Recognizing diversity in wetlands and farming systems to support sustainable agriculture and conserve wetlands

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Abstract

Agriculture is a main driver of global decline in wetlands, but in addressing its impact the diversity in agricultural production systems and their catchment interactions must be recognized. In this paper we review the impacts of food production systems on wetlands to seek a better understanding of agriculture-wetland interactions and identify options for increasing sustainability. Eight farming system types were defined based on natural resource use and farming intensity, and their impact on different wetland types assessed through their direct drivers of change. Indirect drivers (such as decision-making in food systems, markets, globalization and governance) were also summarized. Findings show that most inland wetlands are influenced by farming directly, through changes in water and nutrient supply and use of pesticides, or indirectly through catchment water, sediment and nutrient pathways. Coastal wetlands are mostly influenced indirectly. More sustainable food production can be achieved through continued protection of wetlands, improving efficiency in agricultural resource use generally (e.g. through conservation agriculture), but also through more integration within production systems (e.g. through crop-livestock-fish integration) or with wetlands (integrated wetland-agriculture), more support for small-scale producers, and a transformation towards balancing the provisioning, regulating and cultural ecosystem services of wetland agroecosystems within catchments.

Key words

Food systems, wise use of wetlands, wetland ecosystem services, farming systems, livelihoods, catchment management, Ramsar Convention, sustainable agriculture

Introduction

Wetlands provide a wide range of ecosystem services, from food production, regulation of water quality, and flood protection to climate change mitigation and providing habitat for many animal and plant species (MEA 2005; Russi et al. 2013; Ramsar Convention 2018a). They also support human livelihoods and sustainable development (Ramsar Convention 2018b). Despite their importance for humankind, the global area of natural wetlands has been declining from before 1700, with a peak in loss rates in the mid-20th century (Fluet-Chouinard et al. 2023). Since 1700, where data are available, about 21% of the world's wetlands have been lost, with those remaining under pressure from drainage, pollution, invasive species, unsustainable use, disrupted flow regimes and climate change (Fluet-Chouinard et al. 2023).

Since around 1950, the world population more than tripled from around 2.5 billion to almost 7.9 billion in 2021, and is projected to grow further and then stabilize at almost 11 billion by the end of the twenty-first century (UN 2021). Global food production (crops, livestock, fish) has largely been able to keep up with this population increase through intensification of production by more than doubling the area equipped for irrigation, increasing the proportion of irrigated land from about 10% to the current 25%, increasing the use of fertilizers and pesticides, genetic improvement of crops, livestock and fish, and other improvements in farming technology (Pellegrini and Fernández 2018). However, this increased production has resulted in multiple impacts on the environment in the form of loss of forests, wetlands and natural grassland, decline and loss of species populations, increased sedimentation, acidification and eutrophication, and greenhouse gas (GHG) emissions (Foley et al. 2005; Clark and Tilman 2014).

Awareness of the link between wetland decline and agricultural development is increasing, and agricultural development is consistently identified as one of the main drivers (Zedler 2003; Wood and Van Halsema 2008; Verhoeven and Setter 2010; van Asselen et al. 2013, Ramsar Convention 2022). In the high-income countries in the northern hemisphere (particularly Europe and North America), conversion of wetlands to agriculture started hundreds of years ago as part of economic development (Fluet-Chouinard et al. 2023). In these countries, growing awareness of the importance of wetlands has led to protection and conservation policies for wetlands and to restoration efforts. In large parts of Latin America, Africa and South and Southeast Asia, economic development and population growth can be expected to continue exerting pressure on wetlands. The impacts of agriculture occur on two levels (Wood and van Halsema 2008): the most visible impacts are direct, by draining wetlands and converting wetland vegetation to crops. For example, 11-15 percent of peatlands are estimated to have been drained globally, largely for cropping, plantations or livestock (FAO 2020a). But wetlands can also be affected indirectly, when agricultural activities upstream of wetlands alter the flows of water, nutrients and species to wetlands. With both types of impact, the role of wetlands in landscapes is affected, with consequences for water quality, habitats and biodiversity, and ecosystem services and human well-being.

Not all agriculture-wetland interactions are negative, and wetlands under certain conditions can also support agriculture, e.g. as a source of irrigation water (Rijsberman and Da Silva 2006; Everard and Wood 2018). Many wetlands are part of fertile floodplains in open wooded landscapes with good water availability and favourable conditions for crop or livestock production. Other wetlands are less suitable for agriculture, such as those with peat soils that are mostly acidic and non-fertile, or with vegetation that would be directly affected by agricultural use. With appropriate engineering (including drainage of flooded soils and flood regulation), conditions for wetland agriculture can be optimized for agricultural productivity, although often at the expense of other wetland ecosystem services and biodiversity (Falkenmark et al. 2007).

It is not always clear how knowledge of agriculture's impacts can be used for effective protection and sustainable management of wetlands. Often, the impact of "modern" agriculture consists of conversion of wetlands and the total loss of their ecosystem functions. However, as there are many wetland and agricultural system types the possible interactions are also many, and sometimes complex. Interactions should be understood not only at the wetland scale but also at the wider catchment scale and in terms of the social-economic and institutional context of wetlands, and therefore recognizing the diversity in production systems and their interactions with the landscape, including wetlands, is important. This is related to where agricultural systems are located (altitude, latitude) vis-á-vis wetland types; and to the resource use and emissions of different production systems. It is also important to consider the way in which individual farming systems are operated. While some systems are inherently more efficient in water and nutrient use, differences in farming practices can also lead to different impact on wetlands. The need to analyse wetland-agriculture interactions, and the importance of placing wetland management planning in the context of the river basin or catchment has been recognized for some time (Wood and Van Halsema 2008; Ramsar Convention 2010a; Finlayson et al. 2024).

The objective of this paper is to review agriculture-wetland interactions by detailing the impacts of different food production systems on different wetland types, and to explore the transition to more sustainable food production that maintains or enhances wetland ecosystems. After summarizing the different wetland types and their functions in the landscape, we define the world's dominant food production systems and briefly characterize their resource use. We analyse the direct drivers of change in wetlands created by these production systems, and their interactions with different types of wetlands, both at local and at catchment scale. We also review the indirect drivers of food production systems. Finally we suggest pathways for a transition to more sustainable agriculture-wetland interactions, hoping that a more differentiated analysis will point the way to options for reducing the impacts of food production and slowing down the trend of wetland loss and degradation.

Wetlands and their importance at the global and landscape levels

The most recent estimates show that the total area of wetlands in the world is between 15 and 16 million km^2, or some 10% of the global land surface. The Ramsar classification defines 42 types of wetlands in three broad categories: inland wetlands (20 types), coastal-marine wetlands (12 types), and human-made wetlands (10 types) (Ramsar Convention 2010b), roughly making up 80%, 10%, and 10% of the world's wetlands, respectively (Davidson and Finlayson 2018; 2019). The most recent area estimate for marine and coastal wetlands is 1.42 million km^2 whereas inland wetlands occupy 11.79-12.79 million km^2 (Davidson and Finlayson 2018; 2019). Within inland wetlands, marshes and swamps on alluvial soils (river floodplains), and natural lakes together form about 50% of the area. Peatlands form over 33% of inland wetlands (Davidson and Finlayson 2018). Data on the extent of human-made wetlands (e.g. reservoirs, ponds, rice paddies, constructed wetlands) are incomplete, the global area for which data exist is 1.80 million km^2 (Davidson and Finlayson 2018). When discussing the interactions between agriculture and wetlands, human-made wetlands have a special position because some of them, like rice fields and aquaculture ponds, are wetlands and agricultural systems at the same time. Some loss and degradation of natural wetlands is caused by conversion to agricultural or aquacultural use, and represents a gain in human-made wetlands (Davidson 2014).

A distinguishing feature of wetlands is their water source: rain, surface water, or groundwater. Isolated wetlands are fed by rainwater or groundwater and are not connected to a stream, river or coastal system. Floodplain wetlands are fed by surface flows and floods, in addition to rainfall and possible groundwater input (Bullock and Acreman 2003; Acreman and Holden 2013), while coastal wetlands are influenced by estuarine and marine hydrosystems. Wetlands modify the flows of water and nutrients and the movement and productivity of plants and animals in the landscape. The nature and extent of this modification are determined by the type of wetland and its location in the catchment. The processes underlying these roles are jointly referred to as the functions of wetlands, and are the biophysical basis for their ecosystem services (MEA 2005; Díaz et al. 2015; Ramsar Convention 2018a). Wetland functions can be classified into hydrological, biogeochemical, and ecological functions (Maltby et al. 2009; de Groot et al. 2010).

The *hydrological functions* include floodwater detention, groundwater recharge and discharge, and sediment retention. The flow of sediment and nutrients is determined to a large extent by hydrological pathways in catchments or coastal environment. Rainfall can run off the surface or infiltrate into the soil, where water can move vertically from the earth surface to the base rock, and laterally from hillslopes to streams and rivers. The extent to which wetlands slow down or store water, recharge or discharge aquifers, or retain or export sediment depends on their location in the catchment, the geomorphology and topography of the landscape, and the seasonal variations in rainfall. Floodplain wetlands generally reduce or delay floods, but headwater wetlands can either decrease or increase flood peaks. Wetlands evaporate more water than forests, grasslands and arable lands, and as a result wetlands often reduce flow to downstream rivers in dry periods (Bullock and Acreman 2003; Burt and Pinay 2005; Lohse et al. 2009; Acreman and Holden 2013; Guo et al. 2018; Hu and Li 2018).

