050. The intensification of livestock farming is a key driver of water pollution in India, Africa and South America. By contrast, intensive livestock farming in Europe and China is expected

Table 8.1 | Ranges of BOD concentrations for various wastes and animal products

Source	BOD (mg/litre)
Milk	140 000
Silage effluents	30 000-80 000
Pig slurry	20 000-30 000
Cattle slurry	10 000-20 000
Liquid effluent draining from slurry stores	1 000-12 000
Untreated domestic sewage	200-500
Treated domestic sewage	20-60
Clean river water	5

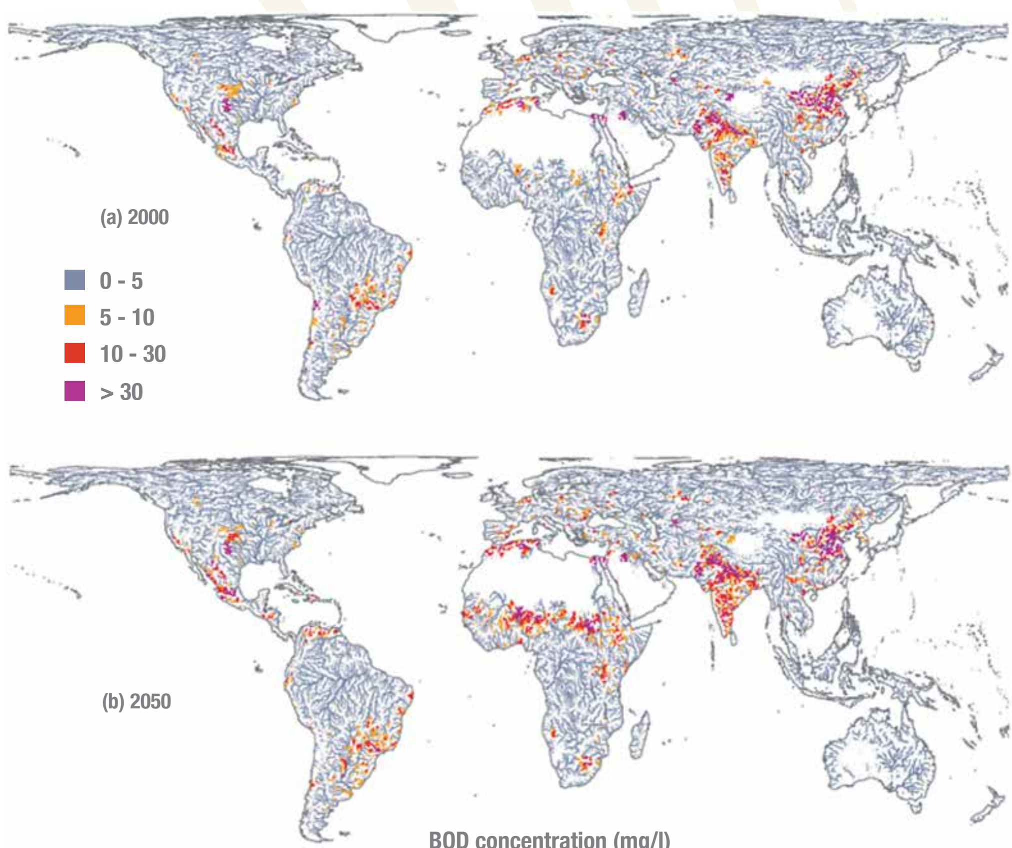
Source: adapted from FAO, 2006.

to stay constant or decline in the coming decades, which may reduce the pressure on very stressed river basins in these regions (Wen *et al.*, 2017).

Locally, aquaculture can be also a major contributor to organic loads in water. In Scotland, for example, the discharge of untreated organic waste from salmon production is equivalent to 75 percent of the pollution discharged by the human population. Shrimp aquaculture in Bangladesh generates 600 tonnes of waste per day (SACEP, 2014).

Organic matter consumes dissolved oxygen in water as it degrades, contributing strongly to hypoxia in water bodies (Malling *et al.*, 2006). The discharge of organic matter also increases the risk of eutrophication and algal blooms in lakes, reservoirs and coastal areas. The number of people thought to be potentially affected by organic pollution (BOD >5 mg/l) is projected to increase from 1.1 billion in 2000 to 2.5 billion in 2050, with developing countries disproportionately affected (Wen *et al.*, 2017). See Figure 8.1.

FIGURE 8.1 Global patterns of computed river BOD concentrations in the years 2000 and 2050



Source: Wen *et al.*, 2017.

8.2 Pathogens

Livestock excreta contain many zoonotic microorganisms and multicellular parasites, which can be harmful to human health. Pathogenic microorganisms can be water-borne or food-borne (the latter especially if the food has been irrigated with contaminated water or with untreated or partially-treated wastewater). Some pathogens can survive for days or weeks in animal faeces that have been discharged onto land and they may later contaminate water resources via runoff (FAO 2006; WHO 2012).

Pathogens from livestock that are detrimental to public health include bacteria such as *Campylobacter* spp., *Escherichia coli* O157:H7, *Salmonella* spp., *Clostridium botulinum* and parasitic protozoa such as *Giardia lamblia*, *Cryptosporidium parvum* and *Microsporidia* spp., all of which cause hundreds of thousands of infections every year (Christou, 2011).

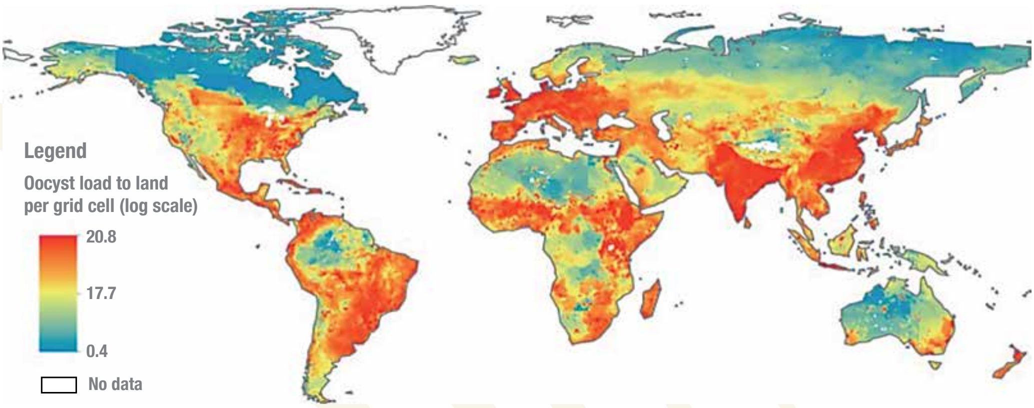
Domestic animals, such as poultry, cattle, sheep and pigs, generate 85% of the world's animals faecal waste, proportionally a far greater amount than is contributed by the world's human population. The faecal production rate and contribution to the environment by these animals can be as high as 2.62×10^{13}kg / year (WHO 2012).

Understanding and quantifying the loads, transport and fate of pathogens are challenging tasks because the environmental pathways are complex and observational data is very scarce (Atwill *et al.*, 2012, Ferguson and Kay, 2012). The risks associated with animal waste are episodic in nature, because of sporadic loads or transmissions (e.g. after rainy events). Further complexity is added by the extreme spatial and temporal variability of weather patterns and livestock management practices in different regions of the world. Therefore, the analysis of loads and environmental pathways, particularly at scales larger than local, are typically based on models.

Cryptosporidium has been proposed as a good indicator for modelling at the global scale as it is a widespread water-borne livestock pathogen that has relatively high incidence in child diarrhoea (Liu *et al.*, 2015). Vermeulen *et al.*, (2017) modelled the global loads of livestock Cryptosporidium, which they estimated to be 3.2×10^{23} oocysts per year. The study showed that cattle, especially calves, are the largest contributors to oocyst loads, followed by chickens and pigs. Spatial differences (see Figure 8.2) are linked to animal distributions. North America, Europe and Oceania together account for nearly a quarter of the total oocyst load, meaning that the developing country regions account for the largest share.

The human risks associated with pathogen pollution of water from livestock have not yet been well defined (WHO, 2012) but they are potentially high, given the number of

FIGURE 8.2 Global oocyst load from livestock to land (E) per grid cell per year



Source: Vermeulen *et al.*, 2017.

outbreaks of infections with zoonotic pathogens that have been reported and documented among swimmers and other water users (Dufour *et al.*, 2012, McBride *et al.*, 2012). For example, in one outbreak in Swaziland, cattle manure was thought to have caused more than 40 000 cases of water-borne infections (Effler *et al.*, 2001). A more recent example is the 2016 outbreak of gastroenteritis on the North Island of New Zealand, which was attributed to the ingestion of water polluted by livestock faeces. Thousands of people were infected (Reiff, 2016).

Agricultural irrigation with untreated or partially-treated wastewater may result in the pollution by human pathogens, particularly in shallow groundwater aquifers. In the majority of low income countries, where most domestic and municipal wastewater goes untreated, alternative approaches are necessary to prevent pathogens from entering agricultural food production chains through wastewater irrigation. The WHO guidelines for the safe use of wastewater, excreta and greywater in agriculture (WHO, 2006) and the wastewater quality guidelines for agricultural use developed by FAO (1985; 1992) provide regulatory recommendations on the suitability of water for irrigation and identify possible restrictions in use.

8.3 Emerging pollutants

Emerging pollutants comprise a wide range of chemicals, substances and microbial pollutants that enter water bodies from various sources, including municipal wastewater treatment plants, agricultural runoff and industrial effluents. Emerging pollutants are also collectively referred to as '*emerging contaminants*' or '*contaminants of emerging concern*'.

Emerging pollutants are broadly grouped into pharmaceuticals, personal care products, pesticides, and industrial and household chemicals. Diverse types are present in highly variable concentrations in freshwater resources such as rivers, streams lakes and groundwater. Currently, more than 700 emerging pollutants, their metabolites and transformation products, are listed as being present in the European aquatic environment (NORMAN, 2016). Nevertheless, they are rarely controlled or monitored and further research is needed to assess their impacts on human health and the environment (UNESCO, 2015b). The potential human health risks of emerging pollutants through exposure to drinking water or agricultural products is a concern (UNESCO, 2011).

Pharmaceuticals are one of the largest groups of emerging pollutants detected in water bodies. For the most part, they are excreted by humans and animals, reaching the water through the means described above. A recent study by UNESCO and HELCOM (UNESCO, 2017) found the presence of 58 different pharmaceuticals (out of 111 monitored) in rivers in the Baltic Sea region. The study reported average concentrations of less than 0.1 μg/l for top 20 pharmaceuticals in river water samples; the highest concentrations exceeded 1.0 μg/l for twelve compounds (UNESCO, 2017). The World Health Organization also has reported the presence of pharmaceuticals used for human and veterinary therapeutic and diagnostic purposes at trace levels in the range of nanograms to low micrograms per litre in surface and groundwater resources (typically less than 0.1 μg/l or 100 ng/l) (WHO, 2011).

Many emerging pollutants are known to interfere with hormone biosynthesis and metabolism in humans and animals and therefore are commonly referred to as 'endocrine-disrupting compounds' (EDCs) (Diamanti-Kandarakis *et al.*, 2009). Endocrine-disrupting compounds include natural and synthetic hormones (such as estrogen), pharmaceuticals with hormonal side effects (ibuprofen, diclofenac, etc.), organochlorinated pesticides and industrial chemicals, plastics and plasticizers, fuels, and many other chemicals that are widely used in household or personal care products.

As noted above, emerging pollutants are released into the aquatic environment from a wide range of sources. Agriculture is a major source of some types of emerging pollutants. The main classes of emerging pollutants arising from agriculture and their potential routes of agricultural releases into surface waters are summarized in Table 8.1 (Boxall, 2012). These pollutants contaminate soil, groundwater and surface water through leaching and/or runoff from agricultural fields and livestock breeding facilities.

A dominant source of emerging pollutants released from agriculture into water arises from the veterinary use of medicines and hormones in aquaculture and veterinary

practices. The overuse of veterinary medicines, such as antibiotics and artificial growth hormones in industrial farming, results in the release of their residues into soil, groundwater and surface waters through leakage from animal waste storage and disposal tanks and the use of animal manure as a fertilizer. Furthermore, the excessive use of antibiotics in agriculture, farming and aquaculture contributes to the development of antimicrobial-resistant bacteria and the presence of these bacteria in water bodies (see Box 8.1). For example, some studies suggest that manure from antibiotic-treated pigs enhances the spread of antibiotic resistance in soil bacterial communities (Heuer and Schmalla, 2007), whereas filtered water from agricultural drains reduced the abundance of aquatic resistant bacteria in a shallow coastal lake (Schallenberg and Armstrong, 2004). The UNESCO and HELCOM study cited above (UNESCO, 2017) also pointed out the lack of data on the sale and consumption of veterinary pharmaceuticals, and their sources, pathways and loading onto soils, surface and groundwater systems and the

BOX 8.1 | Antimicrobial resistant microorganisms and the role of agriculture

Large amounts of antibacterial compounds (ACs) are used across the globe to treat animals. These substances and antibacterial-resistant microbes can be released to the environment during the manufacturing process as well as through the release of animal excreta and the disposal of medical waste. Consequently, a wide range of antibacterial compounds has been detected in surface waters, groundwaters, soils and sediments in different geographical regions (Monteiro and Boxall, 2010) and elevated levels of antibacterial-resistant bacteria and markers for these have been detected in soils and surface waters impacted by human activities.

There is growing evidence that the presence of antibacterial compounds and antibacterial resistant genes (ARGs) in the natural environment indirectly affects human health and contributes to the global antibacterial resistance problem (Wellington *et al.*, 2013), which is predicted to result in millions of deaths by 2050. Once in the environment, these contaminants may persist or dissipate and will be distributed around the different environmental compartments: air, water, sediment, and biota. ACs and other chemical pollutants may also select for resistance in the environment. Humans can then be exposed to the ACs and ARGs through the breathing of dust; consumption of contaminated drinking water, plants, meat, fish and shellfish; recreational and bathing activities; and contact with wildlife. The level of exposure will be driven by a range of socio-economic, health and environmental drivers.

There is an urgent need to manage the loads and the environmental exposure to these contaminants within the agricultural sector and international organizations are already taking action to address this emerging and potentially critical issue (FAO, 2016).

aquatic environment (including agriculture and aquaculture). The study stressed the need to fill this data gap and noted that even the very scarce data available on the sale and consumption of veterinary pharmaceuticals indicate that the annual turnover can be comparable to the amount used in human medicine. Given the extent and continued growth of the livestock sector, and taking into account that animal manure is commonly applied onto the agricultural lands as fertilizer, agriculture could be a significant pathway for medical compounds to reach the aquatic environment in all regions of the world.

Agriculture may become one of the predominant sources of nanomaterials in the aquatic environment, with the use of engineered nanopesticides and nanomedicine in agriculture in the future.

