CHAPTER 11. ON-FARM AND OFF-FARM RESPONSES

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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.

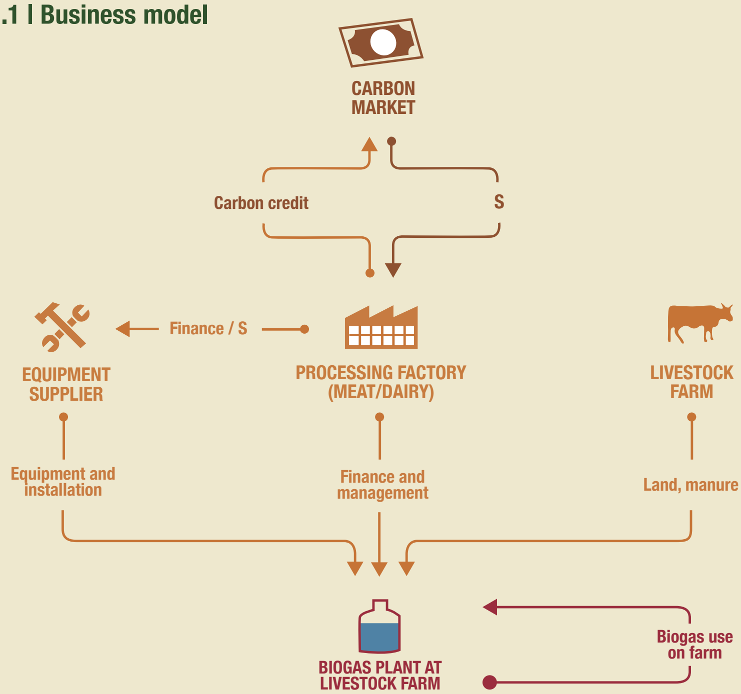
In addition to the well-validated cases reviewed by Otoo and Drechsel (2017) or by FAO (2015), there are a good number of promising innovations with the potential to be upscaled. A number of technologies have arisen from the United States of America EPA's nutrient recycling challenge, a competition hosted 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.

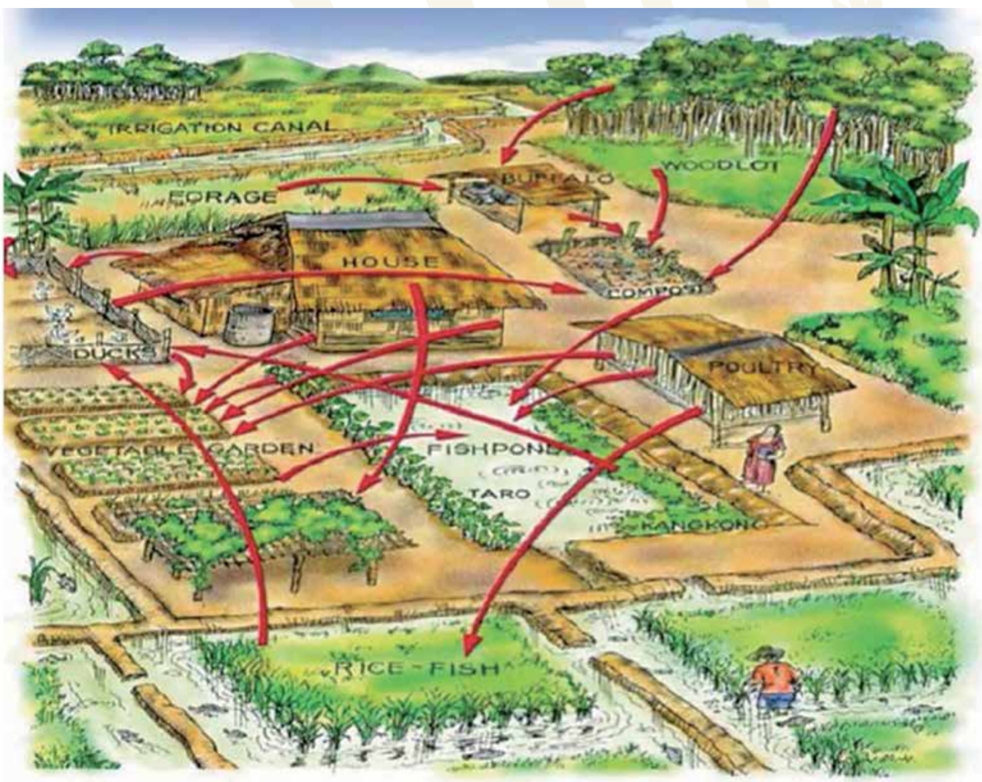
To minimize such risks, aquaculture farms should adopt good management practices that protect the surrounding aquatic environment. These practices include establishing a suitable production biomass based on the carrying capacity of the water body; avoiding excess feed by standardizing feed inputs; using 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 I 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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