The *biogeochemical functions* include nutrient retention and export, carbon retention, trace element storage and export, and organic carbon concentration control through processes like sedimentation of particulate matter, uptake and storage of nutrients in vegetation, and microbial processes. Nutrients can enter the wetland system through surface or sub-surface inflow or through aerial deposition, and leave by streamflow or release to the atmosphere. Nutrients can also be stored in vegetation biomass or by adsorption to soil particles. Surface flows carry sediment, nutrients adsorbed to sediment particles, and dissolved nutrients. Detachment and transport of soil and sediment particles is caused by water erosion (the physical force of surface runoff) but can also be caused by wind or ice and by human activities such as soil tillage (Montgomery 2007; Labrière et al. 2015). Subsurface flows transport dissolved compounds, including nutrients, metals, and dissolved organic compounds. As water passes through wetlands, residence time often increases and biological processes increasingly influence the composition of the water (Burt and Pinay 2005; Lohse et al. 2009; Pärn et al. 2012). These processes are controlled by the oxygen content of the sediment and by vegetation processes, and are influenced by the degree of water-logging, the hydraulic retention time, and hydraulic loading.

Large amounts of carbon, nitrogen and phosphorus are stored in wetland vegetation and soils, particularly in wetlands with organic soils, where flooded conditions prevent rapid aerobic decomposition of organic matter. Stored nutrients can be released when conditions change, e.g. when accumulated sediment is flushed out of the system by peak flows, or when wetlands are drained and soil organic matter starts decomposing. Peat wetlands, mangroves, saltmarshes and seagrass beds store the largest part of the world’s soil carbon (in coastal wetlands now often referred to as 'blue carbon'), and anthropogenic disturbance (e.g. from agriculture) and climate change contributes to accelerated release of this carbon to the atmosphere (McClain et al. 2003; Fisher and Acreman 2004; FAO 2014; Johnson et al. 2018; Were et al. 2019). Dissolved organic C and N, and the inorganic forms of nitrogen (NO_3, NO_2, NH_4) can all be transported in surface and sub-surface flows, and about 25% of N inputs into catchments are exported by streams, regardless of catchment size and land use (Howarth et al. 1996; Vitousek et al. 1997; Galloway et al. 2004; Durand et al. 2011). Denitrification is another important natural N export pathway from catchments, and floodplain wetlands with available NO_3 and anaerobic, flooded conditions are an important location for denitrification (Piña-Ochoa and Álvarez-Cobelas 2006; van Cleemput et al. 2007).

The *ecological functions* of wetlands include ecosystem maintenance, providing habitat to a wide range of species, and food-web support (Maltby et al. 2009; De Groot et al. 2010; Ramsar Convention 2018a). Wetlands are among the most biologically productive ecosystems, and wetland plants and animals are important for the cycling and storage of nutrients. Streams and wetlands provide connectivity in catchments which is of crucial importance for the dispersal of species and the natural maintenance of network populations (USEPA 2015; Boudell 2018; Cosentino and Schooley 2018). Many plant seeds are transported by wind or by animals such as birds and fish. Dispersal by water is important for plant propagules, fish and macroinvertebrates; but less important for aquatic plant seeds (mostly produced out of water) and freshwater insects (usually emerge from the water and fly to other streams). In coastal and marine environments, aquatic habitats are more continuous and it is mostly the larval stages that disperse. Wetlands are also important for migratory birds, amphibians and reptiles and their movement between reproduction habitats and non-breeding habitats where they forage and become adults (Begon et al. 1996; Horn et al. 2011; Rittenhouse and Peterman 2018).

Agricultural production systems and farms

Classification of agricultural systems

Food production can be realized by cultivating crops or animals, or by capturing them from wild populations. Most current definitions of agriculture and 'agrifeed systems' include forestry and capture fisheries, both inland and marine (Campanhola and Pandey 2019), but here we focus on the interactions of crop, livestock and fish cultivation with wetlands and exclude terrestrial forestry and capture fisheries from the current analysis while recognizing the importance of conducting a similar analysis for these sectors (FAO 2020b; FAO/UNEP 2020). Agricultural production is then defined as any form of fish or livestock rearing, cultivation of crops or a combination of these for the production of food, feed or fibre (Lewandowski et al. 2018). It is realized in a wide variety of cultivation and husbandry systems, and is an important component of food security policies although food security is a much broader concept which also encompasses the physical and economic access to food and its nutritional quality (FAO 1996). Agriculture also includes the production of biomass for energetic or material use (Lewandowski et al. 2018). The occurrence of agricultural systems across the globe depends on the biophysical characteristics of different earth regions (climate and landscape), as well as cultural, socio-economic and other factors. The diversity of agro-ecological environments is shaped by rainfall patterns, water availability and soil quality, topography, and temperature. Agricultural systems evolved in the environment in which they are practiced, and are continuously developing (Tow et al. 2011).

Different attributes have been used to characterize farming systems (Tivy 1990; Dixon et al. 2001; Harrington and Tow 2011; FAO 2011b, 2016, 2018a,b; Lewandovsky 2018a; FAO/IWMI 2018): water availability, climate, altitude, farm size, production intensity, dominant liveilhood source, and location (e.g. forest-based, coastal). The key factors in most classifications are: the natural resource base (water, soils, genetic resources, climatic and landscape factors); the dominant pattern of farm activities and household livelihoods, including relationship to markets; and the intensity of production activities. This led to different farm system categorizations, e.g. for developing countries (Dixon et al. 2001), and for rainfed systems worldwide (Harrington and Tow 2011; FAO 2011b).

The categorization of farm systems developed for this review, building on the existing classifications, allows for a global analysis of interactions of farming with different types of wetlands. The main cross-cutting criteria are: (i) product category: crop, livestock, or fish; (ii) resource use and intensity, especially in terms of water and nutrients, as this determines to a large extent the direct interactions between agriculture and wetlands; and (iii) landscape factors, related to climate zones and geographical location (latitude, altitude). The resulting global categorization (Table 1) is based mostly on the first two criteria. The third criterium is considered in a number of the system types. The eight farming system categories are: crop systems, both rainfed and irrigated, on a scale from extensive to intensive (including horticulture); livestock systems; and aquaculture systems, all from extensive to intensive (see more detailed description in Supplementary Material).

Definition of farms

A farm or agricultural holding is the "economic unit of agricultural production under single management comprising all livestock kept and all land used wholly or partly for agricultural production purposes, ..." (FAO 1999). Farms are important because they represent the lowest scale-level at which decisions related to crops, livestock, resource use (land, water, inputs, labour) and farming practices are made, directly determining the possible impacts of farming on wetlands. The management can be exercised by an individual or household, or by a clan or tribe, or by a corporation, cooperative or government agency. Farms can be dedicated to one type of farming system, or can incorporate combinations of different systems. The total number of farms in the world was estimated at about 600 million (Lowder et al. 2016). Small farms of less than two hectares comprise more than 80% of these, but cover only 12% of the world's farmland. The largest 1% of farms operate more than 70% of the world’s farmland, and more than half of the world's farmland is operated by farms that are 100 ha or larger. There are strong regional differences. Most farms (70-80%) in East and South Asia, the Pacific, and Sub-Saharan Africa are small and cover 30-40% of the farmland, whereas in high-income countries small farms cover less than 10% of the farmland (Lowder et al. 2016; Anseeuw and Baldinelli 2020). On about 90% of the farms (75% of the farmland), labour is provided predominantly by members of one family. The other 10% of the farms, operating about a quarter of all the farmland globally, are held by corporations or other organizations (e.g. cooperatives, government or religious organizations; Lowder et al. 2016). In most low-income countries, the number of farms is increasing and farm size is decreasing. In high-income countries large farms are getting bigger (Anseeuw and Baldinelli 2020).

Table 1. Key characteristics of eight agricultural system categories; main sources: Wood and van Halsema (2008); FAO (2011a, 2011b); Lewandowski et al. (2018); IPBES (2018).