Table 8.2 | Routes of emerging pollutants to surface waters and their importance compared to other sources

Emerging pollutant class	Route of input from agricultural systems	Other sources and routes to the environment	Relative importance of agricultural sources in terms of water contamination
Natural toxins	Release from plants, algae and fungi	N/A	High
Veterinary medicine	Excretion to soils by animals at pasture; application of contaminated manure and slurry to land	Manufacturing releases; disposal of containers	High
Hormones	Excretion of natural and synthetic hormones by animals at pasture; application of manure and slurry to land	Discharge of sewage sludge, containing natural and synthetic hormones from the human population	High – hormonal substances arising from animals Low – hormonal substances arising from the human population
Transformation products (TPs)	Produced from human-induced chemicals that are applied directly to agricultural systems or in activated sludge/irrigation water	Formed in wastewater treatment processes	Depends on the nature of the parent compound; High – TPs of veterinary medicines Low – TPs of pharmaceuticals, personal care products
Nanomaterials	Excretion of nanomedicines by livestock; application of sewage sludge to agricultural land as a fertilizer; irrigation with wastewater or contaminated surface water	Emissions from wastewater treatment plants; disposal of waste to landfill; manufacturing releases	Currently low, as nanomaterials are mainly used in personal care products and paints and coatings Importance could increase in the future as nanopesticide and nanomedicine markets develop

Emerging pollutant class	Route of input from agricultural systems	Other sources and routes to the environment	Relative importance of agricultural sources in terms of water contamination
Bioterrorism/ sabotage agents	Sabotage of crops and livestock	Chemical incidents in cities	Has the potential to be high (depending on the agent)
Human personal care products	Application of sewage sludge to agricultural land as a fertilizer; irrigation with wastewater or contaminated surface water	Emissions to surface waters from wastewater treatment plants	Low
Emerging persistent organic pollutants (e.g. flame retardants)	Application of sewage sludge to agricultural land as a fertilizer; irrigation with wastewater or contaminated surface water	Emissions to surface waters from wastewater treatment plants	Low
Human medicines	Application of sewage sludge to agricultural land as a fertilizer; irrigation with wastewater or contaminated surface water	Emissions from wastewater treatment plants; disposal of unused medicines to landfills; manufacturing releases	Low

Source: adapted from Boxall, 2012.

Agriculture is not only a source of emerging pollutants, it also contributes to the spread and introduction of these pollutants into aquatic environments through wastewater (re) use for irrigation and the application of municipal biosolids onto the land as fertilizers (Bolong *et al.*, 2009; Munoz *et al.*, 2009).

An estimated 35.9 million hectares of agricultural lands are subjected to the indirect use of wastewater (Thebo *et al.*, 2017). The potential long-term effects of emerging pollutants on human health and ecosystems as a result of wastewater use are not yet known (UNESCO, 2015b). The occurrence of a wide range of emerging pollutants in wastewater used for irrigation presents not only potentially-serious risks to human health and food safety through contaminated crops, but also spreads these pollutants to the aquatic environment and soil.

A UNESCO (2018) case study in the Oued Souhil area in Nabeul, Tunisia, indicated the occurrence of emerging contaminants in irrigation water – both in wastewater used for irrigation and in groundwater – as well as in soil. As the use of raw, insufficiently

treated or reclaimed wastewater in agricultural irrigation continues to grow, concerns about emerging pollutants is highly significant since most of these pollutants are not adequately removed during conventional wastewater treatment. Advanced wastewater treatment technologies, (membrane/nano/ultra-filtrations, and reverse osmosis) partially remove some chemicals and pharmaceutically active compounds (Gonzalez *et al.*, 2016). For example, the removal rate of pharmaceuticals during wastewater treatment in secondary and tertiary wastewater treatment plants in the Baltic Sea region were reported at lower than 50% for nearly half of the 118 compounds monitored; only nine were removed with an efficiency higher than 95% (UNESCO, 2017).

The use of biosolids may also lead to the contamination of human food crops and animal feed through the soil as a mediating compartment (Carballa *et al.*, 2007). Pollutants known as endochrine disrupting chemicals (EDCs) have been found to be present in wastewater sludge (Barnabé *et al.*, 2009).

The effects and risks of individual pollutants on aquatic organisms have been studied to a certain extent, whereas potential effects on human health have been evaluated only marginally. Cumulative effects of a mixture of different types of pollutants on human health and ecosystems have not been studied at all (UNESCO, 2011). Several research suggest that endocrine disruptors affect fertility and reproductive health and cause birth defects and developmental disorders, whereas other pollutants may cause cancerous tumours and the development of bacterial pathogen resistance (including multi-persistence) in humans and animals even at low concentrations (Poongothai *et al.*, 2007).

The current scientific understanding on the fate and transport of emerging pollutants and their accumulation in the environment is sadly limited. They behaviour of these pollutants will differ depending on the compartment in which they are found in the environment (e.g., groundwater, surface water and sediment), due to different transformations that may take place, such as the production of by-products (metabolites and transformation products) with different ecotoxicological effects. The by-products of some pollutants are often more persistent than their parent compounds, exhibit greater toxicity and can mimic estrogenic properties (La Farré *et al.*, 2009). Research is needed to improve our understanding of the dynamics of these pollutants in water resources and the environment, and the methods needed to remove them from wastewater (UNESCO, 2015b). More research is also needed to assess the human health and environmental risks and effects of long-term exposure to emerging pollutants, as well as to evaluate their behaviour and accumulation in ecosystems.

There are also huge gaps in the existing regulatory and monitoring frameworks, as well as in data availability regarding the occurrence of emerging pollutants in wastewater and receiving water bodies (UNESCO, 2015b). Efforts to monitor, regulate and control emerging pollutants in water resources and wastewater is limited in both developed and developing countries (UNESCO, 2015a). Data on the presence of emerging water pollutants are very scarce (UNESCO, 2011). Regulations specifically addressing these pollutants are lacking at national and international levels. Emerging pollutants are not only a major challenge facing developing countries, where water quality and pollution control is poor due to inadequate regulatory frameworks and technical and human capacities. It is also a concern in developed countries – even if a country has achieved put in place effective agricultural pollution control measures – because most emerging pollutants are currently not regulated, routinely monitored and controlled.

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CHAPTER 9. THE ROLE OF MODELS

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With contributions from Joost Alberts

Models represent systems in the real world. Using them helps us to gain a holistic understanding of problems by identifying relationships (cause and effect), and enabling future predictions (scenarios). Models can simulate the mobility of pollutants and the resulting changes in the state of water quality. They can help us to understand the impacts of pollutants on human health and ecosystems. Models can also be used to determine the effectiveness and cost of remedial actions. The aim of modelling can be research or management oriented. This chapter is devoted to discussing the application of mathematical models in agricultural water quality management.

9.1 Why are water quality models useful?

As a first step towards effective water quality management, it is necessary to know the current status of water quality and the spatial and temporal distribution patterns of contaminant emissions or loads and concentrations in water environments. For example, if pollutant loads in a given water body are high, identifying where, when and from whom the pollutant originated is necessary to ensure an appropriate response.

Although direct measurements of water quality status can be obtained through monitoring, the question of origin cannot be easily answered by simply relying on water quality monitoring data. Agriculture pollution typically comes from diffuse sources and pathways

(Mayorga *et al.*, 2010). Compared to point source emissions, diffuse source emissions into surface waters (also called non-point source emissions) are more difficult to measure.

The term diffuse pollution is sometimes thought to imply that the contribution to loads is sourced evenly across all parts of an agricultural landscape. However, this is rarely the case. The pollution emission rate from agricultural land depends on a number of local site properties, such as climate, topography, soil properties, land use, and management practices etc. (Chapin *et al.*, 2011), which can vary significantly over space and time. In addition, the proportion of load that is exported from a given farm or landscape is transported by different pathways driven by water fluxes. Moreover, pollutants stored in bottom surface water sediments can be released from the sediment, increasing the pollutant concentration in water bodies. It is thus hard for a water quality monitoring network, even in developed countries, to have enough station density to identify the main sources in diffuse pollution. Furthermore, the magnitude and timing of emission rates can be highly variable, and are often driven by extreme climate events, such as storms. The high cost of water quality analysis may prevent sampling with enough frequency to capture temporal variability. For all of these reasons, we require water quality modelling tools to help us to explain what we observe.

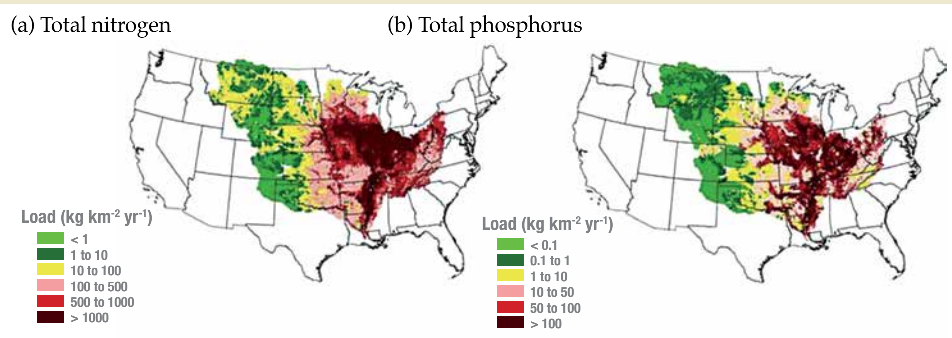
Broadly speaking, water quality models incorporate knowledge about a variety of physical, chemical and biological processes that control the transport, transformation and retention of pollutants. Well-built models can represent pressures, states and impacts at appropriate spatial and temporal scales, and, by linking causes and effects, they offer a way to assess water quality status and identify critical sources of agricultural pollution.

While models could be used to probe the current water quality situation, many water quality models are developed as predictive tools (Argent, 2004). These models can anticipate the effects on water quality as a result of changes in population density, socio-economic development, climate and land use. For example, water quality variation is forced by climate. By introducing climate forcing data, the model can be used to assess the impacts of climate change on water quality. As another example, many water quality models take land use and management practices as input parameters. By varying these input parameters, the models can tell what the water environment quality would be like after land use patterns change or new land management measures are taken. Thanks to their predictive capacity, models are recognized as valuable tools for the development of water quality regulatory programmes and policies. Because the costs of mitigation measures are often considerable and expended well in advance of the materialization of benefits, modelling can be a cost-effective way of to ensure that policies, strategies and actions are on the right track.

BOX 9.1 Using models to understand the role of agriculture in nutrient delivery: the case of the Gulf of Mexico

The Mississippi River and the Atchafalaya River flow through the main agricultural region in the USA. These rivers drain 3.1 million square kilometres in total. The nutrient delivery resulting from intensive agricultural activities in the Mississippi-Atchafalaya River Basin has long been perceived as a culprit for the hypoxia in the Northern Gulf of Mexico. Hypoxia is oxygen depletion in water due to the fast growth of algae blooms stimulated by an over-enrichment of nutrients. In a study by United States Geological Survey scientists (Alexander *et al.*, 2009), the SPARROW (SPAtially Referenced Regressions On Watershed attributes) model was used to estimate the load of nutrients (nitrogen and phosphorus) and contributions of different sources across the river basins, including ungauged areas.

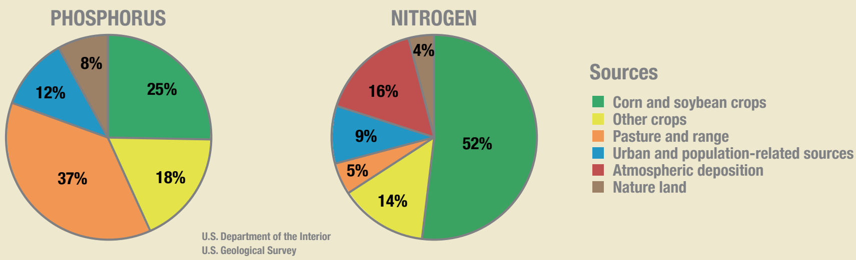
Figure 9.1 I Nutrients delivered to the Gulf of Mexico



Source: National Research Council, 2011.

The SPARROW modelling results showed that agricultural sources contribute more than 70% of the nutrient loads delivered to the Gulf of Mexico. While corn and soybean cultivation contributes 52% of total nitrogen load, manure on pasture and rangeland is the largest source of phosphorus and accounts for 37% of the total phosphorus load.

Figure 9.2 I Sources of nutrients delivered to the Gulf of Mexico



Source: ibid.

BOX 9.2 Water quality models to support policies: the case of the Total Maximum Daily Load (TMDL) programme, USA

To protect water quality from pollution, the United States Environmental Protection Agency (USEPA) launched the Total Maximum Daily Load (TMDL) programme. A TMDL is "the maximum amount of a pollutant that a waterbody can receive and still safely meet water quality standards" (USEPA, 2018). Implementing the TMDL programme involves identifying pollutants, estimating the assimilating capacity of the receiving water body and the current levels of pollution from all sources, determining maximum allowable loads and allocating them to different polluters. The determination of maximum allowable loads and load allocation often requires modelling tools with predictive skill (USEPA, 2004).

In a TMDL case study on nutrients and sediments, the Soil and Water Assessment Tool (SWAT) model was used to evaluate the effects of load reduction under various allocation schemes until a scheme was identified that ensures that the predicted 30-day average concentrations of pollutants at the watershed outlet meet water quality requirements. According to the SWAT simulation results, in some months, nutrients and sediment loads from 29 large, concentrated animal feeding operations (CAFOs) in the study river basin need to be reduced by up to 70%-80% (USEPA, 2004).

9.2 Types, capabilities and limitations of water quality models

Discovering the mechanism and factors impacting water quality and building a water quality model to describe those related processes in mathematical language represents a highly challenging task. It often requires research, which may have considerable costs. Fortunately, practitioners usually do not have to start from scratch. Today, dozens of models with different strengths and limitations are used in the field of water quality. These models operate at different scales (Borah and Bera, 2004; Wang *et al.*, 2013) to support researchers, planners and policy-makers in designing cost-effective measures for addressing water pollution in agriculture. Table 9.1 lists a number of commonly used models with water quality simulation capacity.

Water quality models vary substantially in their complexity and capability, and can be classified in a number of ways. For example, models can be classified on a scale of increasing complexity or scientific rigor.

Input-output models are relatively simple. A typical application of an input-output model is to keep track of nutrient balance. 'Simple' input-output balances can be done

Table 9.1 I Selected models for water quality simulation

Model	Description
MONERIS (Modelling Nutrient Emissions in Rivers Systems)	Designed to calculate emissions of nitrogen and phosphorus to surface waters via different pathways as well as the in-stream retention and transport in the surface water network; moderate demand of input data at river sub-basin level, free of charge, open software license concept (Behrendt *et al.*, 2000; Venohr *et al.* 2011)
GLEAMS (Groundwater Loading Effects of Agricultural Management Systems)	A field scale model developed to evaluate the impact of management practices on pesticide and nutrient leaching (Knisel, 1993)
PELMO (PEsticide Leaching Model)	A 1D model simulating the vertical movement of pesticides in soil by chromatographic leaching (Klein, 1995)
SHETRAN	A 3D finite difference model designed to simulate flow, and sediment and contaminant transfer (Ewen *et al.*, 2000)
QUAL2E & QUAL2K	1D river and stream water quality model that simulates daily water quality parameters, including biological oxygen demand, nitrogen, phosphorus, coliforms and pH (Brown and Barnwell 1987; Chapra *et al.*, 2003; Park and Lee, 2002)
SWAT (Soil and Water Assessment Tool)	Integrated river basin-scale model developed to quantify the impact of land management practices in large, complex watersheds with subroutines designed to simulate transport and fate of nutrients and pesticides (Arnold *et al.*, 1998; Srinivasan *et al.*, 1998)
AGNPS (AGricultural Non-Point Source Pollution Model)	A model developed to estimate pollutant loads from agricultural watersheds; the model can simulate surface water runoff, nutrients, sediment, chemical oxygen demand, and pesticides from point and nonpoint sources of agricultural pollution (Young *et al.*, 1989).
HSPF (Hydrological Simulation Program – Fortran)	An integrated river basin model that simulates runoff and water quality (e.g. nutrients, pesticide, sediment) from various, including agricultural, sources (Donigian, 1995)
L-THIA (Long Term Hydrologic Impact Analysis)	A tool used to evaluate long-term average of runoff and amount of several non-point source pollutants according to land use and soil combinations (Ma, 2004)
WEPP (Water Erosion Prediction Project)	A model that simulates runoff, erosion, and sediment delivery at field or small watershed scale (Flanagan *et al.*, 2007).
BATHTUB	A steady-state water quality model designed to simulate eutrophication conditions in lakes and reservoirs (Walker, 1987; Walker, 1996)

Model	Description
REMM (Riparian Ecosystem Management Model)	A model designed to simulate hydrology, nutrient dynamics and plant growth for land areas between the edge of fields and a water body (Lowrance *et al.*, 2000)
SPARROW (SPAtially Referenced Regressions On Watershed attributes)	Model developed to identify the source and fate of contaminants in large inland watersheds and water bodies by linking water quality monitoring data with watershed attributes (Alexander *et al.*, 2009; Schwarz *et al.*, 2006)
STEPL (Spreadsheet Tool for Estimating Pollutant Load)	This model uses simple algorithms to estimate nutrient and sediment loads from different land uses and to evaluate the effectiveness of implementing various best management practices (Tetra Tech, 2011).
LSPC (Loading Simulation Program in C++)	A watershed modelling tool that is closely related to HSPF with a simplified stream transport module (Tetra Tech, 2009)
GWLF (Generalized Watershed Loading Function)	A watershed model that simulates runoff, sediment and runoff loading (Haith *et al.*, 1992)
WARMF (Watershed Analysis Risk Management Framework)	A modelling system designed to calculate TMDLs for coliform, total suspended solids (TSS), biochemical oxygen demand (BOD) and nutrients and to guide stakeholders to reach consensus on the implementation of a water quality management plan (Goldstein, 2001)
VFSMOD (Vegetative Filter Strip Modelling System)	This system models field-scale processes associated with filter strips or buffers by routing storm runoff from an adjacent field through vegetative filter strip and calculating outflow, infiltration, and sediment-trapping efficiency (Muñoz-Carpena and Parsons, 2009)
PLOAD	A simple GIS-based model that estimates annual non-point source pollutant loads in watersheds (CH2MHILL, 2001)
MIKE	A commercial system that includes a range of models that simulate hydrological and hydrodynamic phenomena and water quality processes at the river basin scale (Refsgaard and Storm, 1995)
Global NEWS (Global Nutrient Export from Water(S)heds)	An integrated model that determines nitrogen, phosphorus and carbon exports through rivers into coastal areas on a global scale. The model enables future projections of nutrient export and the potential coastal eutrophication risks (Mayorga *et al.*, 2010)
ANSWERS (Areal Nonpoint Source Watershed Environment Response Simulation)	A hydrological and sediment transportation model that describes processes of infiltration, drainage, subsurface export, runoff, soil erosion, and sediment transport (Beasley, 1980)

Model	Description
CASC2D-SED	A simulation model that determines water and sediment runoff temporally and spatially. Overland flow is simulated on a two-dimensional grid and channel flow on a one-dimensional grid (Johnson *et al.*, 2000).
DWSM (Dynamic Watershed Simulation Model)	A model that simulates surface and subsurface runoff, propagation of floodwaves, soil erosion, and export of nutrients, pesticides and nutrients in rural and agricultural watersheds during a rainfall event (Borah *et al.*, 2002)
KINEROS (KINematic runoff and EROSion)	A kinematic and event-oriented model designed to simulate hydrological and sedimentation processes in watersheds (Woolhiser, 1990)
INCA (Integrated Catchement Model)	An integrated watershed model that simulates the transport and fate of nutrients, sediment, carbon, metals and mercury in water environments (Whitehead *et al.*, 1998)
WASP (Water Quality Analysis Simulation Program)	A widely used water quality model allowing for 1, 2 and 3 dimensional simulation of in stream water quality processes (Wool *et al.*, 2001)

on a spreadsheet and can be readily used by qualified consultants and farmers. Such models are easy to implement on farms, where record keeping on land management practices is seen as a basic management activity. Although the nutrient balances revealed by the budget model provide little insight on dynamics and processes, it effectively describes long-term average conditions.

Empirical models attempt to relate water quality variables to input variables without paying attention to the processes behind the correlation. A good example of this type of model is the SPARROW (SPAtially Referenced Regressions On Watershed attributes) model, which correlates pollutant loads and in-stream water quality with spatially referenced watershed attributes. The modelling exercise is data driven and tends to have intensive data requirements. In a study on the Mississippi-Atchafalaya River Basin, monitoring data came from 425 stations (Alexander *et al.*, 2007). This feature may restrict the application of empirical water quality modelling techniques in developing countries, where water quality data are typically scarce.

Process models explicitly describe water quality processes according to physical laws or causal relationships. This type of model may constitute the largest class of water quality models. Indeed, most of the models in Table 9.1 fall into this category. Of course, no

sharp dichotomy exists between empirical and process models. Many process models contain empirical elements. In its extreme form, a process-based water quality model consists of a set of equations derived from mass conservation and other laws of chemical and biological kinetics. Process models are typically used to simulate the transport and transformation of pollutants in water bodies. Due to the embedded knowledge in the model, process models may work under conditions in which water quality monitoring data are limited or even in unmonitored regions.

Mixed models combine process-oriented and empirical approaches to model the fate and behavior of chemical substances in water bodies and their catchment. An example of this type of model is the MONERIS (MOdelling Nutrient Emission in RIver Systems, Venohr *et al.* 2011). MONERIS is a semi-empirical and process oriented model, which has gained international acceptance as a robust meso- to macro-scale model for nutrient emissions. MONERIS is used to calculate nitrogen, phosphorus and silica emissions into surface waters, in-stream retention, and resulting loads on a river catchment scale. The model distinguishes between sources (atmospheric deposition, fertilizer application, human disposal and industrial discharges); recipients (urban areas, agricultural and other areas); and emission pathways (atmospheric deposition on surface waters, surface runoff, erosion, tile drainage, groundwater, emissions from sealed urban areas and point sources). Compared to other models MONERIS has a moderate demand for input data, has a short computing time, and is applicable to large river basins. An implemented scenario manager can help quantify the effects of potential regionally differentiated measures to reduce nutrient emissions and loads from agricultural and urban sectors in surface waters. Over the past several years, MONERIS results have been used by various national and international river commissions (e.g. Danube, Oder, Elbe, Weser, Sanggan He, São Francisco) to develop river basin management plans and programes and have been the basis for national reporting obligations (e.g. Germany, Austria).

Models can also be grouped by loading models, receiving water models and integrated models. Loading models are designed to estimate pollutants from sources (e.g. crop land, pasture, feedlot, etc.) while receiving water models simulate the transport and fate of pollutants in water bodies (rivers, lakes, reservoirs, wetland, estuaries, and groundwater, etc.). Integrated models combine knowledge from two or more domains into a single framework. An integrated model can be used to address questions such as how reducing the application of fertilizer to conserve water quality will influence crop yields and what is the trade-off between water quality and agricultural productivity. Answering these questions requires simulation of both water quality and crop production process and the interactions between them.

BOX 9.3 Nutrient fluxes and losses in the Danube river basin

The Danube River is the most international river system in the world (Sommerwerk *et al.*, 2010). It drains a catchment area of 809 000 km² across 19 countries. From the Alps, over semi-arid regions to extended lowland plains, the Danube covers a wide range of hydrogeological conditions and shows a wide variation in land-use intensities (e.g. fertilizer application rates, population densities). Management of the Danube is a special challenge since the share of emission contribution and their effects on the water quality is unevenly distributed among the 19 countries – as are the financial resources for the implementation of management plans. Within the framework of the 1st and 2nd Danube River Basin Management Plans (DRBM) MONERIS (Venohr *et al.*, 2011) was applied to quantify the spatial and temporal pattern of nutrient emissions and loads under contract of the International Commission for the Protection of the Danube River (ICPDR) and country representatives (ICPDR 2009, ICPDR 2015).

The moderate data demand and robust model structure allowed the application of MONERIS to the Danube river basin with different data availability and quality from participating countries as well as a complex mixture of management problems and interests. The basin-wide modelled phosphorous (P) and nitrogen (N) emissions for the reference period (2009–2012) indicated that the diffuse sources dominate, making a contribution of 84% (N) and 67% (P). Whereas groundwater is the most important diffuse pathway for N (54%), soil erosion (32%) generates the highest diffuse emissions of P. The agricultural (N: 42%, P: 28%) and urban water management sectors (N: 25%, P: 51%) are responsible for most of the nutrient emissions (ICPDR, 2015). The economic situation of the countries also reflects the spatial distribution of source emissions. While nutrient emission rates from urban sources were relatively low for upstream countries, urban nutrient emissions become more dominant in the downstream countries, indicating the high potential to improve wastewater treatment. In contrast, N emissions from agricultural areas are higher in upstream countries, due to high nitrogen surpluses on agricultural lands. About 32% of the N and 42% of P emissions in the Danube basin are retained in the sediments of lakes, reservoirs, rivers as well as in connected floodplains before being transported to the Black Sea. Although emissions into the Danube's surface waters and groundwater decreased mainly due to waste water treatment measures implemented over the past decade (N: 12%, P. 34 %), a further nutrient load reduction (N: 40%, P: 20%) has been identified by modelling as necessary for improving the water quality of the Black Sea.

Using a set of measures for the short- (realistic) and long-term (vision) development provided by the country representatives, a further decrease of nutrient emissions was modelled. By implementing ambitious measures, a reduction of 20% (N) and 41% (P) seems achievable, although a trend of decreasing nitrogen emissions in the upstream countries and an increasing trend in the downstream countries due to land use intensification was ascertained.

The last two decades have witnessed the development of a number of integrated models. A good example is the Soil and Water Assessment Tool (SWAT) (Arnold, 1998). The SWAT model is the result of combining features of several predecessor models. For example, CREAMS (Chemicals, Runoff, and Erosion from Agricultural Management Systems) (Knisel, 1980) contributed routines for simulating hydrology, erosion and nutrients; EPIC (Erosion-Productivity Impact Calculator) (Williams *et al.*, 1984) provided the original algorithm for crop growth, and the pesticide component came from the GLEAMS (Groundwater Loading Effects on Agricultural Management Systems) model (Leonard *et al.*, 1987). SWAT also includes modules that implement the QUAL2E algorithm for in-stream water quality simulation (see Table 9.1).

It is also possible to classify water quality models in other ways, which may be more relevant to the technical specification of the models.

Steady state vs. dynamic model: Steady state models assume all input and state variables used in water quality simulation are time-invariant, whereas the dynamic models are capable of simulating time-varying water quality phenomena.

1D, 2D and 3D model or lumped vs. distributed model: The simulation of transport and transformation of pollutants can be carried out in one dimension (1D), two dimensions (2D) and three dimensions (3D). 1D simulation would suffice if the water quality in each longitudinal division is assumed to be homogeneous, while 2D and 3D simulations are required when the water quality variability in other dimensions cannot be ignored, such as on large lakes and estuaries. The terms 'lumped' and 'distributed' are mostly used to classify a loading model. In a lumped model, the study area is regarded as a single entity. By contrast, in a distributed model, there is a partition (usually a grid of cells) in the study region; input variables and model parameters are allowed to vary across the study area.

Continuous model vs. event model: Continuous simulation is used to generate estimates/predictions over a relatively long-term period. Continuous models can be run on a daily, monthly or even yearly basis. Event-based water quality simulation is primarily used to address pollution related to storm events. Such models typically run at hourly or even smaller time steps.

Given the large number of models/modelling techniques that have been developed for water quality simulation, choosing an appropriate model to use is by no means a simple task. When there are options, answering the following questions may help the practitioner decide:

- Can the output from the model satisfy the needs of the study (in terms of reported outcome variables, spatial and temporal resolution etc.)?
- Are the required input data available?
- Are the computational costs affordable?

It is worth noting that computational efficiency may be an important factor in the decision, especially when model calibration and uncertainty analysis are carried out. Models can merely provide a simplified representation of reality. Any modelling activity involves uncertainty (see Box 9.4). Quantifying and analyzing such uncertainty should be an integral part of model-based water quality studies. A number of calibration and uncertainty analysis techniques have been developed and these typically require a large number of model runs.

Finally, while this chapter hopefully provides some support for practitioners choosing an appropriate model to use for their water quality modelling work, there are reviews and comparison studies that provide discussions on this topic from a more technical perspective. (e.g., Borah *et al.*, 2003 and 2004; Kronvang *et al.*, 2009; Malagó *et al.*, 2015; and Wang *et al.*, 2013) Interested readers are encouraged to consult the literature for further information.

BOX 9.4 A caveat on uncertainty in water quality simulation using process-based deterministic models

Process-based deterministic models are widely used for water quality simulation. Uncertainty may arise concerning the model input data and values of model parameters that are used for the simulation. When water quality monitoring data or other observations related to model output variables are available, the parametric uncertainty can be reduced through calibration. A model can be calibrated by selecting parameters that maximize model fit to observed data given certain criteria (e.g. Kling-Gupta efficiency coefficients or the Nash–Sutcliffe model efficiency coefficient). Although this approach is still extensively used in water quality modelling practices, more sophisticated calibration methods for deterministic simulation have been developed (e.g. Beven and Binley, 1992; Kennedy and O'Hagan, 2001; Refsgaard *et al.*, 2007; Efstratiadis and Koutsoyiannis, 2010). These approaches enable predictions or predictive intervals to indicate the parametric uncertainty resulting from model calibration. ➤

➤ Since parameters in process-based models often have physical meaning, a knowledge of parameters from literature or other studies can be used to improve the estimates of these parameters. This idea is particularly useful with regard to modelling unmonitored or poorly monitored regions. A recent well-known endeavour is the International Association of Hydrological Sciences' initiative on predictions in ungauged Basins (Hrachowitz *et al.*, 2013), which investigated the transferability of model parameter values at river basin scale according to watershed attributes.

Uncertainty may also originate from the structure of the model. No matter how sophisticated, a model can only provide an approximate representation of the real world. As Box (1987) observed, "essentially, all models are wrong, but some are useful." A method to cope with model structural uncertainty is model averaging (Hoeting *et al.*, 1999). When alternative models are available, instead of trying to select the 'best' one, a modeller can combine or average the results from multiple models. By synthesizing predictions from multiple models, model averaging helps to improve the accuracy and reliability of the prediction. For example, for a study of the Patuxent estuary, Maryland, USA (Boomer *et al.*, 2013), six models were used to predict water, nitrogen, and phosphorus discharges into the estuary. After comparing the results with observed data, it was found that the predictions constructed by combining simulation results from the six models outperform predictions from any single model.

9.3. Linking the outcome of water quality modelling to water policy

Water quality modelling reports the pollutant loadings from different sources and the resulting concentrations in water environments. When the outcome is used to inform water policy, it is often necessary to carry out further analysis to reveal the implications of different policies on water related ecosystem services.

Effectively linking water quality modelling and water policy requires being knowledgeable about relevant areas, such as water quality standards. Such standards define the water quality goal of a water body according to its designated use and are key elements in water quality management. Agriculture is an important source of nutrient pollution. The limits of nutrients in drinking water have been well established through epidemiological studies (WHO, 2006). However, developing water quality criteria to protect aquatic ecological systems from pollution remains challenging. In some countries, such as China, maximum concentrations of nutrients in ambient water environments are set, and the water quality

standard is enforced uniformly nationwide. This type of water quality standard has the advantage of easing implementation, but it apparently neglects the variability in ecological water quality requirements. In 1998, the United States Environmental Protection Agency (USEPA) initiated an effort to develop numeric region-specific nutrient criteria. As of July 2017, the endeavor was still in progress due to the complexity of determining water quality requirements in ecological systems (US EPA, 2017). In Europe, many water bodies are still affected by pollutants and only 53% were found in 2015 to exhibit a good ecological status. In 2000, the European Environment Agency (EEA) established the European Water Framework Directive (WFD) for European Union member states to achieve the good qualitative and quantitative status of all water bodies in the EU by 2027. To achieve this goal, environmental quality standards and threshold values have to be complied for 41 chemical pollutants across the EU. If these values are exceeded, the contaminant sources have to be examined and measures implemented to restore the good status.