Agricultural system	Water use	Fertilizer use	Nutrient efficiency	Chemical use	Potential erosion	Agricultural diversity	Impact on biodiversity	Geographic location
a) Extensive rainfed	low, mainly for livestock	low-med, also organic	med-high in good practice	low-medium	low-med	med-high	low-med	close to high productive and arid areas
b) Intensive rainfed	low-med, processing of harvest, livestock	med-high	med-high, depends on practice	high	high	low	high	mainly temperate, lowlands
c) Intensive irrigated	high, irrigation and processing of harvest	high	often high	high	high	low	high	arid areas, basins, lowlands
d) Horticulture	high	high	high	high	low-med	low-med	med	areas with good water access, high productive regions
e) Extensive livestock	low	low indirect (fodder)	high (because of low input)	low or indirect	low-med	usually high	low	arid areas, mountain regions, only pastures feasible
f) Intensive livestock	high	high indirect (feed/fodder)	low-high, depends on practice	high indirect (fodder)	high - low, indoor	low	high	lowlands with good water availability
g) Extensive aquaculture	low	low	med-high	low	low	low	low-med	areas with good freshwater access; coastal areas
h) Intensive aquaculture	low/high (depends on system)	high, also indirect (feed)	low-high, depends on practice/ system	med	low	low	high	areas with good freshwater access and terrain for ponds; coastal areas

Direct drivers of wetland modification originating from agriculture

Classification of agricultural drivers

Here we focus on the negative change in wetland ecosystems resulting from agricultural land use change and activities. First the direct drivers are considered, defined as "natural or human-induced causes of biophysical changes at a local to regional scale" (van Asselen et al. 2013). In the Global Wetland Outlook four types of direct drivers of change in wetlands were distinguished: (i) structural change drivers, that alter wetlands and their immediate environment through permanent changes to the geomorphology, hydrology or vegetation (e.g. drainage, conversion, burning or removal of wetland vegetation); (ii) physical regime drivers: factors whose conditions and pattern of variation are altered by humans, related to changes in inflow quantity and frequency, sediment load, salinity and temperature; (iii) extraction drivers, that change wetlands by partial or complete removal of ecosystem components, such as water, species, and soil or peat; (iv) introduction drivers: addition of nutrients (fertilizers), chemicals (pesticides), invasive species, or solid waste; or atmospheric deposition (Ramsar Convention 2018a; van Dam et al. 2023). Agriculture causes change to wetlands in all of these four categories. In the following sections, the impacts of different agricultural systems on wetland types and water and nutrient flows will be discussed in relation to these driver categories. Where relevant, examples of specific wetland and agricultural systems are given.

Structural change drivers

The most common agricultural system types that lead to structural conversion of wetlands are rainfed agriculture, irrigated systems and horticulture. For cropping, structural conversion consists mainly of drainage, either sub-surface drainage using pipes or surface drainage by ditches and canals, and removal of the original wetland vegetation to plant crops. Sub-surface drainage has often been part of agricultural development, e.g. in north America (Zedler 2003). Drainage reduces the water table and soil moisture, aerates the soil and makes it more suitable for cultivation. Subsurface drainage generally increases the total water flow from fields, but reduces surface runoff of sediment, P and adsorbed materials on sloping terrains. Nitrate is very soluble and is transported well in drainage flows, much better than compounds with higher sorption capacity (e.g. phosphate, ammonium, organic N). The latter are retained better on clay or organic soils than on sandy or loamy soils. Cracked soils or road drains can cause completely different effects through preferential flow paths (Gramlich et al. 2018). The effects on aquatic ecosystems of agricultural drainage are direct loss or alteration of habitats (loss of connectivity, fragmentation, structure and function of wetland vegetation), effects on water quality, and effects on the hydrology (Blann et al. 2009).

Drainage and vegetation removal can be seasonal and, if they do not interfere permanently with the hydrology of the wetlands, may allow wetlands to flood again and vegetation to recover during parts of the year. This is common in African floodplains where traditional seasonal flood-recession farming depends strongly on rainfall. Near Lake Victoria, East Africa, papyrus wetlands are converted to farming during dry conditions and re-flood during the rainy season (Kipkemboi et al. 2007). Wetlands that are drained and converted to intensive rainfed farms are lost permanently, for example in the upper Namatala wetland in Uganda, where years of wetland farming have broken the lateral connectivity between river and floodplain (Namaalwa et al. 2013, 2020); and in other parts of Africa where large-scale commercial farms have appropriated wetlands (Kronenburg García et al. 2022). Extensive areas of New Zealand have also been lost due to permanent conversion (Robertson et al. 2019). Many irrigation systems are located in permanently converted floodplains, e.g. in Asia for lowland rice cultivation in floodplains. Many small-scale vegetable farms in Africa and Asia are located in or near wetlands. In northern Europe and North America, peat extraction for use as a growth medium in horticulture leads to permanent conversion of wetlands (Clarke and Rieley 2010). More permanent structural change is caused by filling in or conversion of isolated wetlands in upland plains (e.g. wheat or maize farming in the prairie pothole region of the USA and Canada) or to conversion of floodplain or delta wetlands (e.g. cereal and potato farming in western Europe). In the five upper midwestern states of the USA, 42-89% of wetlands were lost during the last two and a half centuries (Zedler 2003).

In principle, extensive livestock systems do not cause conversion of wetlands but with increasing stocking densities, drainage and removal of vegetation can occur. Burning of vegetation is used widely by pastoralists to improve the quality of forage for livestock or wildlife leading to loss of volatile substances to the atmosphere, deposition of ash (which can alter soil pH), exposure of the soil to solar radiation, and increased availability of nutrients (Kotze 2013). Under more intensive management, the original vegetation may be replaced by forage grasses and fertilization may be applied. In the Zoige wetland on the Tibetan Plateau, the traditional communal watering places did not interfere with the hydrology of the basin; land reform led to privatization of the land, fencing and draining of the wetlands, and digging of wells which caused a drop in the water table (Yan and Wu 2005). High livestock densities can also lead to trampling and soil compaction, which increases runoff and the potential for erosion, and reduces infiltration resulting in a lower groundwater table and negative effects on vegetation. Stock density also increases the intensity of grazing, which affects the structure and composition of the vegetation and ulitmately can reduce vegetation cover (USDA/NRCS 2003). Intensively managed pasture involves seeding of forage grass and legumes, which means replacement of the original vegetation. Where water logging occurs, drainage is applied. In the

Netherlands, where large areas of drained peatlands are used for intensive dairy farming, subsidence caused by low water tables and peat oxidation occur. Landless livestock systems do not create structural change in wetlands, unless they are constructed on converted wetlands, but indirectly can contribute to structural change if converted wetlands are used for the production of feed ingredients like soybeans and cereals (Barona et al. 2010; Nepstad et al. 2014).

Construction of aquaculture ponds in natural wetlands leads to habitat modification and loss. Generally, surface water is diverted from streams or rivers by gravity but water can also be pumped from wells. Ponds can be constructed in valley bottoms, by damming of headwater streams, in floodplain areas close to streams and rivers, or in deltas and coastal plains. For marine fish and shrimp, ponds are also constructed in mangroves or in reclaimed mudflats in tidal zones. Pond construction in mangroves led to loss of mangroves in southeast Asia and northern south America in the 1990s. About half of the mangrove area in the Philippines was converted to ponds between 1951 sand 1988 (Primavera 2006). In Indonesia, Brazil, India, Bangladesh, China, Thailand, Vietnam, and Ecuador 51.9% of the mangrove forest area was lost between 1970 (pre-shrimp) and 2004, and commercial aquaculture accounted for 28% of total mangrove loss (Hamilton 2013). Sometimes agricultural land is converted to shrimp ponds (Ilman et al. 2016; Jayanthi et al. 2018). The impact of pond construction on wetlands depends strongly on the scale of development. One or a few small ponds integrated with a farm will not create a big impact, but large-scale development or construction of hundreds of ponds in a floodplain strongly influences the hydrological and ecological functioning of the landscape.

The physical structure of coastal culture systems (fish pens or cages, poles, lines, rafts, anchor blocks) can degrade nearshore seagrass beds and sediment communities. Floating cages or pen systems for the culture of predominantly finfish (although other species can also be grown in cages) can be located in surface water of sufficient depth, depending on the size of the structure. Small-scale cage operations with a few cubic metres of cage volume can be located in a small lake, river, or in the coastal zone, whereas large scale cage farms operate cage systems that are up to 30 m wide and 10 m deep and can cover large surface areas in lakes, reservoirs, or large rivers, coastal bays, lagoons, and subtidal areas. Cage farms can also be located in the open sea or ocean, but are then technically (when in waters of > 6 m deep, because of the Ramsar definition) not in a wetland. Coastal mollusc (mostly bivalves) and aquatic plant (mostly seaweed) aquaculture are practiced in shallow coastal water, including coastal lagoons and intertidal and subtidal zones of estuaries. Molluscs can be grown on the bottom, in sheltered beds where the densities can be managed; or on ropes spiraled around poles or suspended from rafts or from floating long lines (Simenstad and Fresh 1995; McKindsey et al. 2011). Seaweeds are grown tied to lines that can be suspended from poles or from floating lines (Zhang et al. 2022). Coastal aquaculture influences the ecosystem by changing material flows (e.g. by taking up nutrients and converting them into biomass and waste products; or when the structures are colonized by species other than the cultured species) and through disturbances from the culture activities such as seeding, maintenance, or harvesting (Dumbauld et al. 2009; Spillias et al. 2023). Unregulated development of cage aquaculture, mollusc farms or seaweed farms can block the acces to other coastal resources (e.g. for coastal fishing communities) or create navigational hazards. In some places, structural materials are obtained by harvesting from natural wetlands (e.g. from mangroves; Primavera 2006).

Physical regime drivers

Physical regime changes for wetlands through diversion of water and sediment flows are particularly observed in areas with rainfed and irrigated systems (including horticulture). The connectivity and heterogeneity of wetlands are determined to a large extent by water flows, hydroperiod and water level and are seriously affected by fragmentation of the river and stream network (Fuller et al. 2015; Larkin 2018; Rasmussen et al. 2018; Middleton 2018; Rittenhouse and Peterman 2018). Regulation and construction of dams and reservoirs, interbasin diversion and water abstraction has affected 71% of the larger rivers in North America, Europe and the post-Soviet states (Nilsson et al. 2005; Lehner et al. 2011; Palmer et al. 2008; Hanna et al. 2018), with profound impacts on the ecological character of streams, rivers, floodplains, lakes, isolated wetlands and coastal ecosystems. Agriculture uses around 70% of global freshwater use and agriculture withdrawals continue to increase. Agricultural use ranges from 28 to 76 % of total water withdrawals in different regions (Figure 1). In large areas of Asia, northern Africa, Australia, and the Americas, agriculture intensification drives high water stress with consequences for people and wetlands (FAO 2020).