Ecological modelling tools have been developed as part of the effort to address the water quality needs of aquatic ecosystems. A few of such tools are listed in Table 9.2. In a review by Bartell (2001), AQUATOX, CATS, CASM and ECOWIN were ranked as having the

Table 9.2 I Selected ecological modelling tools

Model	Description
AQUATOX	A modelling system distributed by USEPA and designed to predict the effects of multiple stressors (suspended sediment, nutrients and organic toxicants, etc.);
CATS (Contaminants in Aquatic and Terrestrial Ecosystems)	An integrated ecosystem modelling system developed by the National Institute of Public Health and Environmental Protection (RIVM), Bilthoven, the Netherlands to simulate bioaccumulation and the combined effects of nutrients and toxicants;
CASM (Comprehensive Aquatic Systems Model)	A modelling system that uses bioenergetics to simulate population dynamics of multiple aquatic organism;
SIAM (System Impact Assessment Model)	A model developed by the US Geological Survey (USGS) and consisting of a suite of tools, among which SALMOD (Simulation by Means of an Analytical Lake Model) is an ecological model developed to simulate lake phytoplankton and zooplankton;
ECOWIN	A model that provides an object-oriented approach to modelling aquatic ecological systems;
PhytoBasinRisk	A water quality model that simulates the risk of critical phytoplankton biomass and composition in large river basins. The model is free of charge and is based on an open software license concept.

highest level of realism, and SALMOD in SIAM were considered to have a medium level of realism. Ecological modelling tools include water quality simulation components that can be linked to water quality models to evaluate the effects of water quality change on the habitat suitability of an aquatic community. Ecological models have been successfully used in a number of case studies. However, in general, simulating the transport and fate of toxic chemicals in a biotic system is more challenging than doing so in an abiotic environment. The development of ecological models remains firmly in the realm of research, mostly due to the time intensity of data collection required for calibration. There is also a considerable amount of work to be done in observing, capturing and simulating processes and dynamics in ecosystems. Ecological modelling will constitute a main topic in future research and efforts to strengthen agricultural water quality management.

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PART III: RESPONSES


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CHAPTER 10. POLICY RESPONSES

Javier Mateo-Sagasta and Hugh Turral

Water pollution from agriculture is complex and multidimensional, and managing it effectively requires a range of responses. Such responses need to act on the key drivers of agricultural expansion and intensification, such as unsustainable dietary shifts. They also need to limit the export of pollutants from farms, protect water bodies from agricultural pollution loads and help restore affected water ecosystems. Influencing both farm- and landscape-scale practices may require regulation, the use of economic instruments, education and awareness-raising, cooperative agreements, and research and innovation.

Recent analyses suggest that a combination of approaches (regulations, economic incentives and information) works better than regulations alone (OECD, 2012; OECD, 2017). This chapter focuses on a broad set of policy solutions, which can provide the enabling environment for the adoption of effective on-farm and off-farm practices and technologies (discussed in Chapter 11) and thus prevent and mitigate pollution in practice.

10.1 Prevention vs remediation

The most effective way to mitigate pressure on aquatic ecosystems, and on rural ecologies more generally, is to limit the export of pollutants at the source, or intercept them before they reach vulnerable ecosystems. Once in the system, the costs of remediation progressively increase (Hardisty and Özdemiroglu, 2005). A recent assessment of the environmental performance of agriculture in OECD countries concluded that the economic costs of

treating drinking water to remove nutrients and pesticides are already substantial. In the United Kingdom, for example, the cost of water pollution from agriculture amounted to €345 million in 2003/04. The eutrophication of marine waters also imposes high economic costs on commercial fisheries in some other countries (e.g. Korea, the United States of America) (OECD, 2008).

Broadly speaking, the contamination of groundwater is much harder to remediate than surface water and is consequently more expensive. The remediation of contaminated groundwater is a long-term undertaking (Rivett *et al.*, 2002; Rivett *et al.*, 2008) and may, in some cases, not even be feasible. Similarly, coastal hypoxia leads to serious and worsening social, economic and ecological costs, as has been experienced by some OECD countries. It may require 10-30 years to return hypoxic zones to acceptable conditions, although improvements usually manifest after the first few years of reclamation efforts (Kemp *et al.*, 2009).

Since remediation is expensive and not always effective, it is preferable to start by acting on pollution drivers (e.g. diets) and to manage and minimize the emission of pollutants at source with sustainable agricultural practices. Water quality modelling (see Chapter 9) can play a key role in identifying and quantifying the sources of diffuse pollution and understanding their dynamic behaviour to be able to anticipate the expected impacts and act in advance to prevent them.

10.2 Acting on drivers: sustainable diets and reduced food waste

Different diets have different environmental footprints. An increase in demand for food with large environmental footprints, such as meat from industrial farms, is contributing to unsustainable agricultural intensification and water quality degradation. However, this can be changed. The right policies and incentives can encourage people to adopt diets that are more sustainable and healthy and thereby moderate increases in the demand for food with a large footprint. For example, financial incentives, such as taxes and subsidies on food and coupons for consumers, have been shown to positively influence dietary behaviour (Purnell *et al.*, 2014). However, with the possible exception of organic labelling (see Box 10.1), there is little evidence that environmental food labelling plays a major role in the food choices of consumers. The approach would need to be combined with broader environmental awareness campaigns to turn an increasing concern among consumers about sustainability into a change in food purchasing habits (Grunert *et al.*, 2014; UNEP, 2005).

Another key issue relates to food supply and how food systems will respond to the projected growth in food demand. Food losses and waste should be reduced as much as possible to bring food production closer to actual demand and to minimize the waste of resources and

BOX 10.1 Organic labeling

Organic produce accounts for about 15% of market value (less in terms of product volume) in OECD countries, but it is rising in importance, as wealthier consumers make more informed choices about the way their food is produced. Organic labelling has benefitted from consumer demand in the USA and Europe and has been supported by clearly defined standards, a strong certification system and a system of enforcement (OECD, 2003).

In other countries, such as China, there has been a sudden rise in consumer interest in organic produce. The volume of 'organic produce' quadrupled (from an initially low level) between 2003 and 2005, with a change from export to local focus. Since then, there have been a number of campaigns to improve consumer safety with regard to pesticide residues on fruit and vegetables. The campaigns were initiated by local and international NGOs, but have been taken up more broadly with programmes on the internet and TV. Three 'environmental' labels are now used in food certification: organic, green and pesticide-free. The policing of organic certification is growing tougher. According to one recent China Daily report, about ten percent of the organic food sampled in Beijing was counterfeit (Yang *et al.*, 2007).

associated environmental impacts. About one-quarter of produced food is lost along the food supply chain. Producing this lost and wasted food accounts for 24 percent of the freshwater resources used in food crop production, 23 percent of total global cropland area and 23 percent of total global fertilizer use (Kummu *et al.*, 2012). Nitrogen pollution has a major impact on water quality. Grizzetti *et al.* (2013) calculated that the nitrogen pollution associated with global food waste was 6.3 teragrams per year, and that, in the European Union, 12 percent of water pollution from using nitrogen in agriculture is linked to food waste. FAO has extensively reviewed options for reducing food loss and waste (e.g. FAO, 2013a; FAO, 2015).

10.3 Regulatory instruments

Typical regulatory instruments include water quality standards; pollution discharge permits; mandatory best environmental practices; restrictions on agricultural practices or the location of farms; and limits on the marketing and sale of dangerous products. Some agricultural activities may be restricted without an environmental impact assessment or specific protective measures, such as the creation of buffer zones adjacent to water courses. Many regulatory approaches require inspection or self-reporting to ensure compliance, with violations subject to penalties such as fines and compensation payments. Enforcement remains a challenge, however.

Well-known principles for reducing pollution, such as 'polluter pays,' are hard to apply to non-point agricultural pollution because identifying the actual polluters is neither easy nor cheap (OECD, 2017). Assessing compliance and the effectiveness of regulations, (e.g. the adoption of best practices to manage diffuse pollution) is also difficult as it requires multiple steps, such as nutrient management plans; bookkeeping for fertilizers, pesticide and manure management on farms; nutrient accounting; and soil analysis. Therefore, regulations alone are typically not cost-effective for diffuse sources, although they have worked reasonably well with wastewater treatment plants, industry and intensive livestock units (UN-Water, 2015).

Regulations to protect water quality need to be enforceable. Water quality targets also need to be realistic and time-bound, and they need to balance the costs of adopting a solution and the benefits resulting from higher water quality. In addition, water quality targets need to take into account time lags between the introduction of a given practice and measurable outcomes (this is particularly relevant for the restoration of aquifer water quality). Once a target is set, planners need to find the most cost-effective combination of policy instruments (UNU-EHS/UNEP, 2016; OECD, 2017). As noted above, pollution prevention will typically be cheaper than the restoration of affected aquatic ecosystems.

BOX 10.2 | Regulations to control point source effluents from intensive livestock in USA

Pollution from factories and sewage treatment plants has been dramatically reduced in the United States of America over the past 40 years, but runoff from agricultural activities, including animal feeding operations (AFOs), continues to degrade the environment and puts drinking water at risk. To address this, and after intense debate, the US Environmental Protection Agency (US EPA) issued in 2003 the *national pollutant discharge elimination system permit regulation and effluent limitations guidelines and standards for concentrated animal feeding operations* (US EPA, 2003). As per these rules, a farm that meets certain size criteria and/or has the capacity to pollute is defined as a Confined Animal Feeding Operation (CAFO) and is subject to legislation associated with point source pollution, namely the National Pollutant Discharge Elimination System (NPDES) permits under the authority of the Clean Water Act. CAFOs must have certified animal waste management plans, including a nutrient management plan; a waste utilization plan that includes a 30 metre quarantine zone between surface waters and manured areas; and a standardized recordkeeping and reporting system. While this regulation will assist in reducing the impairment of United States of America waters, the actual effectives of such regulations are still debated and have not been well assessed (Burkholder *et al.*, 2007).

BOX 10.3 | Pesticides are needed as is their regulation: examples from France and India

France, the major user of pesticides in the European Union(EU), enacted the Loi Grenelle in 2009 with the intention of making significant reductions in the use of pesticides of all types, by implementing a range of activities. One target is to increase the certified organic area of the country from 2% to 20% by 2020. A secondary thrust is to certify 50% of farms as "nature-friendly" through compliance with set standards and norms. A third component is the Ecophyto programme, which has 8 gears: 1) Assessing progress with pesticide use reduction; 2) Identifying and prioritizing agricultural systems for pesticide use reduction; 3) Encouraging innovation in design development of low pesticide input practices and cropping systems; 4) Better training in safe use; 5) Better surveillance and monitoring; 6) Meeting pesticide residue requirements in foreign markets; 7) Reduction in use of pesticides in non-agricultural settings (gardens); and 8) Overseeing the plan at national and regional levels and managing stakeholder involvement and consultation. This program is expected to withdraw 40 pesticides, targeting a 50% reduction in pesticide use for plant production by 2018 (Crosskey, 2016).

Many developing countries are lagging behind in the design and implementation of effective pesticides regulations. Some have old statutes on the books relating to pesticides and many provisions are honored in the breach. Nevertheless, some countries are now seeking to update legislation and to find better means of ensuring implementation. For example, the Government of India has drafted the Pesticides Management Bill (GoI, 2017), which will replace the Insecticides Act, 1968, providing a more effective regulatory framework for the country. The new act will regulate the import, manufacture, export, storage, sale, transport, distribution, quality and use of pesticides. It also codifies harsher punishments for manufacturers of spurious pesticides in order to prevent risk to human beings, animals or the environment.

Increasingly policy-makers are interested in regulating pollution outputs, rather than the use of farm inputs. This requires reaching a consensus on the maximum tolerable concentration of a given pollutant in a waterbody so that, with models, maximum pollution loads (caps) can be calculated. Subsequently, pollution caps can be allocated to individual landowners. Land managers can use innovative farm practices that minimize pollution without necessarily restricting the inputs they use. However, the allocation of caps to farmlands in a cost-effective and equitable manner remains challenging (OECD, 2017). Additionally, there are some limitations on the use of models related to the uncertainly of data or model components, and these require continuous efforts on data collection and model accuracy (see Chapter 9).

Nevertheless, the implementation of pollution caps is an emerging reality. On the east coast of the United States of America, a total maximum daily load (TMDL) programme is used to reduce nitrogen, phosphorus and sediment loading to the Chesapeake Bay (Batiuk *et al.*, 2013). Korea is also adopting a TMDL management system, which aims to control both point and diffuse pollution with a permitting system and the support of water quality models (Kim *et al.*, 2016, NIER, 2014).

10.4 Economic instruments

Economic instruments are increasingly employed to improve or replace simple legal provisions or regulations. They include taxes, 'set-asides' (the conversion of agricultural land to natural uses) and payments to limit production or the intensity of land use.

Taxes include polluter payments, dedicated environmental taxes and taxes on technologies, products and inputs that have adverse ecological consequences (e.g. pesticides), according to the level of hazard.

Incentives encompass tax breaks for the adoption of practices that minimize farm export of nutrients and pesticides; revolving funds for upgrades to water treatment plants such as the US EPA Clean Water State Revolving Fund with $5 billion on account; and reverse auctions – for example, the sale of irrigation water to a private or state buyer for environmental use. Some European countries make substantial payments to farmers for 'landscape maintenance', and the Conservation Reserve Program in the United States of America pays farmers to take land out of production for specified periods.

Agri-environmental payments (AEPs) have been widely used to encourage farmers to adopt more ecologically-friendly practices. In the postwar era, subsidies were provided to farmers in Europe and North America to improve the quantity and quality of food, at ever-cheaper prices to the consumer. This resulted in overproduction and in no small measure contributed to the high use of fertilizer and pesticide in increasingly intensive agriculture. Support payments under the Common Agricultural Policy (CAP) were designed to protect small traditional farmers from the economic 'efficiency' of larger, more industrial producers. With continued overproduction, the burden of support payments, a better understanding of the externalities of intensive agriculture, and the limited success of production quotas, the CAP eventually morphed support payments, first into set-asides and later into payments for specific environmental outcomes on-farm.

More complex economic instruments are emerging. One that took its lead from carbon trading (climate change mitigation) is nutrient credit trading (Corcoran *et al.*, 2010). The

opportunity for nutrient trading exists because of substantial price variations between markets for different nutrients, although it is not clear that the environmental cost of the nutrients actually varies from place to place. If a farmer removes more nitrate or phosphate loading from a watershed than is required by law, these credits can be traded. Since it is difficult to monitor the actual export of nutrients, farm credits require proxies such as changed fertilizer rates, production practices and crop patterns or the retirement of land from cultivation. Water quality trading initiatives have started in Australia, Canada, New Zealand and the United States of America. Water quality trading in Australia is not a market activity – the Salt Credit Scheme (1994) is designed to limit the total salt contribution to the Murray River from each riparian state. Each state has, in effect, a quota and in order to manage rising salinity in one area, it must mitigate the salinity in another part of its territory. This has provided a flexible framework for investment to prioritize and manage salinity across each state and across the basin. In the long term, the salt credit available (measured as the median concentration at Morgan, in South Australia) to each state is intended to decline.

10.5 Education and awareness

Policies to change farmer behaviour and incentivize the adoption of good practices are critical to preventing pollution at the source. Such policies need to include (free) advisory services and training for farmers. Demonstrating the economic benefits of adopting good practices has also been shown to be effective. Benchmarking can promote behavioural change among farmers by showing them how they perform as compared with their peers (without identifying the best and worst performers). Benchmarking can be applied to the application of fertilizers, manure and slurries, and pesticides. A subtler form of persuasion is the incorporation of environmental modules into school curricula and motivating students to raise environmental issues in their communities.

Information can be provided directly through training and extension, radio and TV broadcasts and voluntary codes of practice. Farmer awareness of high water tables and incipient salinity has been raised through a community programme in Australia, known as Water Table Watch, which involves schools in monitoring water levels in their community. Similar initiatives have been undertaken to monitor flora, fauna (birds) and habitat.