Changes in water flows and levels affect soil and biota directly. River regulation leads to a reduction in transition zones between aquatic and terrestrial zones, loss of surface water connectivity, and reduction of hydrological connectivity between river and floodplain to groundwater pathways, which affects isolated wetlands. Headwaters are affected most by the construction of dams, while lowland zones are damaged more by floodplain reclamation, channelisation and construction of barriers for navigation. This influences migration of aquatic organisms, both for species preferring flowing water and still-water (Buijse et al. 2005; Opperman et al. 2009; Tockner et al. 2010). Changes in soil moisture and inundation patterns drive changes in vegetation composition, sometimes providing opportunity for invasive species with far-reaching consequences for other plant and animal groups in the trophic network (Middleton and Kleinebecker 2012). Fish, macroinvertebrates, and amphibians are vulnerable to changes in flow (Maddock et al. 2013; Allen et al. 2020). Fish habitats for feeding and reproduction are degraded, and fish migration (e.g. upstream for spawning, and downstream for eggs and larvae) is obstructed (Pusey and Arthington 2003; Dugan et al. 2010), which also leads to significant impacts on fisheries production both in inland and coastal and estuarine systems (Gillson 2011). Changes in water flow also change the supply of mineral and organic sediment to wetlands (Galbraith et al. 2005). Changes in freshwater supply cause changes in tidal ranges and velocities in coastal wetlands, affecting erosion and saltwater migration that determine species composition and succession (Teal 2018; Tiner 2018), including the spawning area of diadromous fish species. With climate change, changes in freshwater and sediment supply are often combined with changes in sea level, e.g. in mangroves, where both landward and seaward extension can be the result (Asbridge et al. 2016).

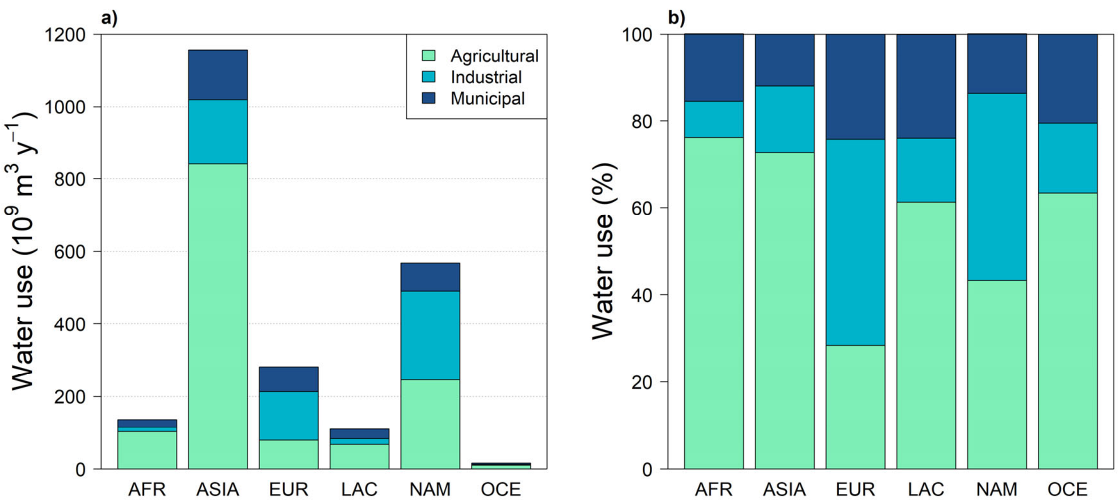

Figure 1. Agricultural, industrial and municipal use by world region for 2013-2018, as total use (a) and as a percentage of total use (b). Data source: AQUASTAT, https://www.fao.org/aquastat/en/databases/maindatabase, accessed 3 March 2021. AFR = Africa, EUR = Europe, LAC = Latin America and Caribbean, NAM = Northern America, OCE = Oceania.

In many crop systems, land preparation and farm practices lead to an increase in water and tillage erosion. Erosion is also possible from open field horticulture but is less from greenhouses, especially when crops are grown on trays and in artificial substrates. Sediment is supplied to wetlands through wind or water erosion, including natural bank erosion in streams and rivers. Erosion reaches peak values during periods of high rainfall and discharge or during intensive agricultural activity like land preparation or harvesting (Labrière et al. 2015), as demonstrated in wetlands converted to rice cultivation in Uganda (Namaalwa et al. 2020) and Rwanda (Uwimana et al. 2018a,b). In the Amazon region in Brazil, conversion of forest to pasture and soy farming can have indirect effects on wetlands through increases in erosion (Barona et al. 2010; Nepstad et al. 2014). In rangelands, erosion can stay undetected initially as it starts slowly, but then advances exponentially as the combination of soil compaction, reduced vegetation, increased runoff, exposure of the soil, enhanced decomposition of soil organic matter and the loss of fine soil particles all reinforce each other. After reaching a critical point, soil erosion cannot be reversed by reducing stocking densities but only by active restoration (USDA/NRCS 2003).

Erosion affects both inland and coastal wetlands. In streams and rivers, floodplains, lakes and forested wetlands, erosion upstream leads to water quality degradation and eutrophication (Allen 2004; MEA 2005), but also to siltation in reservoirs (Schleiss et al. 2016). Excessive addition of sediment changes lake character through modification of lake shore habitats, infilling or increased turbidity. In Lake Tana, Ethiopia, the delta of the Gumara River expanded by 5 ha annually between 1984 and 2014 as a result of agriculture and settlements in the Gumara catchment leading to increasing sediment concentrations in the river (Abate et al. 2017). Sensitive coastal wetlands like seagrass beds and coral reefs are also affected by erosion (Fabricius 2005; Wenger et al. 2015). Shallow marine waters, seagrass beds and kelp forests can be substantially degraded by excessive sediment from storm-event erosion (Rabalais et al. 2010; UNEP 2014).

Salinization is caused by the accumulation of soluble salts. Agricultural drainage water can have elevated salt concentrations as it flushes out minerals from soils. Intensive water abstraction for irrigation can cause elevated water table of saline or brackish groundwater, and in combination with high evaporation rates lead to salinization of inland wetlands in arid and semi-arid regions (Gell and Reid 2014; Herbert et al. 2015). Similarly, surface or groundwater abstractions for irrigation in coastal wetlands can lead to intrusion of salt surface or groundwater (EEA 2009; Herbert et al. 2015; White and Kaplan 2017). Salinity changes can alter the ecological character of wetlands by changing the composition of microbial communities and their biogeochemical processes, and by influencing the solubility of gases and redox potential of soils and the osmotic regulation of organisms. This affects phosphorus adsorption, denitrification and other ecological functions. Prolonged salinization can lead to replacement of freshwater by salt-tolerant or brackish communities (Herbert et al. 2015; Zhou et al. 2017) and contamination of water supplies used for agriculture or human consumption.

Dam construction for aquaculture ponds in natural catchments, especially when it replaces valley bottom or floodplain wetlands, has multiple impacts on the surface and subsurface flows of water, with implications for the functioning of streams, water and sediment retention, particulate organic matter production (e.g. from phytoplankton), water chemistry, and migration of aquatic species. While these are generally seen as negative, in agricultural catchments dams and fishponds can serve as traps of sediment and suspended matter, and have a positive impact on nutrient retention (Four et al. 2017; Uwimana et al. 2018b). The physical structures of coastal aquaculture can modify current regimes which has effects on a number of ecosystem processes, including reduced exchange rates and increased sedimentation (Dumbauld et al. 2009).

Introduction drivers

While some farming systems, particularly extensive farms and small-scale subsistence systems have no or only low levels of chemical inputs (fertilizers, pesticides), other forms of more intensive agriculture with higher input levels result in more leaching of nutrients and chemicals and contamination of surface- and groundwater. Globally, the application of N and P fertilizers and pesticides have increased since the 1960, but there are strong regional differences both in terms of total use and in application rates per ha (Figures 2 and 3). Horticultural systems and especially greenhouses are often associated with releases of nutrient-enriched waters and leaching of pesticides, but much depends on the way these systems are operated (Bergstrand 2010). Open systems discard the nutrient solutions used, but increasingly closed recirculating systems are used which collect and recycle the nutrient solution to the crops (although groundwater contamination remains a risk if systems are not completely closed).

In extensive livestock systems, manure is returned to the landscape but as this is produced from fodder growing in the same system, it does not represent a nutrient import on a catchment scale. Locally, the concentration of livestock at watering places can create accumulation of nutrients and organic matter, leading to water quality problems and eutrophication (Malan et al. 2018) and the risk of microbial contamination (Khaleel et al. 1980; Bicudo and Goyal 2003; Garzio-Hadzick et al. 2010). Intensive pasture requires fertilization to produce enough forage. Manure and fertilizer applications lead to nutrient surpluses in intensive dairy farms. In Australia and New Zealand, median annual losses were 27 kg N ha^{-1} (range 3-153) and 1.6 kg P ha^{-1} (range 0.3-69), depending on landscape conditions and management (McDowell et al. 2017). Generally, intensification of livestock systems and the de-coupling of livestock and crop systems lead to increasing loads of nutrient on the environment, including nitrate leaching to groundwater (Menzi et al. 2010; Sahoo et al. 2016). Intensive landless systems (pigs, poultry, cattle) are relatively independent of landscape or climate features, and depend more on the vicinity of input and consumer markets, and the availability of transporting and processing infrastructure. For pigs and poultry, they are often indoor systems. Feedlots or feed yards are intensive operations in which animals (often beef cattle) are kept in enclosures in high densities and fed high-protein diets consisting of forage, grains, minerals and supplements. Manure produced in feedlot systems is often recycled to the fields used to produce the feed, sometimes mixed with bedding material (straw, or other products). Because of concentrated manure production, intensive livestock systems can cause eutrophication problems in surface water, including contributing to eutrophication of coastal wetlands. Leaching of nutrients (especially nitrate) to groundwater is widely reported (Sahoo et al. 2016).

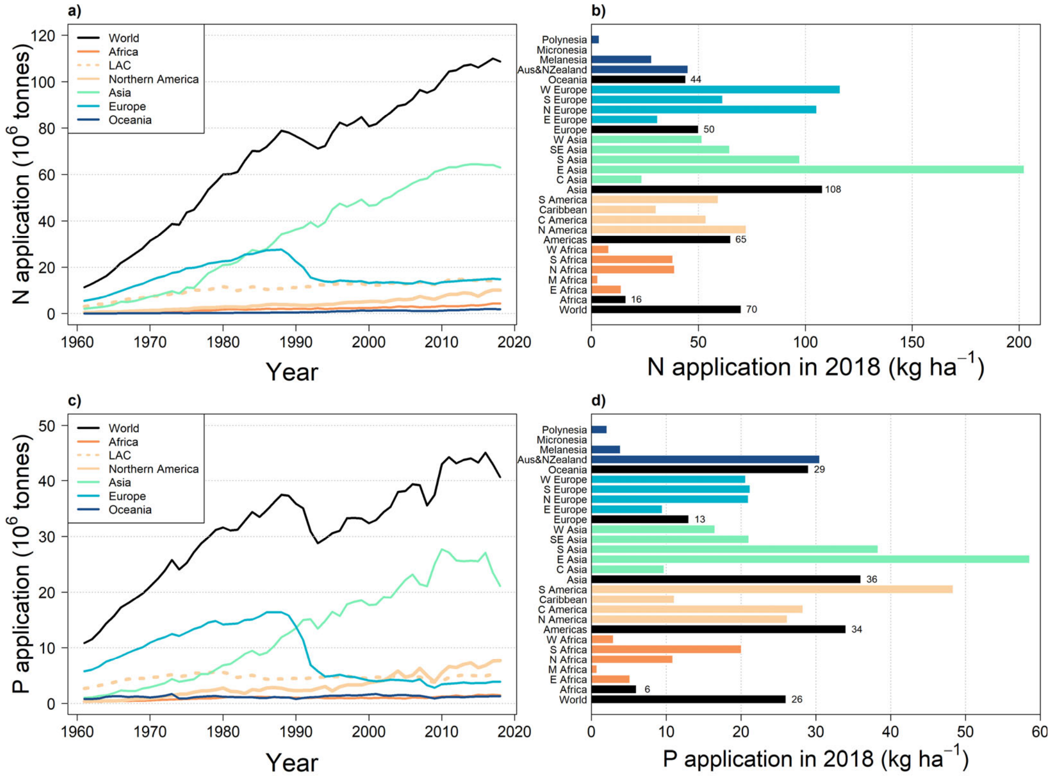

Figure 2. a) Total nitrogen (N) application by world region between 1990 and 2018; b) Total N application rate (kg ha^{-1}) by world regions and sub-regions in 2018; c) total phophorus (P) application by world region between 1990 and 2018; d) total P application rate (kg ha^{-1}) by world regions and sub-regions in 2018 . Data source: FAOSTAT, http://www.fao.org/faostat/en/#data, accessed 21 February 2021.

Semi-intensive and intensive aquaculture ponds need nutrient inputs in the form of manure, fertilizers or feeds, but a large part of the non-assimilated nutrients accumulate in the pond sediments (Hargreaves 1998) and are released at the end of the culture period. Intensive tank systems do not have natural sediments and release all of their nutrient wastes with the discharge water if no wastewater treatment is applied. Fish cages discharge waste feed, metabolites of the fish (mainly ammonium) and fish faeces into the water of the bay or lake. Nutrients and organic loading can become problematic when the carrying capacity of the ecosystem is exceeded or when cages are not properly sited. In Bolinao Bay in the Philippines, algal blooms and fish kills were observed as a result of the high density of milkfish aquaculture cages (1170 units) that far exceeded the planned number (544 units; Geček and Legović 2010). Expansion of cage culture is also observed in lakes in Africa, e.g. Lake Volta in Ghana (Asmah et al. 2014).

In coastal aquaculture, biodeposition from cultured molluscs and other organisms on the structures increases the organic loading, and therefore influences the oxygen levels, pH, and redox potential of the sediment. It can also affect benthic respiration and nutrient fluxes, and benthic infaunal communities. Because of the limited use of external inputs, seaweed and mollusc culture in coastal systems hardly increase the nutrient concentrations of the water. Mollusc culture can reduce nutrient concentrations even to the point where phytoplankton productivity is reduced, sometimes to levels that affect other parts of the ecosystem (McKindsey et al. 2011).

Catchment studies have demonstrated the impact of agricultural land use on the export of nutrients from catchments. In the Gjern river basin in Denmark, 76% and 51% of the N and P export, respectively, were from agricultural areas in the basin. After restoration of riparian zones and a lake, N and P retention increased because of permanent N removal through increased denitrification and increased P removal through sedimentation in the floodplains (Kronvang et al. 1999). Dissolved nutrients, especially nitrate, can infiltrate and leach to the groundwater. P binds more strongly to sediment particles and is more likely transported via surface runoff. Nutrient export is also related strongly to discharge. Drainage of prairie pothole wetlands in Canada and the USA led to increased loads of nutrients and organic matter to downstream areas (Brunet and Westbrook 2012). Nitrate export in the catchment of Chesapeake Bay (USA) was related strongly to the baseflow, whereas organic N export was more related to surface runoff and P export was related to the concentration of suspended solids (Jordan et al. 1997). The results of nutrient export and increases in nutrient concentrations in lakes and coastal areas are well known: excessive growth of phytoplankton and macrophytes, loss of oxygen, dead zones, fish kills, loss of biodiversity, loss of aquatic plant beds and coral reefs (Turner and Rabalais 1991; Carpenter et al. 1998; Goolsby et al. 2000; Rabalais et al. 2002; Magner et al. 2004).

Fertilizer and manure use also lead to N emissions to the atmosphere, which return to the surface of the earth, including wetlands, through deposition as nitrous oxides or ammonia (Hristov et al. 2011). Atmospheric ammonia concentrations are significantly elevated over the world's major agricultural areas, particularly in North America, Western Europe, South Asia and East Asia (Warner et al. 2017). Wet atmospheric N deposition in China was significantly correlated with chemical fertilizer use (Zhu et al. 2016). N lost through ammonia volatilization from synthetic fertilizers and animal manure can be as high as 18 and 26%, respectively (Bouwman et al. 2002). Application of effluents and slurries from intensive livestock systems, and composting of manure also lead to ammonia volatilization (Saggar et al. 2010). The effects of N deposition on wetlands can include a direct toxicity effect, accumulation of N in the ecosystem, longer-term effects of reduced N forms, acidification of soil, and increased susceptibility to changes in environmental conditions or diseases (Bobbink et al. 2010). In upland catchments, high N deposition often leads to leaching of nitrate to surface waters. High N loads can exceed the natural absorption capacity of *Sphagnum* mosses, resulting in leaching of N through the moss layer and increased availability of N to vascular plants (Fritz et al. 2011). Decomposition of organic matter increases with N inputs by reducing the natural N limitation on microbial decomposition processes (Bragazza et al. 2006; Song et al. 2013). High deposition in lakes in

Europe and the US shifted phytoplankton growth from N to P limitation (Bergström and Jansson 2006; Elser et al. 2009; Crowley et al. 2012) and led to elevated denitrification and N_2O emissions (McCrackin and Elser 2010). N deposition in coastal waters leads to harmful algal blooms, e.g. in the South Yellow Sea and East China Sea (Liu et al. 2011).