10.6 Cooperative agreements

There is increasing interest in cooperative and voluntary agreements – typically between farmers, water suppliers and authorities – as a means to implement better environmental practices in agriculture. In some cases, private water suppliers have signed agreements with farmers to limit practices (e.g. nitrogen use) that may compromise water quality (and

therefore their products), with the costs paid by the water supplier and ultimately borne by consumers (FAO, 2013b). In other cases, specific areas in river catchments may have been identified as major contributors of sediment (and sediment-borne pollutants) to important ecosystems. To address this, cooperative agreements can be developed between landowners and relevant authorities to reduce erosion, potentially incentivized by policies in favour of agro-environmental payments.

One of the best-known examples is the agreement between Vittel, a well-known producer of bottled natural spring water in the Vosges Mountains in France, and local farmers and pastoralists (FAO, 2013b). Vittel has signed agreements to limit nitrogen use (to zero in some cases) and other farm management practices that may compromise the quality of their product. Recently, specific areas of river catchments feeding into the Great Barrier Reef in Queensland, Australia have been identified as major contributors of sediment and sediment-borne pollutants. Cooperative agreements have been developed between land owners and the state to reduce erosion by a number of means requiring investment and payments (Queensland government, 2018).

10.7 Corporate social responsibility and GAPs

One of the most significant trends in the private sector is the rapid growth in activities related to corporate social responsibility (CSR). Although there is not a standard and commonly accepted definition of CSR, the term typically refers to actions taken by corporations, beyond their legal duties, in support of their employees, broader communities and the environment. Although debates are still ongoing as to whether a good CSR performance contributes to a firm's success, social benefits and environmental improvement (Hatanaka, 2005; Kong, 2012), the reputational and economic risks for companies with deficient social responsibility are unquestionable.

In the food industry, CSR approaches are increasingly shifting from the single firm level to supply chains and networks. Accordingly, agricultural producers are being required by their buyers to provide documentation about their production practices to ensure that good agricultural practices (GAPs) are use. Producers who are unable to provide these assurances to their buyers may find that they will have less opportunity to sell their products. The adoption of GAPs may be important for downstream firms seeking to project the image of a good corporate citizen. This becomes an economic incentive if a good public image encourages buyer loyalty or shareholder investment (FAO, 2003).

While GAPs can be seen as attempts to improve the sustainability of agriculture and can bring reputational benefits to the different companies along the value chain,

concerns have also been raised regarding their potential effect on smallholders in developing countries (FAO, 2003). It is critical that the adoption of too stringent GAPs do not marginalize small producers, by cutting off access to export markets or imposing disproportionately higher production costs on the given the investments that may be needed to adopt good practices.

10.8 Broader policy frameworks

Policies addressing water pollution in agriculture should be part of an overarching water policy framework at the national or river-basin scale, with all pollutants and polluters considered together.

International conventions and declarations play a role in raising awareness and political profile. For example, the Nanjing Declaration on Nitrogen Management was signed in China on October 16, 2004 (Nanjing Declaration, 2004). The declaration, while acknowledging the vital role that nitrogen plays in the production of food and fibre, commits its signatories to optimizing nitrogen management in food and energy production. The declaration was motivated by the increasing recognition of non-point source nutrient export from farms, which is already a serious concern in many regions around the world (Clothier, 2008). This international commitment needs to translate into specific activities in individual countries.

Nevertheless, few countries have national policies and standards to control water pollution from agriculture. There are notorious exceptions, however. For example, both Australia and Sweden have had water quality strategies that consider non-point source pollution for more than 15 years. Broader water quality frameworks, such as the Nitrates Directive (Council of European Communities, 1991[8]) and the Water Framework Directive (European Parliament and Council of the European Union, 2000) in the European Union (discussed in more detailed in Box 10.4), and the Clean Water Act in the United States of America (US EPA, 2017) combine point and diffuse pollutant standards for industrial and agricultural compounds.

National policies need to be coherent. Interventions aimed at increasing food production and farm income on the one hand and at mitigating pollution on the other should be mutually supportive – or at least not in conflict, although this may be hard (politically) to achieve in practice. For example, the subsidies that are often in place for

8 Amended by the European Commission in 2003 and 2008.

agrochemicals do not act as an incentive for efficient use, and they encourage farming on more fragile lands. Effective inter-ministerial cooperation mechanisms are required to increase policy coherence.

BOX 10.4 | Selected policy frameworks for water pollution control in Europe

The European environmental policy is based on three main principles: the precautionary principle; the principal of preventive action; and the polluter pays principle. The actions that should be taken to tackle environmental problems are based on five pillars:

- enhanced implementation of the existing environmental policy;
- integration of environmental concerns in all other policy areas;
- close cooperation with trade, industry and consumers;
- enhancement of the quality and accessibility of environmental information to the public; and
- development of a more environmentally-minded attitude towards spatial planning.

(European Commission, 2002)

Two overarching water quality policy instruments set requirements on ecological health for member countries of the European Union: the Nitrates Directive and the Water Framework Directive. The directives require individual countries to establish policies and supporting actions in line with their legislative and governance frameworks.

The objective of the Nitrates Directive (Council of European Communities, 1991) is to reduce water pollution caused or induced by nitrates from agricultural sources in order to protect human health, living resources and aquatic ecosystems. The directive includes rules for using animal manure and mineral fertilizers. The core of the directive is that a balance should be reached between N supply to soils (including mineral and organic fertilizers) and the nutrient demands of the crop being grown. Member states are required to guarantee that the annual farm application of N, as animal manure, does not exceed 170 kg per hectare. This is equivalent to a stocking rate of about one dairy cow per hectare. In European regions with relatively intensive dairy farming, stocking rates are often much higher and reducing them is a significant challenge.

The implementation of the Nitrate Directive proceeds in five steps:

1. designate 'nitrate vulnerable zones' (NVZ): agricultural land that makes a significant contribution to nitrate pollution in a susceptible area;

2. develop codes of good agricultural practice for farmers. These are voluntary at the national level and compulsory in NVZs;
3. develop action programmes for NVZs;
4. reduce nitrate leaching, monitor programme effectiveness; and
5. undertake national management of nitrate concentrations and eutrophication.

NVZs cover about 47% of the total EU area (European Commission, 2013), largely due to the importance of groundwater in the drinking water supply, with a legal upper limit 50mg/l of nitrate:

The action programmes specify:

- periods when the land application of certain types of fertilizers is prohibited;
- the capacity of storage vessels for livestock manure; and
- limits to the quantity, timing and mode of fertilizer application, consistent with good agricultural practice and the characteristics of the vulnerable zone.

The **Water Framework Directive** came into effect in 2000 and set a goal for all the EU member states to protect all waters and have them in a good condition by 2015 (European Parliament and Council of the European Union, 2000). Three phases were agreed, with preparatory work lasting until 2002, followed by testing of river basin management guidelines in pilot basins between 2002 and 2004 and the finalization of the guidelines and an outline action programme by the end of 2005. The Water Framework Directive has been implemented in steps, such that it was first incorporated into each member state's national law in 2003 with the identification of river basins and their management bodies. By 2006, each member state was required to have an operational system in place for monitoring the ecological and (chemical) water quality status of surface waters. River basin management plans had to be developed by 2009, which specified measures to control point source discharges and non-point pollution; to prevent or limit leakage from point sources (e.g. feedlots, dairies, processing plants); and to promote sustainable and efficient water use.

The river basin plans were required to classify all subcatchments, and define water quality status. Measures to address diffuse pollution in each basin had to be in place by the end of 2012 and ecological health targets had to be achieved (and verified) by 2015.

10.9 Research and data

There are many knowledge gaps around water pollution caused by agriculture. For example, the contribution of crops, livestock and aquaculture to water pollution are frequently not well assessed, particularly in developing countries. Box 10. 5 illustrates – with an example from the Ganges Basin – what is a common reality in many other low and middle-income countries.

Quantifying the relative contribution of agriculture to water quality problems is essential if national governments are to develop meaningful and cost-effective responses. The polluter pays principle cannot be applied if the source of the pollution is unclear. A sustained research and modelling effort, supported by water quality monitoring, is needed to better understand pollutant pathways and the links between the causes and effects of pollution.

The pathways of, and the health and environmental risks posed by, emerging agricultural pollutants, such as animal hormones, antibiotics and other pharmaceuticals, are growing areas of research that require more attention. For example, greater understanding is needed on the contributions of animal medicines to the increasing problem of antimicrobial resistance among pathogens.

There are opportunities for greater innovation in practices and technologies to diminish the use of nutrients and pesticides on farms and reduce the movement of pollutants from farms to sensitive aquatic ecosystems. Research is needed to evaluate policies and instruments for reducing source loads and minimizing pollution along flow paths to the sea. More work is also required to quantify the effectiveness of different approaches to reducing the economic impacts of water pollution on agriculture.

There is scattered evidence of the costs associated with diffuse pollution of water in general and agricultural pollution in particular. While existing studies suggest that the global costs of water pollution from agriculture could exceed billions of US dollars (OECD 2017), there is a need for a more systematic assessment of such costs as a key tool for awareness raising and influencing political will.

Research results need to be used and applied if they are to be effective in reducing pollution in agriculture in practice. It is crucial to establish information systems for transferring new knowledge and technologies to support farmers, water managers and policy-makers. Research projects need to consider, from the conceptual stage, the

BOX 10.5 **Key knowledge gaps around water quality in the Ganges basin (Mateo-Sagasta and Tare, 2016)**

Despite efforts to clean the Ganges River, the main stream still directly receives at least 2.7 billion litres of sewage from medium and large cities every day, of which at least 74% is untreated. Industrial effluents are in the range of 10-20% of the total volume of wastewater directly reaching the Ganga. Although this is a relatively low proportion, it is a cause for major concern because the effluents are often toxic and non-biodegradable.

The Ganga is also impacted by non-point source pollution, but the actual contribution of agriculture, livestock and aquaculture to water quality degradation is not known. Trends in agrochemical use as well as the density of livestock suggest that these pressures could be important in the river basin. To understand the extent of the problem, a sustained research and modelling effort would be needed to track the pathways and loads of nutrients and organic matter from their sources to water bodies. Similarly, the contribution of other non-point-sources of pollution, such as faecal sludge or open defecation, to the degradation of the Ganga is not well understood and will need further research.

The hydrological links between groundwater and surface water in the Ganga basin have not been properly assessed and modelled, therefore it is not possible to estimate the contribution that groundwater pollution may have made to the Ganga and tributaries, and vice versa. Understanding this is particularly important in the case of pollutants such as nitrate, pesticides and salinity.

A comprehensive water quality model at the basin scale, which allows researchers to simulate solutions, will be critical for planning and assessment. Rejuvenating the Ganga will require a massive investment. From the government perspective, it will be crucial to select the most cost-effective combination of solutions to meeting water quality standards and improving river health. These solutions need to include reducing pollution from different sources, restoring appropriate water flows and, ideally, a combination of both. Understanding how these solutions might translate into reduced pollution loads, enhanced water flows and, consequently, improved water quality along the river will require complex water quality modelling, an exercise that has not been done comprehensively in the Ganga basin.

Any water quality assessment and modelling effort will require good quality data. The current water quality monitoring network along the Ganga and its tributaries is very poor and will need to be strengthened with substantially more stations, which will need to monitor more parameters and with a greater frequency. ➤

➤ Finally, a better understanding of how pollution translates into health and environmental impacts, and the costs of such impacts, will help to raise awareness on the size of the problem and will help justify the massive investments that the river needs if it is to be restored.

specific needs of users and engage them in the process, from knowledge generation to environmental and health outcomes.

Research cannot be conducted without data. We need better data to understand the process by which specific waterbodies become polluted and the pressure that this puts on aquatic systems. Because many indicators are subject to temporal and spatial variability, adequate monitoring programmes with appropriate sampling rates and density are key (but expensive) priorities for improvement.

Monitoring data help to determine the state or condition of a waterbody and to quantify the amount of polluting material that is reaching aquatic systems. Data is also needed to understand long-term trends in the state of global water bodies and to better understand the pressures and drivers behind them.

Impacts can be measured directly, but require modelling to predict future behaviour and severity. Modelling ecological impacts often demands intensive calibration and data. Research is needed to evaluate which policies and instruments will work best to reduce source load and minimize pollution along the flow path to the sea. Similarly, work is required on the cost-effectiveness of different technological and economic solutions.

Load and concentration data need to be gathered at key points in the landscape, and this can be done at places where flows are already measured for other purposes: e.g. for flood warning and control, irrigation diversion, etc. Monitoring and characterization does not have to be costly. For example indicators of soil health and nutrient use efficiency can be collected by farmers, and biodiversity can be surveyed on a long-term basis as part of school science activities. Data aggregation and analysis can be facilitated by GIS, which can also assist in the development of cost-effective sampling strategies.

It is relatively straightforward to measure concentrations and loads at the point of discharge from a wastewater treatment plant or feedlot that flows directly into surface

water. It is more difficult to measure the net flux to groundwater below fields that receive heavy applications of manure and slurry. It is even harder (and costlier) to measure the surface and subsurface fluxes from individual fields and farms, although it is possible to monitor when and how much agricultural chemicals are applied.

Typically, monitoring requires sampling representative conditions that differ in time and space. For example, the pesticide contents in a lake should be sampled at a range of depths and locations that enable a good estimate of the average condition of the whole lake. They should be sampled frequently enough so that major changes are not missed. Sediment (and thus phosphate) loads will be highest during storm events that may last one or two days. Gauging stations normally record sample flows at fixed time intervals, perhaps once or twice per day. If recordings are done manually, dangerous weather conditions could make it difficult to collect any data at all.

Both concentration and load provide important information: when concentrations in any flow reach a certain level, they may be directly toxic to some organisms (e.g. pesticides) or they may trigger conditions that commence a harmful algal bloom (e.g. nutrients, dissolved oxygen). In general, the impacts of concentration are of greater concern in low flows. Although the concentration of pollutants in solution are often lower at high flows, sediment-borne concentrations may be greater. The average condition of receiving waters depends more on the load received over the course of time. Load is determined by flow rate and concentration, integrated over time. Thus, both adequate sampling frequency and combined measurement of flow and concentration to determine load are very important. Four types of sediment monitoring are being conducted under the EU's Water Framework Directive: risk assessment, trend monitoring, spatial monitoring and compliance monitoring, with a focus on the type and level of industrial contaminant transported by sediment.

Although watershed boundaries can be clearly determined from topographic maps or by using sophisticated remote sensing data to create digital elevation models, the delineation of groundwater zones and their connectivity may require intensive hydrogeological sampling. Determining the connectivity between surface and groundwater often requires another level of investigation, and is mostly confined to research at the moment.

Ecological monitoring is an emerging science and, as a result, it is rare to find a strong historical data set that allows a clear depiction of trends in ecological health. An interesting approach has been developed in Victoria, Australia, to rapidly survey the

'ecological assets' in a river reach to define their health (using condition scoring) and then prioritize where the best returns to conservation and remediation are likely to be (DPI Victoria, 2006). The inventory of ecological assets provides a framework for further routine monitoring.

BOX 10.6 | Monitoring using remote sensing

Successful techniques in remote sensing analysis tend to find rapid application and, when costs are prohibitive, there is often quick adaptation of the techniques to other more affordable sensors. This has been the case with MERIS (Medium Resolution Imaging Spectrometer), one of the main instruments used on board the European Space Agency (ESA)'s Envisat platform, which gathers data on large inland and coastal waterbodies. The application of remote sensing techniques to smaller water bodies, wetlands and rivers remains expensive and is likely to be done on a research, or one-off diagnostic basis, although as the pace of sensor development and the associated analysis remains high, it is likely that there will be continued and widespread application to environmental monitoring, including water quality issues.