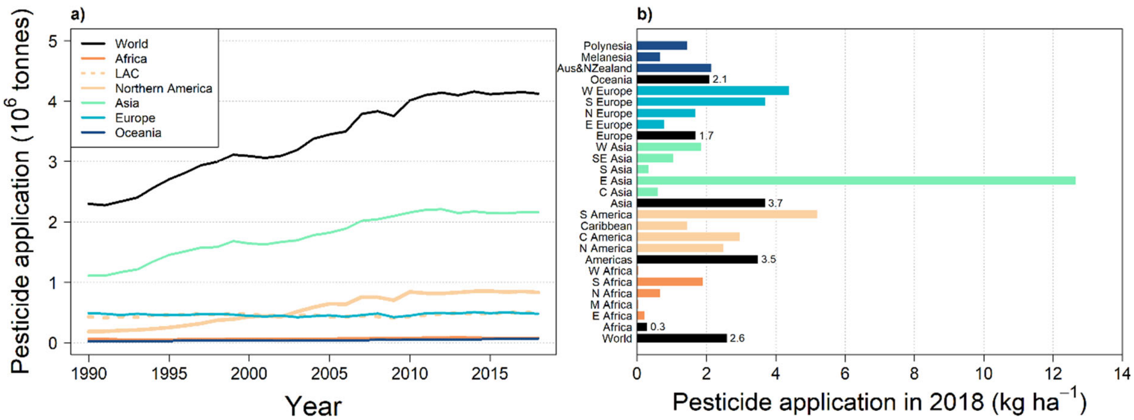

Figure 3. a) Total pesticide application (insecticides, herbicides and fungicides) by world region between 1990 and 2018; b) total pesticide application rate (kg ha^{-1}) by world regions and sub-regions in 2018. Data source: FAOSTAT, http://www.fao.org/faostat/en/#data, accessed 21 February 2021.

Pesticide loads from agricultural land have an impact on aquatic systems and have increased substantially since the 1960s, particularly in Asia (Figure 3). However, the actual impact on wetlands depends on many factors such as the chemical used and its persistence, sorption to soil particles, topography and climate conditions, and distance to the wetland. In catchments of the Great Barrier Reef (GBR) in Australia, the use of pesticides for sugarcane and cereals is common. In the GBR water, insecticides are rarely detected but herbicides were present (Thorburn et al. 2013). In wetlands and floodplains of the Mississippi river (USA), a range of pesticide residues were detected (e.g. Cooper et al. 2003; Evelsizer and Skopec 2018; Callicott and Hooper-Bùi 2019). Horticulture is responsible for a disproportionate share of the global pesticide consumption. The production of fruits and vegetables accounts for 26-28% of total pesticide use. In the USA, fruit and vegetable production uses 2% of the agricultural land but 14% of total pesticide consumption (Weinberger and Lumpkin 2007). In the Netherlands, pesticide use on cropland generally was around 7 kg/ha in 2008, but much higher in the culture of ornamental plants (up to around 100 kg/ha; CBS 2014). In Thailand, the largest amounts of pesticides were applied in intensive vegetable culture with up to 43 kg of active ingredient per ha (Schreinemachers et al. 2012). In a horticultural area near Shenzhen City in China, pesticide application rates (mostly organophosphate insecticides) were 300 kg per ha (Zhang et al. 2015). Surface water in catchments with horticultural land use often contains pesticides and degradation products of pesticides (e.g., Wightwick et al. 2012; Jansen and Harmsen 2011). Often, concentrations do not exceed water quality standards, but persistent pesticides (e.g. HCH (hexachlorocyclohexane) and DDT) that were banned years ago are also found (Zhang et al. 2015).

In intensive livestock systems and aquaculture, the use of veterinary pharmaceuticals is common. Antibiotics, antiparasitic drugs and steroidal hormones are the most widely used. Introduction to the environment happens mostly through the treatment of farm animals and fish, inappropriate disposal of used containers or unused medicine, and in the production process, and leads to intoxication or mortality, or to development of resistance to antibiotics or disturbance of hormonal functions (endocrine disruptors). Excessive use of antibiotics in aquaculture has led to antibiotics resistance in bacteria from fish and shrimp ponds. Detection of pesticides and antibiotics in consumer products has occasionally led to public health concerns (Primavera 2006). Generally, more knowledge on the environmental impact of veterinary pharmaceuticals is needed, especially on the combined effects of several compounds (Bártíková et al. 2016). Some substances may affect the microbial composition of soils, inhibit soil bacteria growth, reduce the hyphe length in active molds, or affect sulfate reduction in soils and inhibit the decomposition of organic matter (Boxall et al. 2003).

Fish escapes from aquaculture systems to the environment happen but are a bigger risk in cage than in pond aquaculture. Genetic introgression of farmed fish in wild populations and effects of parasites and viral diseases on wild populations are hazards related to cage farming. Research on salmon farming in Norway and on sea bass and sea bream farming in the Mediterranean showed that the risk of nutrient and organic loading was low but the impacts of introgression in wild populations and parasites and diseases needed more research (Arechavala-Lopez et al. 2013; Taranger et al. 2015). While seaweed farming generally does not introduce nutrients into the system, it has introduced non-native species but its role in the spread of invasive seaweed species is not clear (Eggertsen and Halling 2021; Spillias et al. 2023). Mollusc culture can also be a source of introduction of exotic species (McKindsey et al. 2007).

Extraction drivers

Extraction of vegetation, fish or other biota from wetlands is common in extensive farming systems where farm households are often engaged in diverse livelihoods activities. Vegetation can be used as fodder to feed animals, but also for a wide range of other purposes including mulching, construction, household utensils, handicrafts, furniture and others (Wood et al. 2013; Biggs et al. 2015). Supplemental irrigation with water from wetlands is applied to rainfed crops to improve yields when rainfall does not provide sufficient moisture, and can improve crop yield and water productivity. With increasing drought episodes in recent years, supplemental irrigation with groundwater or surface water may have impacts on downstream wetlands (Nangia and Oweis 2016).

The extraction of peat for use as a growth medium, mostly as potting soil for container culture or as an amendment for lawn and garden soils, represents a direct impact of horticulture on peat wetlands. Most popular is the incompletely decomposed peat of *Sphagnum* sp. (white peat), but other types like sedge peat (Blievernicht et al. 2011; USGS 2013) are also used. Peat is uniquely suitable for this because of its ability to retain water, air and plant nutrients, its low pH and the option to mix it with special purpose growth media (Amha et al. 2010). The main regions for horticultural peat extraction are northern Europe (about 80% of global production) and North America (Canada and USA, with about 5% of production). On a global scale, about 2,000 km^2 of peatlands are used for extracting horticultural peat (Clarke and Rieley 2010). The effects of peat extraction include damage to wildlife habitats, destruction of the hydrological properties of the bogs, a reduction of the carbon storage function, and the release of the carbon contained in the peat (Alexander et al. 2008; Kern et al. 2017; Leifeld et al. 2019). The demand for peat substrates in horticulture is still increasing. Efforts to reduce its use include less destructive extraction methods, the search for alternative materials, the designation of peat wetlands as protected areas, and raising awareness among actors about the values of intact peat wetlands (Waddington et al. 2009; Verhagen et al. 2013). Non-decomposed *Sphagnum* farmed on re-wetted degraded peat bogs (Gaudig et al. 2014) or on artificial floating mats (Blievernicht et al. 2011) may be a promising alternative.

Wetland vegetation is used widely as natural fodder in extensive pastoral farming. Many different types of wetlands can occur in rangelands, including isolated depressional wetlands, permafrost peat wetlands and alpine wetlands in cold climate zones, streams and rivers and their riparian zones, or sand rivers with mainly sub-surface flow. Ecosystem services of rangeland wetlands can include capturing surface runoff, storing water, groundwater recharge and discharge, sediment retention and improvement of water quality of runoff to streams and rivers, denitrification, and serving as habitat and nursery for plants and animals, production of hay and native plant seeds, livestock production, hunting and fishing, bird watching, and recreation (e.g. canoeing or kayaking; Johnson 2019). Some of the rangeland areas are among the most species-rich areas of the world (Alkemade et al. 2013). Many small-scale dairy farms are located in upland areas where they can affect headwater wetlands. Where livestock herding intensifies, the natural rangeland vegetation may be removed and replaced by managed pasture, and water may be extracted from surface or groundwater for drinking water or for pasture irrigation in dry periods. Generally, intensive pasture systems for feeding dairy cattle are in areas that are less suitable for other crops. In the temperate regions of the world, these are often river or coastal floodplains with a high water table. However, there are also many small-scale dairy farms in upland areas. In many places, wetlands support livelihoods related to livestock, e.g. in the Sudd wetlands of South Sudan, where most of the wetland communities combine livestock with other livelihoods activities such as small-scale farming, fishing and hunting (Ojok 1996; Rebelo et al. 2012).

For aquaculture species for which hatchery reproduction methods are not available, juveniles for stocking ponds, cages or shellfish farms must be obtained from wild catch. For example, in Bangladesh there is a seasonal fishery for eggs of the Indian major carps in Halda River, which are then traded to pond operators for growout (Alam et al. 2013). For the cultivation of eel species (*Anguilla sp.*), glass eels are caught and traded internationally. Because of the decline in stocks, trade in the European eel (*Anguilla anguilla*) is now prohibited under CITES (Kaifu et al. 2019). For most freshwater species, and increasingly for marine species hatchery techniques for artificial reproduction are available.