At present, the focus of the effort around water quality lies in monitoring the extent and dynamics of harmful impacts, notably harmful algal blooms in freshwaters, coastal zones and in the open ocean. The indicators of inland and coastal eutrophication include:

- chlorophyll-A content (Chl-a), which is a measure of phytoplankton concentration;
- phycocyanin (PC), which is an indicator of cyano-bacterial concentration; and
- sediment concentration (TSS) in surface layers.

Chloropyll-A provides a good measure of phytoplankton growth, and can be correlated to the chemical and biological oxygen demands of organic pollutants (CEARAC, 2007). It is a proxy for eutrophication, but high levels of phytoplankton growth do not necessarily indicate eutrophic conditions. The emergence of harmful cyanobacterial algae is a better indication of eutrophication, but at present anoxia cannot be detected. The hazards of toxic cyanobacterial blooms call for frequent and rapid monitoring of waterbodies. Suspended solids can be estimated from turbidity. Estimates of both can be retrieved from water colour. In practice, the estimates of Chl-A and turbidity can confound each other, and other colourations, such as yellow pigmentation from dissolved organic matter (CDOM) can introduce further variability n accuracy. Analysis is based on three categories: inland waters; open ocean waters and coastal waters.

At the farm level, better methods are needed for assessing nutrient and pesticide needs, as are techniques for managing fertilizer applications to minimize accumulation and export. This ranges from soil and plant testing, which are relatively inexpensive, to soil zoning (GIS and precision farming).

Better understanding of the chemistry of organisms and soils may lead to better targeted, more discriminating, shorter-lived and species-specific pesticides. An improved understanding of the same fundamentals can help us to understand and prevent the loss of key ecosystem components, which undermines the health of the trophic chain and hence the whole ecosystem.

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CHAPTER 11. ON-FARM AND OFF-FARM RESPONSES

Javier Mateo-Sagasta

With contributions from Joost Albers

The adoption of best agricultural practices and technologies in the field is essential to preventing pollution emissions from farms. Nevertheless, because agricultural pollution depends on many factors, some of which are out of the farmer's control (such as heavy rains that favor erosion or runoff of pollutants), some degree of emissions from farms may be impossible to avoid. In these cases, solutions such as vegetated buffer zones around farms and waterbodies, as well as other interventions along the landscape, can complement on-farm practices for water pollution control.

Extensive literature exists on agricultural practices that can be used to control water pollution from agriculture at the farm level (e.g. FAO, 1996; US EPA, 2003; EC, 2003; FAO, 2013; OECD, 2016). The aim of this chapter is to summarize such practices for crop, livestock and aquaculture farms. Adopting good agricultural practices provides broad benefits to society but imposes private costs on famers, therefore, farmers will need proper incentives and capacities (see Chapter 10). The chapter also analyses how different agricultural sectors can be better integrated within agrosystems so that the waste from one sector can become a resource for another. Finally, describes off-farm measures and broader interventions at the landscape level that can complement on-farm practices, and minimize the release of pollution into waterbodies.

11.1 Good practices for crop farms

In crop production, management measures for reducing the risk of water pollution due to organic and inorganic fertilizers and pesticides include optimizing the type, amount and timing of their applications to crops. Establishing protection zones along surface watercourses within farms and buffer zones around farms has often been shown to be effective in reducing pollution migration to waterbodies (Dorioz *et al.*, 2006, Zhang *et al.*, 2010). The storage and disposal of pesticide waste and empty containers need to follow safety guidelines (e.g. Geng and Ongley, 2013). In addition, efficient irrigation schemes will reduce water return flows and can greatly reduce the migration of fertilizers and pesticides to waterbodies (Abrahao *et al.*, 2011). Contour ploughing, no or minimal tillage and restrictions on the cultivation of steeply sloping soils are measures for reducing soil erosion (US EPA, 2003). This section summarizes some of these best practices.

11.1.1 Nutrient management

Farmers need to maintain soil fertility and replace the nutrients removed at harvest. At the same time, they minimize nutrient surpluses, which can harm the environment. To this end, farmers should consider some key principles that underpin good nutrient management (US EPA, 2003; FAO, 2006a; Schoumans *et al.*, 2011; Liu *et al.*, 2013), such as:

- Manage soil and nutrients together. Only after farmers have made improvements in the biological, physical and hydrological properties of their soils, can they expect to get the full benefit from supplying additional plant nutrients to their crops.
- Seek yield improvements by identifying and overcoming the most limiting factors (and the limiting nutrients in particular) in order of their diminishing influence on yield. This will help minimize the overuse of agrochemicals that are not actually needed to maximize yields.
- Replenish soil nutrients removed with harvested products through an integrated plant nutrition management approach (FAO, 2006a). Such an approach should take advantage of all possible on- and off-farm sources of plant nutrients, including organic manures, crop residues, rhizobial N-fixation, root mycorrizhal fungi infestation for improved nutrient uptake, transfer of nutrients released by weathering in the deeper soil layers to the surface by tree roots and leaf litter, nitrate and phosphate content of irrigation water, etc. Using these nutrient sources will minimize the need for mineral fertilizers.
- Split fertilizer applications across the most responsive growth stages of a particular crop. Applying split applications of fertilizer N can potentially reduce N leaching

regardless of the watering method used (Nakamura *et al.*, 2004), as can the application of less soluble forms of N or slow-release N fertilizers (Paramasivam *et al.*, 2001).

- Place nutrients beneath and on either side of the plants, at a shallow depth, where there is the highest concentration of roots. A costlier way to limit leaching to groundwater is to install under-field drainage tanks and collect and recycle drainage flows. A more cost-effective alternative is to improve irrigation management to ensure high levels of distribution uniformity and minimize deep percolation below the root zone.
- Apply fertilizers to vegetables frequently and in small amounts. Use soluble fertilizers mixed into the irrigation water, and applied with some precision (e.g. with microirrigation systems). Farmers in Sunraysia, Australia have found that they achieve the highest fertilizer efficiency through fertigation, by applying nitrogen over 10-15-minutes, 25 minutes before the end of the watering period (FAO, 2011).
- Use slow-release fertilizers. Coated fertilizer is used for controlling fertilizer N release to fit requirements for nitrogen at different points in the cropping season. Release rates in soils are determined by soil moisture content, pH and soil temperature, and the particle size of the fertilizer. Studies have shown that slow-release fertilizers have lower leaching and fewer volatile losses of nitrogen (Ni *et al.*, 2011; Azeem *et al.*, 2014). However, slow-release fertilizers are more expensive than the most common types of fertilizers.
- Use green manure, i.e. by leaving uprooted or sown crop parts to wither on a field so that they serve as a mulch and soil conditioner to help improve both soil organic matter and N & P status. This – and other practices preventing the use of mineral fertilizer – are used in organic farming, which has become an increasingly important niche in food production in the OECD and, more recently, in China (OECD, 2003; FAO, 2013). Although certified organic production makes no use of chemical fertilizers or pesticides, the effectiveness of organic production methods in controlling water pollution are more ambiguous (see Box 11.1).

The inefficient use of agrochemicals imposes a direct cost, not only on the environment, but also on the farmer through lost production and the waste of purchase cost. This needs to be effectively communicated to farmers through adequate extension programmes and other awareness strategies (see Chapter 10).

BOX 11.1 Does organic farming reduce water pollution by nutrients?

Organic farming is often promoted as a more sustainable alternative to conventional farming. Nevertheless, a recent meta-analysis conducted in Europe concluded that, while organic farming practices generally have positive impacts on the environment per unit of area, this is not necessarily the case per product unit (Tuomisto *et al.*, 2012).

Organic farms tend to have higher soil organic matter content and lower nutrient losses (including nitrogen leaching) per unit of field area. However, nutrient loses per product unit were higher in organic systems (Tuomisto *et al.*, 2012). Additionally, yields in organic farming are typically smaller than in conventional farms, although, these yield differences are highly contextual, depending on system and site characteristics (Seufert *et al.*, 2012).

As in other types of farming, the yields and impacts of organic farming depend largely on the farm management. With the right practices, organic farmers can minimize pollution and maintain soil fertility and productivity in the long term. Olson-Rutz *et al.*, (2010) summarized these good practices in organic farming as follows:

Grain legumes and green manures that supply N to the soil are the foundation of organic crop rotations and should be present 25 to 50 percent of the time. Reducing tillage and increasing cropping frequency and diversity improves the soil's N supplying power and minimizes potential for soil degradation and erosion. Reintroduction of livestock grazing may be important for the economic and environmental stability of agricultural systems in our region. Manure is an excellent source of many nutrients, but may not be locally available in sufficient quantities. Practices that encourage microbes which increase soluble P should be encouraged, though inoculants should be used with caution until more data are available. Integrated use of crop rotation practices, livestock grazing, and fertilizers/ amendments have the potential to improve soil quality and increase sustainability of organic crop production.

11.1.2 Pesticide management

Chemical pest control has become an important part of agriculture, but as insects and pathogens developed resistance to chemicals, and as other beneficial predatory species were killed out through excess pesticide use, a number of serious pest epidemics encouraged a more intelligent approach, which has become known as Integrated Pest Management (IPM).

IPM encourages a rational and minimal use of chemicals for pest control. It promotes regular monitoring and identification of pest numbers and seeks to preserve healthy populations

of natural predators. IPM combines the breeding and planting of pest-resistant varieties, strategic mixtures of crop varieties with different resistance characteristics as and crop rotation and fallowing. It may also include the introduction or improvement of natural predators of common pests. IPM can be successfully implemented for many types of crops and pests in different agroclimatic conditions, from temperate Europe (FAO, 2017) to tropical West Africa (James *et al.*, 2010).

In the future, pesticides should be highly efficient, with high biological activity but more selective, less persistent in the environment and less toxic to humans and non-target species. The use of these 'pesticides of the future' (Zhang *et al.*, 2011), together with the adoption of other IPM principles, may greatly reduce pesticide use and the pollution of the environment.

Since pesticide use is likely to be higher under irrigated conditions, the importance of solubility should not be overlooked. Some recommendations for good pesticide management under irrigation in Australia are given in Box 11.2, as an example of the precautions that farmers everywhere can take.

BOX 11.2 Considerations for pesticide management under irrigation in Australia (Simpson and Ruddle, 2002)

- Do not apply pesticide immediately before irrigation or in the likelihood of heavy rain.
- Excessive irrigation can carry some pesticides (such as Atrazine) well below the root zone and outside the area of effective weed control, leading to groundwater contamination.
- Reduce soil and sediment loss in surface runoff. Significant reduction in pesticide transport from runoff can result, particularly for pesticides such as paraquat, trifluralin and chlorpyrifos, which have high adsorption on soil particles.
- The risk of significant off-site movement from the farm can be reduced by not treating large areas with pesticides at one time. This will reduce the potential source of pollution if irrigation is scheduled or heavy rain falls.
- Some herbicides, such as Atrazine, ametryn or hexazinone, are highly mobile and can move quickly off farm (either in runoff or by leaching), particularly if irrigation or rainfall occurs.
- Freshly applied pesticides are often more mobile than pesticides that have had time to bind to soil or foliage. ➤

- Irrigation tailwater can contain high levels of pesticide residues. Recycling and avoiding excessive irrigation after pesticide application can minimize off-site losses.
- Additional precautions should be taken when storm or irrigation runoff discharges near streams or sensitive habitats. Good water management is strongly linked to effective pesticide management.
- In highly porous soils or areas with shallow water tables, less mobile alternatives should be considered to minimize the potential contamination of groundwaters or baseflows in streams.

In China, as in other rice-growing countries, natural predators, especially arthropods, have been shown to effectively control major pests. IPM strategies for cotton pests, including cultural, biological, physical and chemical controls, have been developed and implemented in the Yellow River Region, the Changjiang River Region and the Northwestern Region of China over the past several decades (Luo *et al.*, 2014). Also, due to the introduction of transgenic cotton (Bt cotton) in China in 1997, which resists some pests, together with the use of mixed planting systems of cotton, corn, soybean and peanut on small farms, the use of pesticides on cotton have fallen dramatically in the past 20 years (Geng and Ongley, 2013; Luo *et al.*, 2014).

11.1.3 Water management and erosion control

The efficient and safe use of agricultural inputs is key to preventing pollution at the source. Nevertheless, farmers can also manage pollutant carriers, namely water (where pollutants can be dissolved or suspended) and sediments (where pollutants can be adsorbed) (See Chapter 3 on pollution pathways). Any improvement in irrigation management or erosion control that reduces or eliminates leaching and drainage (Abrahao *et al.*, 2011) or sediments transport (Li, 2013) off-farm will likely reduce nutrients and pesticide export.

For example, nutrient and pesticide leaching can be reduced by accounting for rainfall in irrigation scheduling (i.e. using variable scheduling rather than fixed scheduling). In fertigation schemes, optimizing irrigation scheduling is key. The farmer should consider the nutrient demand at different growing stages., and should follow the principle of ‘little but more times’. Research shows that by increasing drip irrigation frequency from

one to eight times per day, the leaching loss of NO_3-N can be reduced by 37 to 66 percent (Vazquez *et al.*, 2006).

Tile drainage has been shown to reduce losses of sediment, phosphorus, and pesticides from agricultural land in the northwestern United States of America (Blann *et al.*, 2007). Subsurface drainage shifts the volume and timing of and the pathway by which precipitation enters surface waters, affecting in-stream peak flows and stream and wetland hydrology.

Controlled drainage can regulate the amount and rate of drainage and reduce the chemical loss from the field, thereby improving the farmers' profits and improving drainage water quality (Duncan *et al.*, 2008; FAO, 2013). Controlled drainage has been used successfully in different countries and agricultural systems to enhance water productivity and to reduce pollution (Yu *et al.*, 2010; Skaggs *et al.*, 2012; Peng et al 2013; Lu *et al.*, 2013; Lu *et al.*, 2016). For example, Lu *et al.* (2016) showed that adopting controlled drainage in paddy fields in southeast China, where diffuse pollution is a critical problem, reduced N loss in surface water by 59- 96 % in most rice phenological stages.

Because soil loss is the main vector for P loss from fields, reducing use of P-fertilizer and controlling soil erosion by mulching, or maintaining a plant canopy cover for as long as possible, are the main ways to prevent the off-farm impacts of phosphorus (FAO, 2013). As discussed in Chapter 7, sediments carry pathogens and pesticides (as well as phosphorus) and can be physical pollutants in waterbodies as well. Sediment loss from arable land can be substantially reduced by adopting minimum tillage in place of conventional ploughing: a 68% reduction in sediment export was shown to equate to a reduction in phosphate loss of 81% (Jordan *et al.*, 2000). Other measures to control erosion include contour-strip cropping (Gitau *et al.*, 2006), no-tillage treatment (Francis and Knight, 1993), terracing (Sharpley *et al.*, 2001), hedgerows (Baudry *et al.*, 2000) and shelterbelts (Ryszkowski and Kedziora, 2007).

11.2 Good practices for livestock farms

Given the important role of livestock as a polluter (FAO, 2006b) it is imperative to accelerate the adoption of good practices in this sector. In extensive livestock farming, soil erosion and sedimentation can be addressed by taking measures against land degradation in pasturelands. Pollution exports from livestock farms can be also tackled through better management of animal diets, feed additives and medicines to minimize surpluses of, for example, drugs, nutrients or hormones, which can pollute water bodies. Improved manure management and better use of processed manure on croplands are also

key to controlling pollution. Industrial livestock production should be decentralized, so that wastes can be recycled without overloading the soils, and subsequently freshwater. Intensive livestock operations, such as feedlots that concentrate livestock, need to be managed as point sources of pollution and should follow specific national regulations (see Chapter 10). This section reviews some of these interventions.

11.2.1 Grazing management

Although pastures look harmless, they can be massive contributors to water quality problems if they are not managed properly, particularly through land degradation and soil erosion. Land degradation can be prevented by respecting the capability of the land: avoiding overgrazing, minimizing pasturing on steep slopes, and protecting stream banks from riverine degradation.

A vegetation cover prevents erosion by maintaining the soil in a condition where it can absorb rainfall, so that runoff does not concentrate into an erosive force. Allowing animals to graze vegetation to the ground (overgrazing) deprives the soil of its protective cover and exposes it to erosive agents. Overgrazing does not only result from having too many cattle on pasture; timing is actually more important, since pastures cannot support the same number of animals in the dry and wet seasons.

Producers can reduce land degradation and soil erosion in pastures (Carey and Silburn, 2006; Zhu *et al.*, 2015) by:

- Matching stock numbers to available feed during different seasons or in different years to avoid overgrazing;
- Regularly monitoring pastures to ensure that stock numbers match available pastures. Long-term weather forecasting, using predictive tools, has improved the options available for predicting droughts and feed availability.
- Using rotational grazing. Moving animals through a series of paddocks allows pasture plants time to recover, reduces soil erosion and improves forage quality.
- Selecting the types of animals to graze in different types of pasture in different seasons. Sheep and goats graze closer to the base of plants than cattle and, in dry periods, they can put more pressure on pastures.
- Locating watering points strategically to minimize stock concentration in areas that are vulnerable to erosion;

- Not using fire to control woody weeds or managing it very carefully since regular burning of pastures will further reduce ground cover and promote runoff and erosion.
- Integrating trees within pastures. Trees provide shade and shelter for animals, help recycle nutrients, provide stability to streambanks and prevent landslip on susceptible steep slopes.
- Managing runoff so that it spreads, rather than concentrating and causing erosion. Because engineering interventions, such as drains, tracks or roads, may cause a concentration of runoff, such interventions need to be planned with care so that they do not contribute to erosion.
- Minimizing pasturing on steep slopes, where the risk of erosion is greater. All soils are erodible – but some are more erodible than others. Broad-scale maps showing land types in particular regions can indicate what soils may occur on a farmer's property and are a useful planning tool.
- Protecting stream banks from riverine degradation. Producers may establish and maintain vegetation in riparian areas, protect these areas with fencing and use alternative water delivery systems to streams for watering livestock.
- Restoring degraded pastures with a mixture of species. This controls soil erosion better than monocultures because of their diverse and developed root systems

11.2.2 Management of feed, feed additives and drugs

In livestock systems, adjusting the animals' diet and/or improving feed conversion can reduce the level of nitrogen, phosphorous and trace element excretion with no harm to animal health, welfare or performance (Dourmad and Jondreville, 2007).

Producers can choose feeds with a high nutrient digestibility; use phytase to increase P digestibility or eliminate antinutritional factors. Nevertheless, these approaches may significantly increase costs to livestock farms. A more cost-effective method to reduce faecal and urinary losses of N and P is to manage feeding to meet the animals' requirements as they grow (Tamminga, 1992; Loyon *el al.*, 2016). For example, reducing the excess of protein content (rich in nitrogen in the form of $-NH_2$) in the diet of livestock has been reported to be a most cost-effective way to cut N excretion (and related NH_3 emissions). For each percentage point decrease in the protein content of the animal feed, total NH_3 emission is cut by 5–15% due to the reduced ammoniacal nitrogen in the manure (UNECE, 2014). A recent European survey revealed that another very common method for pollution abatement in pig and poultry production was 'phase feeding' to

meet an animal's nutrient requirements and prevent periods of overnutrition and the unnecessary enrichment of excreta and urine with nutrients (Loyon *et al.*, 2009).

The use of feed additives, hormones and medicines (including antimicrobial drugs) should adhere to national standards and international guidelines such as the FAO/WHO Codex Alimentarius for feed additives or the WHO guidelines on the use of medically important antimicrobials in food-producing animals (WHO, 2017). These guidelines aim to help preserve the effectiveness of antibiotics that are important for human medicine by reducing their use on animals and recommends that farmers and the food industry stop using antibiotics routinely to promote growth and prevent disease in healthy animals.

11.2.3 Manure management

Manure is one of the main environmental concerns in livestock production. Untreated manure contains pathogens and may also contain antimicrobials, hormones, heavy metals and other chemicals (see Chapter 8) that pose serious risks to human health and ecosystems (US EPA, 2013). For example, dairy cows excrete between 35 and 130 kg N and between 6 to 16 Kg P per year, and growing pigs excrete between 7 to 14 km N and around 2.5 kg P per year. This can contribute significantly to eutrophication and hypoxia in receiving waters (Brandjes *et al.*, 1995). Therefore, manure needs to be stored, treated, handled and disposed of – or preferably reused – safely.

Manure storage

Covering manure storage areas and protecting them from rain and rainfall runoff limits the possibility that the facility will overflow and reduces leaching losses considerably. Ponds or lagoons to store manure should be built in such a way that they minimize seepage to groundwater. The disposal or leakage of liquid manure should be prevented, as should the direct contact of manure with the underlying soil, except on some relatively non-permeable soils. The shape and size of the manure storage facilities is also important. For example, increasing the height of a storage facility reduces surface area and tends to reduce nutrient loss (FAO, 2013).

Urine needs also to be collected in livestock systems and stored to prevent losses of nutrient leaching, runoff and the associated pollution of surface and groundwater.

Manure treatment

Manure can be treated to stabilize organic matter and reduce putrescible material, to reduce its volume and decrease costs of transport, or to remove or degrade pathogens,

antimicrobials, hormones or other hazards to human health. Manure treatments include physical, chemical treatments and biological treatments.

Physical treatment of manure involves separating solids from the liquid fraction, typically through drying, settling, screening or filtration (James *et al.*, 2006). Chemical treatment involves the addition of coagulants, such as lime, alum or organic polymers, to separate the solids from the liquid. Quick lime (CaO) or hydrated lime ($CaOH$) are coagulants that have disinfectant properties as they increase the pH and keels in most pathogens, nevertheless the increase in pH increases NH_3 volatilization, reduces N content in manure and therefore decreases its fertilizing properties (James *et al.*, 2006). Reducing the water content of manure makes it easier to handle and transport.

To some extent, the biological treatment of manure occurs naturally in traditional storage facilities, where existing microorganisms start degrading different organic and inorganic compounds in manure. In addition, manure can be treated with specific methods such as composting or anaerobic digestion. These methods have relevant pathogen removal capacity. During composting aerobic microorganisms, the manure is decomposed in an exothermic process, which increases its temperature and keels or deactivates most pathogens, with the exception of some viruses and worms (US EPA, 2013). Recent research suggests that composting can also promote antimicrobial degradation and reduce the concentration of hormones (Dolliver *et al.*, 2008). Anaerobic digestion occurs in the absence of oxygen when anaerobic microorganisms degrade manure and generate biogas, which contains methane that can be reused for energy production. There are different types of anaerobic bioreactors, including plug flow reactors, complete mixed reactors and covered lagoons. Methane generation can contribute to energy saving at the farm level, or even to income generation if the energy is sold to a local utility. Anaerobic reactors separate solids from liquid to an extent. The liquid fraction has good fertilizing properties as nutrients are not removed in the process. Digested manure, as compost, has good properties as organic soil conditioners and some fertilizing capacity when manure is mineralized and nutrients released.

Manure utilization

Intensive livestock (and therefore manure) production is frequently concentrated in areas where logistics and the enabling environment are favourable. Given the high transport cost of manure per unit of nutrient, it tends to be used around intensive livestock farms. In these areas, nutrient budgeting and management plans are necessary to minimize nutrient surpluses. Such plans should aim to minimize nutrient accumulation in soil beyond a defined threshold and to reduce nutrient exports. Both the dosage and the timing of manure application are key.

Because of the N:P ratio in manure, NO_3- leaching as a result of manure application is usually not a problem if a zero P balance is maintained (i.e. the P applied through manuring does not exceed the amount of P removed by the crops). In such areas, mineral fertilizers should be used only when there is a deficit and the application plan should also consider the residual effects of previous applications. When the fertilizing effect of manure is thus calculated, the risk of NO_3- and P leaching is reduced.

The extent of NO_3- leaching and surface runoff is influenced by the time lapse between manure application and the growing period of the crop. The application of manure should be synchronized as closely as possible with the period of the crop's nutrient demand. In the wetter parts of Europe, an easy way of reducing nitrate leaching has been to ban slurry application in winter when rainfall is high, especially on sandy soils.

Resource recovery from manure

Livestock generate millions of tonnes of manure every day. This manure is increasingly considered as an economic asset rather than a liability because the resources it contains, mainly organic carbon and nutrients, can be recovered and used for energy generation (FAO, 2013), soil organic conditioning or fertilization (FAO, 2015). Yet manure is not always managed in a way that permits farmers to derive value from its reuse; meanwhile, millions of farmers struggle with depleted soils.

Otoo and Drechsel (2017) did a very comprehensive review of cases, mainly from the developing world, where resources were recovered from urban and agricultural waste and reused for beneficial purposes. From these cases, a number of business models for resource recovery and reuse were derived (see e.g. Box 11.3). Each model explains the value proposition and value chain of the business, the institutional set up and the risks in terms of viability and safety. The business models show pathways to increasing cost recovery (or even to achieving full profitability) from the sale of recovered resources to create livelihoods, enhance food security, support green economies and reduce waste.

In addition to the well-validated cases reviewed by Otoo and Drechsel (2017) or by FAO (2015), there are a good number of promising innovations with the potential to be upscaled. A number of technologies have arisen from the United States of America EPA's nutrient recycling challenge, a competition hosted by US EPA and its partners to develop effective and affordable ways to extract nutrients and create products that farmers can use, transport, or sell more easily to places where nutrients are in demand (US EPA, 2017). For example, phosphate can be recovered as struvite from biodigesters used to treat farm wastes and slurries.

BOX 11.3 Business model for generating power from manure (Otoo and Drechsel, 2017)

Business characteristics

Geography: Rural regions with livestock farming and a large livestock industry;

Scale of production: 16 KW up to 5 MW of electricity; 22 000 to 700 000 tonnes of carbon dioxide (CO_2) equivalent/year in carbon credits;

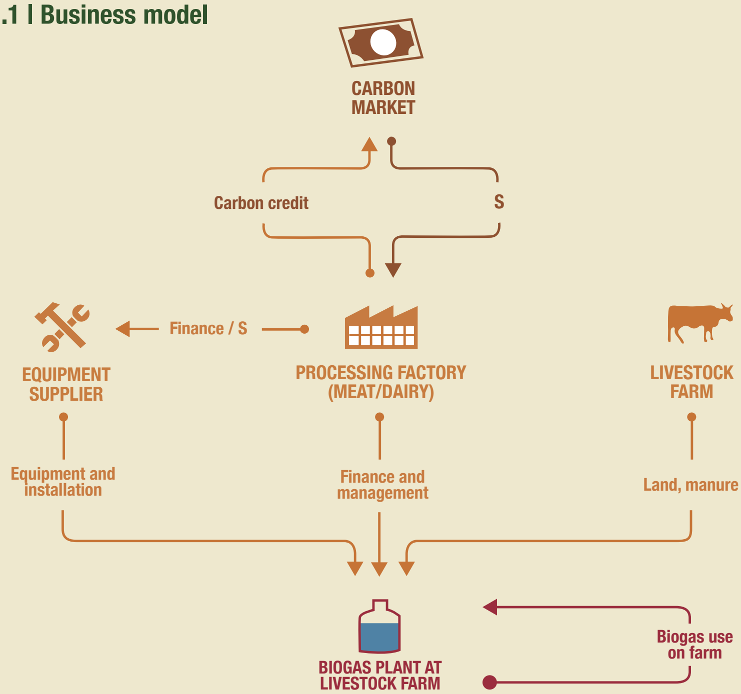
Type of organizations: Food companies, livestock processing factories, farms and/or communities with livestock;

Investment cost range: US$500-5 000/KW for capacities ranging between 1 MW and 3 MW

Key costs: Investment costs (engineering, construction, equipment, commissioning); costs of training farmers; and operational and data management costs (labour and maintenance);

Revenue stream: Trade of carbon credits; savings from avoided electricity costs and potential sales of electricity, or biogas and bioslurry (fertilizer).

Figure 11.1 I Business model



Source: Otoo and Drechsel, 2017.

Business model description

This business model uses livestock manure to produce power and/or thermal energy that can be used internally by an enterprise, or sold to the grid or to households and businesses. Using anaerobic processes, manure is fed into a biodigester to produce biogas for ➤

➤ electricity generation. Additional revenue can be earned from the sale of carbon credits and biofertilizer, a byproduct of the process. The business can be established by either a livestock processing factory, a farm or a remote community with personal livestock. In the first instance, the factory owner installs biodigesters on the farms in its supply chain in order to ensure sustainability and gain additional revenue from carbon credits. The factory finances the installation of the biogas plant by an equipment supplier on one of its farms. The farm then operates and maintains the plant, and gradually pays back the factory owner by transferring its carbon credits until it gains ownership of the plant. The energy produced from the livestock waste is used on the farms. For a remote community with livestock, the regional government can install biodigesters as part of a rural electrification programme.

11.3 Good practices on aquaculture farms

The pollutants produced by aquaculture, as for livestock, mainly originate from uneaten feed and excreta from aquatic organisms. Fed aquaculture is typically more polluting than non-fed aquaculture (Li and Shen, 2013), which can even depollute water when fish or mollusks uptake or filter already existing nutrients in the water.

The larger risks from aquaculture pollution come from ammonium, nitrate and nitrite, phosphorus and organic matter, which are present in feces or unutilized feed, as well as other inputs such as growth hormones or pesticides (e.g. bactericides, fungicides, algaecides, herbicides, molluscicides, etc.). The abundance of organic matter can lead to oxygen deficiency, which can kill fish, and as well as causing the release of poisonous or harmful substances, such as ammonia and hydrogen sulfide.

To minimize such risks, aquaculture farms should adopt good management practices that protect the surrounding aquatic environment. These practices include establishing a suitable production biomass based on the carrying capacity of the water body; avoiding excess feed by standardizing feed inputs; using fish drugs correctly and avoiding prohibited drugs; and removing, treating and disposing of excessive nutrients in fishponds (Li and Shen, 2013).

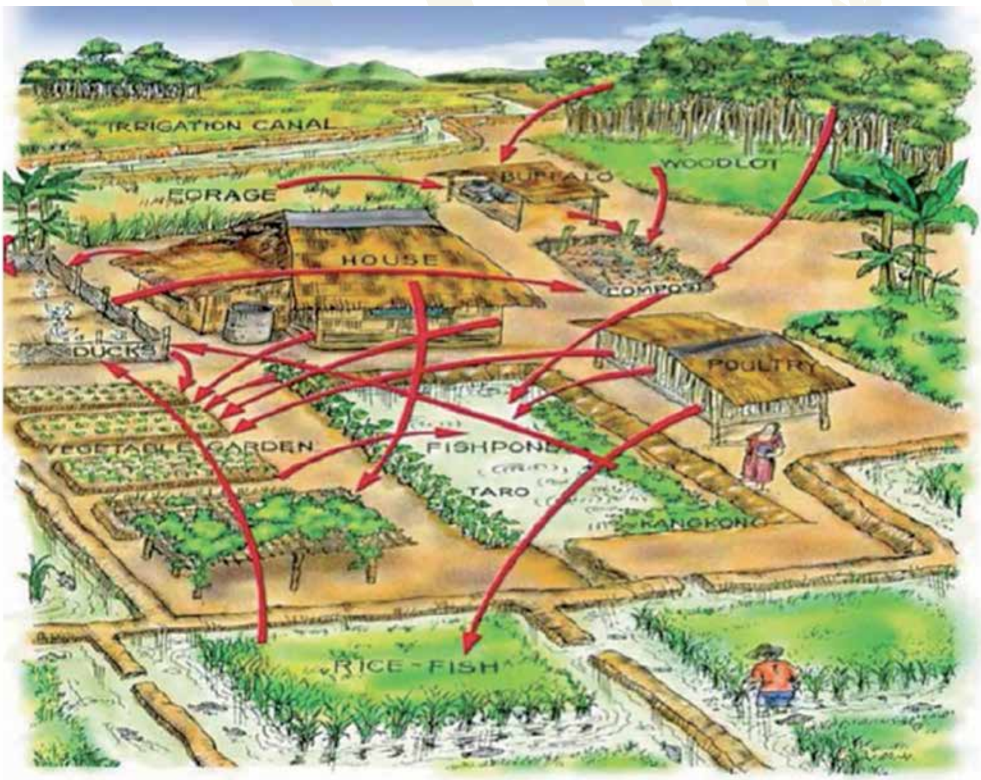
Promoting integrated systems in which the waste of one species serves as a food source for another can be also a cost-effective way of minimizing water pollution. Such integration is a key element of the 'ecosystem approach to aquaculture (EAA)', which 'is

a strategy for the integration of the activity within the wider ecosystem in such a way that it promotes sustainable development, equity, and resilience of interlinked social and ecological systems' (Soto *et al.*, 2008).