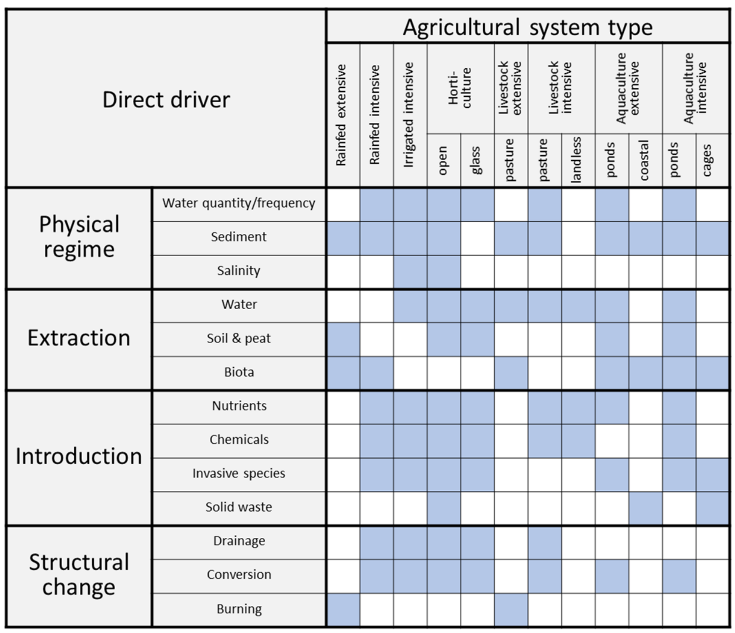

Figure 4. Direct drivers of change in wetlands originating from different agricultural systems. Blue cells indicate impact of the agricultural system type on the driver type in the row. Intensity or scale of impact are not indicated as these are strongly local context-specific.

Wetland types and functions affected by agricultural systems

By juxtaposing the different wetland types and agricultural systems, the impacts of agriculture on wetlands can be unpacked. A differentiated analysis results from looking at the impact on wetlands of the direct drivers of change originating from the various agricultural systems (Figure 4). All types of farming create some impact on wetlands, but intensive farming systems (intensive crop and livestock systems, including horticulture) have, unsurprisingly, the strongest and broadest impact through their water and soil management, and the application of nutrients (fertilizer), chemicals (pesticides) and sometimes invasive species. For extensive systems, impacts on soil, vegetation and other biota are important but fertilizer and pesticide applications are generally lower. Most inland wetland types are affected by agriculture, either directly (by wetland conversion) or indirectly through modification of water, sediment, and nutrient flows of catchments. Coastal wetlands are affected mostly indirectly, except by coastal aquaculture which can have a direct impact.

Indirect agricultural drivers of wetland modification

Indirect drivers are the broader, more diffuse mechanisms and processes that influence the direct agricultural drivers of wetland change (MEA 2003; Ramsar Convention 2018a). They are related to the demographic, socio-economic, institutional and cultural processes that influence the decision-making of all actors. Wetlands are thus elements within food systems, which comprise not only agricultural production but all activities from food production to food consumption (including processing, packaging, distribution, and retailing), and the outcomes in terms of food security, environmental security, and social welfare (Ericksen 2008; FAO 2018d). Within food systems, agricultural products are components of value chains which comprise the steps through which a product goes from production, processing, delivery to consumers, and disposal of waste products (Kindervater et al. 2018). The development from traditional to modern food systems usually comes with replacement of small, diverse farms by larger monoculture farms, with short local supply chains becoming longer (sometimes global), and with change from societies with under-nutrition to the prevalence of dietary diseases and obesity (Ericksen 2008). Since 1970, the global food system has shown a growth in agricultural productivity through technological innovation, globalization of markets and trade, and increasing awareness of the environmental implications of food production (Brooks and Place 2019). Besides farmers, a range of other actors in the value chains and food systems play a role in determining what and how farms produce food and other products (Table 2). Farmers, agribusiness companies, and financial institutions are active primarily with the production of agricultural products and the necessary inputs. Consumers, and food processing and retail companies determine the demand for food and the product markets. Finally, governments and non-governmental organizations play a moderating role in the governance of food systems.

On the production side, the indirect drivers are related to the decision-making of farmers about the choice of crops or livestock, the farming system and technologies applied and the use of resources and inputs, and include economic and social factors, socio-cultural beliefs and norms, and the institutional environment. The decisions of farmers and farming households are strongly related to how they perceive the natural environment (including climate) and the available resources, the economic risk involved and the institutional support they can expect, but also to the perception of their own role as farmers and how they adapt to change (McGuire et al. 2013; Singh et al. 2016;

Emerton and Snyder 2018). In low and middle-income countries, many small-scale farmers produce at subsistence level or for local markets. With increasing production for markets, farmers can invest in intensification of production through more inputs (e.g. fertilizers, pest control), better genetic material (e.g. improved seeds) and technology (irrigation, mechanization). To achieve this, sufficient natural resources (land with good soil quality and sufficient water) and an enabling environment (e.g. credit, extension services) are also needed.

Table 2. Actors in the global food system.

Actor	Description
Farmers / producers	More than 80% of farms globally is 2 ha or smaller but occupy only 12% of farmed land. More than half of global farmland is with farms > 100 ha.
Agribusiness companies	Produce agricultural chemicals, fertilizers, seeds and farm machinery. Dominated by three multinational megacompanies who control 70% of the industry.
Financial institutions	Commercial banks, private and institutional investors, international and agricultural development banks, insurance companies, cooperatives, credit unions.
Consumers	Depending on income, buying from local (super) markets, restaurants. Sometimes organize to advocate for food safety or other issues.
Food processing and retail companies	Limited number of multinational food and beverage corporations responsible for processing and retailing food. Two thirds of global food trade consists of processed products.
Knowledge and research institutions	Universities and national and international (agricultural) research institutes.
National governments	Organized in policy sectors (agriculture, energy, environment etc.). Uses policy and economic instruments to implement policies.
Multilateral/international agencies and NGOs	Includes e.g. UN organizations (e.g. Food and Agricultural Organization FAO), environmental agreements (including Ramsar Convention), international cooperation and development community, but also international trade agreements such as World Trade Organization.

Other important actors for food production are the agri-business companies that provide agricultural inputs (chemicals, fertilizers, seeds) and farm machinery to farmers. In recent years several global level mergers have resulted in consolidation, with three trans-national megacompanies now controlling some 70 percent of the agrochemical industry and over 60 percent of commercial seeds. These companies also dominate the research, spending the equivalent of 75% of all private sector research in this sector (Fuglie et al. 2011). This raises concerns about the lack of competition, companies claiming patents and ownership of gene traits and germplasm, and research defending existing positions in chemical-intensive agriculture and genetic engineering, rather than exploring innovations that are better for consumers and the environment (Howard 2009). The large agrobusiness companies also spend millions on lobbying programmes, e.g. with law makers in the US and in Europe (Elsheikh and Ayazi 2018). The large countries with developing agricultural sectors such as Brazil, India and China have been the growth markets for agribusiness companies in recent years (Phillips 2019).

Land ownership and tenure are strong determinants of the enabling environment for food production. Tenure types range from formal statutory forms, where the state owns the land, to cooperative or customary land tenure where land is owned by local or indigenous communities who manage the land according to their customs and traditions. Changes from traditional land tenure, in which wetlands are often managed as common pool resources, to private ownership can lead to unsustainable use (Adger and Luttrell 2000). On the other hand, many small farm households do not have secure tenure, which influences the decision-making about sustainable farm practices and land management (UNCCD 2017). In some parts of the world, local communities that managed the (wet)lands under a communal or customary arrangement are displaced by powerful outsiders who are allowed to buy land ('land grabbing'; see e.g. Kronenburg García et al. 2022). Other elements of the enabling environment include extension support for farmers, and credit facilities or subsidies that allow farmers to obtain loans for investing in good quality inputs and farm mechanization (Teng et al. 2016; Thierfelder et al. 2018; Terlau et al. 2019).

On the demand side, consumers determine the demand for food items and the products derived from these. However, consumer demand is influenced strongly by the food processing and retail industries. The global food and beverage industry is worth more than 10% of the world's GDP, and is also dominated by a small number of large transnational corporations. Similarly, the retail industry is dominated by three to five companies that control more than half (often 75% or more) of the market (Burch et al. 2013). These large and powerful companies influence markets and price levels, consumption patterns (e.g. through advertising of processed foods) and production standards and methods. Increasingly, they impose private production standards which can have negative effects on small producers who struggle to comply (Richards et al. 2013). Because of liberalization in foreign direct investment, supermarkets have also increased their market share in low and middle income countries, resulting at times in competition with traditional retailers and exclusion of small local suppliers (Reardon and Hopkins 2006; Swinnen and Vandeplas 2010). Demand is also determined by dietary patterns, which have changed during the 20th century as a result of economic development and urbanization, from largely home or artesanal food preparation using unprocessed ingredients to a situation with two thirds of the food trade consisting of processed products (Moubarac et al. 2014; Brooks and Place 2019). Convenience foods gradually displace long-established traditional foods and dietary patterns that were presumably more healthy; this happens increasingly also in low and middle-income countries, where the growing urban population and rising incomes lead to an increase in consumption of animal products and refined carbohydrates (Monteiro and Cannon 2012; Stuckler et al. 2012; Monteiro et al. 2013; Tilman and Clark 2014).

The formal governance of food systems is the remit of national governments who have traditionally dealt with food security and safety, economic development and environmental policy. Governments can influence the decision-making of actors in three ways: by using economic instruments to intervene in markets; by setting rules (laws and regulation); and by influencing the awareness and attitude of actors (e.g. using education and communication, or by stimulating stakeholder participation) (ten Brink and Russi 2018; ten Brink et al. 2018). In most government systems food and environment are confined to separate policies and implementing agencies. National agricultural policies aim at food security, and government interventions often focus on correcting failures of the market that lead to unwanted price fluctuations or food shortages. Market failures are also targeted by environmental policies, where the lack of monetary valuation of e.g. wetland benefits leads to disregarding these benefits in decision-making.