Integrated aquaculture-agriculture (IAA) promotes the integration of crops, vegetables, livestock, trees and fish to achieve more stability in production, efficiency in resource use and conservation of the environment (Figure 11.1). In addition to reducing pollution through waste recycling, IAA can also limit pesticides use. Evidence shows that although rice yields are similar to those in simple rice systems, an integrated rice–fish system uses 68 percent less pesticide than does rice monoculture (Xie *et al.*, 2011). Together with the fact that most broad-spectrum insecticides are a direct threat to aquatic organisms and healthy fish culture, knowledgeable farmers are much less motivated to spray pesticides (FAO, 2012).

The same principle applies to integrated multi-trophic aquaculture (IMTA,) which involves farming different aquaculture species together in a way that allows the waste of one species to be recycled as feed for another.

FIGURE 11.1 | **Integrated agriculture-aquaculture**



11.4 Off-farm responses

The use of simple, natural off-farm techniques can be a cost-effective way to reduce the amount of pollution entering into surface waters (WWAP, 2018). Two ecological engineering measures are widely applied to limit the movement of pollutants through the landscape: 1) constructed wetlands that capture, filter and detoxify agricultural pollutants; and 2) buffer strips on-farm and along waterways that filter the water and prevent pollutants from entering the water system and being transported further downstream.

11.4.1 Constructed wetlands

Constructed wetlands have been mainly employed to treat point-source wastewater, including urban and agricultural stormwater runoff (Libhaber and Oerozo-Jaramillo, 2012; Birch *et al.*, 2004). Constructed wetlands can also be used remove sediments, nutrients and other pollutants from agricultural drainage systems (Verhoeven *et al.*, 2006).

Constructed wetlands have been shown to be effective in trapping or removing different pollutants (nutrients, sediment, coliforms, pesticides, heavy metals). For example, it is estimated that restoring the total wetland area of the Baltic Sea Basin (1 700 000 km^2) would increase N removal rates before discharge to the sea from a range of 5–13% to 18–24% (Jansson and Dahlberg, 1999). In South Africa, while water quality in the Lourens River has been declining over the last few decades, it was determined that 75–84 percent of suspended sediment, orthophosphorus, and nitrate were sequestered by a downstream wetland (Laan, 2009).

The capacity of wetlands to capture and treat agricultural effluents depend on various factors, including the type of pollutant. Birch *et al.* (2004) report on the performance of a small constructed wetland in Sydney (700 m^2), which serviced an urban catchment area of 480 000 m^2. The average removal efficiency of trace metals Cr, Cu, Pb, Ni, and Zn was 64%, 65%, 65%, 22%, and 52%, respectively, whereas Fe and Mn increased in the outflow by 84% and 294%, respectively. The average removal efficiency of NOx and total nitrogen was much lower at 22% and 16%, respectively. Sediment trapping in storm runoff was impressive: during two high-flow events, the removal efficiency of total suspended solids was between 67% and 98% compared to lower values at lower flow rates of 9% to 46%.

The design of constructed wetlands, often used in small urban catchments, is increasingly grounded in basic guidelines, such as the following, which were used in South Australia (Cooper and Moore, 2002):

- Constructed wetlands should be designed to require minimal maintenance.
- Constructed wetlands should mimic natural systems.
- The use of natural energies should be incorporated in the design.
- Wetland systems must be designed with the landscape in mind.
- Multiple objectives should inform the design, with at least one major objective and several secondary objectives.
- Sufficient time must be allowed for the system to start operating properly.

Constructed wetlands can also remove pesticides from water to an extent. Darby (1995) determined that the majority of the organophosphate insecticide chlorpyrifos entering a constructed wetland was rapidly bound to the sediment and plant material in the inflow area of the wetland cells. Follow up experimentation with chlorpyrifos and two herbicides (atrazine and metolachlor) showed that 55 percent of the chlorpyrifos was attached to sediments and 25 percent was stored in plant material (Moore *et al.*, 2002). The same wetland reduced spikes of atrazine in storm flows and decreased atrazine concentrations by 26 to 64 percent from inflow to outflow.

11.4.2 Riparian Buffer zones

Riparian buffer zones are vegetative strips at the margins of fields or along river and stream banks that contain native trees, bushes, shrubs, flowers, grasses and/or plants (Gregory *et al.*, 1991; Martin *et al.*, 1999; Osborne and Kovacic, 1993). Riparian buffer zones can differ in design, vegetation type and distribution of vegetation. They can comprise a single vegetation type, for example a grass species (also called grass filter strips) or trees species (referred as buffer forest), or they can include mixed vegetation containing both grass and trees. The design can include up to three individual zones, each of which utilizes a different vegetation type. In general, the most recommended design is a buffer strip with three zones and mixed vegetation (Welsch, 1991).

Buffer zones are a well-established measure that have proven effective in decreasing the concentration of pollutants and sediment entering waterbodies. In agriculture and some forestry operations, a buffer zone normally implies a strip of vegetation that acts as a filter for sediment, nitrogen, phosphorus and pesticides. A buffer zone can have other functions as well, including stream shading and water temperature cooling (by tree canopies); reducing runoff velocity; flood reduction and water storage; carbon sequestration; biomass production; economic benefits from, for example, logging or harvesting fruit;

soil and channel stabilization; erosion prevention; water purification (e.g. of bacteria and pathogens) the provision of terrestrial and stream habitats, food sources and hydrological connectivity; and finally, cultural services (Anderson and Masters, 1997). Despite decades of research on riparian buffer zones, the scientific literature remains mostly biased towards single functions. Buffer zones should be viewed as a conservation practice to be used in conjunction with other on-site management strategies that reduce erosion, sediment transport, and runoff. To be truly effective, they should be designed, constructed and regularly maintained (e.g. by removing tree and plant litter). Further information on design guidelines for buffers zones can be found in Bentrup (2008).

Although riparian buffer zones are being established along thousands of streambank miles throughout the United States of America, the benefits of different designs (e.g. in terms of width, length, slope, type of vegetation and placement in the watershed) are still not well understood (Fischer and Fischenich, 2000). A simple guideline remains true: 1) wider strips are appropriate for higher flows (floods) and for higher removal rates of nutrients and sediment, but a strip width that is economically viable depends on farm size and setting; 2) removal rates of nutrients and sediments are higher when buffer zones are placed adjacent to smaller streams than larger ones (Norris, 1993); 3) trees are more effective in removing nitrogen and phosphorus from groundwater, whereas grass species are better in removing nitrogen and phosphorus attached to sediment in surface runoff (Martin *et al.*, 1999; Osborne and Kovacic, 1993); 4) removal rates increase as a slope gets steeper, but after exceeding 10% steepness, removal rates decline (Zhang *et al.*, 2010); and 5) fencing around buffer zones is recommended to keep cattle and humans away. Buffer zones are most effective when the flow is shallow (non-submerged), slow, and enters the strip uniformly along its length. In hilly terrain, flow concentrates rapidly, producing higher velocities and larger flow depths that can rapidly submerge the vegetation and significantly reduce the effectiveness of the filter strip.

Studies vary as to their assessment of the effectiveness of buffer zones. There is a large body of literature, dating back 30 years, that documents the performance of buffer zones in removing sediment, nitrogen, phosphorus and pesticides. A literature review on different study outcomes showed that phosphorus removal by grass buffer zones varies from 40 to 100% (Dorioz *et al.*, 2006). In general, buffer zones are able to remove nitrogen by 2 to 100%, phosphorus by 22% to 100%, sediment by 9.8 to 100%, and pesticides by 4.2 to 100% (Zhang *et al.*, 2010). The effectiveness of removal depends on factors, previously mentioned, such as width, slope, placement and vegetation type. However, in some cases riparian buffer zones even function as a source of nutrients and sediment instead of a sink (Sabater *et al.*, 2003).

Different widths and combinations of vegetation buffer zones are appropriate for different slope, vegetation soil conditions and loads. A summary of guidelines for different functions and conditions is presented in Table 11.1 The United States of America National Conservation Buffers Initiative (USA-NCBI) targeted 3.2 million kilometres of riparian zone, over an area of 3 million hectares for completion by 2005. It set minimum and maximum widths that landowners would need to establish in order to receive funding assistance, ranging from a minimum of 9 metres (recommended by Wenger, 1999) for some herbaceous filter strips, up to a maximum of 45 metres for forested riparian buffer strips. As a separate programme, the USA-NCBI also funds the development of habitat corridors to enhance biodiversity and habitat connectivity.

Table 11.1 l General Riparian Buffer Strip Width Guidelines, USA

Function	Description	Recommended width
Water quality	Buffers – especially dense grassy or herbaceous buffers on gradual slopes – intercept overland runoff, trap sediments, remove pollutants, and promote ground water recharge. On low to moderate slopes, most filtering occurs within the first 10 metres, but greater width is necessary for steeper slopes, in buffers comprised of mainly shrubs and trees, where soils have low permeability, or where non-point source pollution loads are particularly high.	5 to 30 m
Riparian habitat	Buffers, particularly diverse stands of shrubs and trees, provide food and shelter for a wide variety of riparian and aquatic wildlife.	30 to 500 metres +
Stream stabilization	Riparian vegetation moderates soil moisture conditions on stream banks, and roots provide tensile strength to the soil matrix, enhancing bank stability. Good erosion control may only require that the width of the bank be protected, unless there is active bank erosion, which will require a wider buffer. Excessive stream bank erosion may require additional bioengineering techniques.	10 to 20 metres
Flood attenuation	Riparian buffers promote floodplain storage due to backwater effects, they intercept overland flow and increase travel time, resulting in reduced flood peaks.	20 to 150 metres
Detrital input	Leaves, twigs and branches that fall from riparian forest canopies into the stream are an important source of nutrients and habitat.	3 to 10 metres

Source: Fischer and Fischenich, 2000.

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CHAPTER 12. MAIN MESSAGES

In the light of growing urbanization and associated challenges affecting our aquatic environment, the significance of agricultural water pollution appears to be sidelined. However, the increasing need to feed the growing global population has required agriculture to expand and intensify. The farm area equipped for irrigation has more than doubled since the 1960s; the total number of livestock has more than tripled since the 1970s; and aquaculture has grown more than twenty-fold since the 1980s, especially inland-fed aquaculture and particularly in Asia. Moreover, land, water and other agricultural inputs are used more intensively than ever before. In addition to population growth, changes in calorie intake and diets have increased the demand for a wider variety of foods, including more meat and dairy products, and led to an increased water footprint in terms of water quality. Where the resulting agricultural intensification is not well managed, its benefits for society are often accompanied by significant environmental and health costs, in particular through water pollution.

Historically, the analysis on water pollution has focused primarily on individual sources, their nature and impact. However, more attention is needed to understand linkages between these factors, as well as pollution drivers, types and loads, distribution dynamics and comparative risks for different aquatic ecosystems. Of major concern are:

- Excessive nutrient application: Intensified cropping systems with limited or no fallow periods can rapidly deplete agricultural soils of important plant nutrients

unless fertilizers are applied. During the 20th century, the use of fertilizers rapidly increased as did the discharge of surplus nutrients to water from global agriculture.

- Pesticide overuse: Pesticides are another important requirement of many intensified farming systems. The overuse of pesticides is often associated with the accumulation of persistent organic pollutants in soil and water resources, potentially affecting the food chain. Although the risks of pesticides in the environment are better understood than in the past, regulations, as well as the monitoring of their use, often remain ineffective or inefficient.
- Salinity: Salinization of soils and freshwater bodies is still a leading concern for water quality and agricultural production, especially in arid and semi-arid regions. With an estimated global volume of 1,260 km^3 every year (which corresponds volume-wise with the minimum flow of the Congo river), drainage from irrigation mobilizes and transports billions of tonnes of salts to freshwater bodies.
- Increased erosion and sediments: Agricultural expansion on formerly uncultivated slopes, as well as changes in land use from forestry to agriculture, have accelerated runoff and erosion with increasing sediment loads affecting river quality and aquatic life, as well as the functionality of storage reservoirs. The average global erosion rate on cropland is estimated at 10.5 tonnes per ha per year, which can increase in hilly landscapes of the tropics and subtropics to 50–100 tonnes per ha. With eroded topsoil, soil organic matter, nutrients and, for example, pesticides also find their way into water bodies.
- Livestock: The trend towards increasing consumption of meat and dairy products has led to increasing investments in livestock production. Similar to fertilizer, livestock wastes also constitute major nutrient sources of global water pollution, leading to the potential contamination of drinking water and eutrophication of lakes, rivers and coastal areas. Moreover, animal manure and slurries also contain large amounts of pathogens, as well as veterinary medicines, such as antibiotics, which can affect aquatic life and the food chain. More than 85 percent of the world's faecal waste is from domestic animals, such as poultry, cattle, sheep and pigs.

Given the qualitative and quantitative complexity of possible pollutants, system-based modeling approaches are increasingly needed to support science-based policy responses. New models that are capable of simulating interactions between production systems, agricultural inputs (and livestock wastes), considering temporal, as well as spatial changes in aquatic ecosystems, would help establish a more solid understanding

of the different water pollution pathways and potential remediation scenarios. These could also provide regulatory support by calculating, for example, maximum pollution loads (or caps). However, the results of water quality modeling can only be as good as the data used and, so far, many regions lack credible water quality data from farm to watershed. Increased data collection will help develop water quality models and translate their results into better water policies.

Water quality degradation has a variety of economic impacts, including human health, ecosystem health, agricultural and fisheries productivity and recreational and amenity uses. Although some of these effects are tangible, and costs appear significant, many impacts are difficult to value, especially given the paucity of data.

The most effective way to reduce water pollution from agriculture is to limit pollutants at the source or intercept them before they reach vulnerable ecosystems. Once in the system, the costs of remediation progressively increase. Despite significant progress on pesticide and fertilizer regulations, enforcement and actual monitoring of the final use of these inputs remains challenging. To adopt good agricultural practices (GAP) farmers also require more education, awareness and economic incentives, ideally leading to cooperative agreements and wide adoption across landscapes.

Policies that address water pollution from agriculture should therefore form (a) an integral part of overarching water policy frameworks at the national or river-basin scale, and (b) influence policies at a higher level of food security and nutrition to encourage people to adopt diets that are more sustainable in view of human and environmental health.

Current patterns of agricultural expansion and intensification are bringing unprecedented environmental externalities, including impacts on water quality. While water pollution is slowly starting to receive the attention it deserves, the contribution of agriculture to this problem has not yet received sufficient consideration.

We need a much better understanding of the causes and effects of agricultural water pollution as well as effective means to prevent and remedy the problem. In the existing literature, information on water pollution from agriculture is highly dispersed. This report is a comprehensive review and covers different agricultural sectors (including crops, livestock and aquaculture), and examines the drivers of water pollution in these sectors as well as the resulting pressures and changes in water bodies, the associated impacts on human health and the environment, and the responses needed to prevent pollution and mitigate its risks.

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