Economic instruments include subsidies and other incentives that influence the decision-making of actors by changing the price levels of inputs or products. In agricultural policy, subsidies make it more attractive for farmers to grow certain crops or change their ability to compete with other producers, and the development of subsidy initiatives may not consider environmental outcomes, such as potential for wetland modification. Subsidies can also be used to reduce risks for producers, e.g. with new production methods or technologies (Cong and Brady 2012; Bruckner 2016). Minimum prices (often accompanied by import tariffs) can guarantee the income of farmers, and export subsidies make it easier to sell products on the world market. Another form of agricultural incentive are input subsidy programmes (ISPs), such as fertilizer subsidies that support agricultural development but can also contribute to the overuse of fertilizers (Li et al. 2013; Huang et al. 2015). The potential of fertilizer subsidies is debated because the effectiveness of fertilizers also depends on water availability and other inputs (Jayne et al. 2018). Subsidy programmes, sometimes as part of of international cooperation programmes, are also used to support the development of aquaculture (e.g. in India and in the European Union) and agriculture (e.g. horticulture in eastern Africa) in wetlands.

Agricultural subsidies are often asssumed to be harmful, but the effects on wetlands and their biodiversity are difficult to trace through the complex networks of actors, value chains and markets. Harmful subsidies are defined as "government action that confers an advantage to consumers or producers ….. but in doing so, discriminates against sound environmental practices" (Withana et al. 2012; Oosterhuis and ten Brink 2014). Besides subsidies, this also includes waiving certain taxes or ineffective implementation that tacitly allows illegal activities (Withana et al. 2012; Dempsey et al. 2020). In some cases, governments have actively cut subsidy programmes for the benefit of the environment, e.g. the reduction of subsidies for chemical fertilizers in Bangladesh and France, and elimination of subsidies for wetland draining in Denmark (Dempsey et al. 2020). Generally, the amounts spent on stimulating environmentally friendly practices are small compared to the amounts spent on stimulating harmful practices (Withana et al. 2012). The lack of coordination between policy sectors, economic and political interests in supporting certain sectors, and the strong lobbies from the corporate sector are all obstacles for the reform of agricultural subsidy programmes.

Economic instruments can stimulate desired behaviour, or discourage undesired behaviour. They can also be used to correct market failures, e.g. by incorporating the costs of environmental damage in the price of goods or services, or by rewarding actors for protecting or maintaining wetland ecosystem services (ten Brink and Mazza 2013). This can take several forms: taxes or fines on environmentally damaging activities or the use of resources, or auctioning of permits to pollute or exploit a resource. Specifically for wetland conservation, tools like payment for environmental services (PES), certification schemes, offsets and mitigation banking seem to offer potential (Kinzig et al. 2011; ten Brink and Russi 2018). For example, the aquaculture stewardship council (https://www.asc-aqua.org/) certifies environmentally and socially responsible seafood. Mitigation banking, common in the USA but also occurring in Germany and Australia, is a system that issues credits for conservation, restoration, or creation of wetlands that can be purchased by developers to mitigate wetland loss caused by their project (Bennett 2018; Brownlie 2018).

Legal, regulatory instruments regulate human activities including land and water use practices. In agricultural policy, they are used mostly to deal with human or animal health concerns, such as food safety risks related to agricultural chemicals or pharmaceuticals, or to animal diseases and welfare. The use of certain pesticides can be banned or bound to strict rules. At an international level, there are the International Code of Conduct on the Distribution and Use of Pesticides and the London Guidelines for the Exchange of Information on Chemicals in International Trade. Since 1998, the Rotterdam Convention creates legally binding obligations for the implementation of the Prior Informed Consent (PIC) procedure which aims at promoting responsible international trade and environmentally sound use of hazardous chemicals. Environmental policy encompasses a wide range of regulatory instruments, such as land use planning, emission standards, production standards, best practices, permits/licences, property/access/exclusion rights, integrated landscape management, and ecosystem restoration (ten Brink et al. 2018). The efficiency and effectiveness of the regulations and of a "command and control" regime depend strongly on the ability of governments to implement policy measures effectively. Without sufficient capacity, governments struggle with enforcement of rules, monitoring compliance and outcomes, and reporting (Blackman et al. 2018).

Policy instruments related to communication and participation represent a more long-term approach to influencing the awareness and perceptions of actors. This relies on building social capital and informal governance processes through information and education programmes directed at different target groups. It can also involve participation of stakeholders in the preparation and planning of policies or in decision-making. In practice, economic incentives, regulation, and communication and participation are strongly related and cannot be separated from each other. Subsidy programmes are often accompanied by legal arrangements to ensure correct and fair implementation. Increased awareness among stakeholders enhances the chances of success of economic and regulatory measures. Examples include the National Ramsar Committees or Wetland Fora in many countries that bring together a variety of stakeholders to discuss wetland policy; and World Wetlands Day. An initiative aimed specifically at wetlands and agriculture is the Roundtable on Sustainable Palm Oil (RSPO) which brings together industry stakeholders (producers, processors, traders, consumer goods manufacturers, retailers, banks/investors, NGOs) to develop and implement global standards for sustainable palm oil, including (https://rspo.org). Another example is the Peat Working Group (PWG) in Great Britain that makes recommendations for future policy and best practices in peatlands (Alexander et al. 2008).

Discussion: towards more sustainable, wetland-friendly food production

The relationship between wetlands and food production has a dual nature: agriculture is a major driver of wetland loss and degradation (the focus of the review presented here), but also depends strongly on healthy inland waters and wetlands (Ringler et al. 2022). The analysis shows that to understand the pathways and mechanisms of agriculture-wetland interactions (e.g., water and nutrient flows, pesticide use, material extraction) it is necessary to differentiate between different farming systems and wetland types. Even the eight different farming system types presented here do not provide a sufficiently detailed analysis of the impacts to identify sustainable management options in a decision-making setting. More detailed, local context-specific analyses are needed for specific crop or livestock systems and wetland types and how they respond to changes in technology, markets, institutions or other drivers of change. This is a challenge for managers and policy-makers: only a thorough understanding of wetlands and food production systems in the local biophysical, socio-economic and institutional context can point the way to sustainable forms of food production with outcomes that are acceptable to both ecosystems and people.

In recent years, awareness of the challenges related to the current global food system has increased, with concerns about environmental sustainability, inclusivity and equity, human health and nutrition, climate impact, and vulnerability and resilience of livelihoods and ecosystems to extreme events and market shocks; as a result, a consensus is now emerging about the need for some form of transformative change in the way we produce, process, and market food (Willet et al. 2019; Webb et al. 2020) and for the benefit of wetlands (Convention on Wetlands 2022). During COP28 of the UN Climate Change Conference, 134 countries signed a declaration affirming that "agriculture and food systems must urgently adapt and transform in order to respond to the imperatives of climate change" (UN Climate Change Conference 2023). The increase in food production and in the productivity of agricultural systems since the 1950s was based on a productionist, food security paradigm, which emphasizes the need to apply technologies to intensify food production, stimulate economic growth, and process and distribute food through free global markets and trade. Thinking is now shifting towards a food sovereignty paradigm, which is more concerned with agricultural landscapes as social-ecological systems, and with the decision-making process of different actors in the food system, environmental sustainability, cultural diversity and regional markets (Altieri 2002; Davila and Dyball 2017; Ruben et al. 2021). Research on the changes that are needed in food systems suggests that besides technological innovation and a more equitable wealth distribution, it is particularly collaboration, collective action and changes in governance that are needed to facilitate transformation (van Bers et al. 2016 2019; FAO 2022).

We expect that such a transformation will not only have positive effects on wetlands but will also benefit from paying more attention to wetland integrity. To support this expectation, we can consider the implications for wetlands of policy frameworks for sustainable food systems, such as the five principles for sustainable agriculture presented by the Food and Agriculture Organization of the UN (Soto et al. 2008; FAO 2018d; Table 3). For example, improved efficiency of resource use (Principle 1) starts with increasing water use efficiency and productivity of farming systems and protecting the hydrological functions of wetlands in the landscape. This includes protecting water sources for wetlands, reducing water abstraction and changes to water flows near wetlands, and maintaining environmental flows of rivers. Irrigation can be more effective by improving infrastructure and management. Natural or human-made wetlands can play a role in water harvesting and retention. New technology (e.g. soil moisture sensors, improved weather forecasts) can contribute to optimizing water management. Overuse of fertilizers and pesticides can be addressed by matching applications to crop needs, e.g. through integrated nutrient (INM) and pest management (IPM) and by coordinating applications with irrigation management. Nutrient leaching to wetlands and surface water can be avoided by limiting fertilizer applications near wetlands. Integration of nutrient removal by riparian buffer zones, or constructed wetlands with recycling of nutrients to agriculture can reduce the need for fertilizer while mitigating fluxes of nutrients to downstream aquatic ecosystems. Precision agriculture limits the emissions of nutrients and contributes to more efficient resource use. Changes in the marketing of agricultural inputs and educating farmers can contribute to avoiding overuse (Vymazal 2017; UN Water 2018; FAO/IWMI 2018; Jiang and Mitsch 2020; Miralles-Wilhelm 2021).

More efficient resource use and reduced impact on wetlands also go hand in hand when crop, livestock and fish production are more integrated. All approaches for integration (e.g. integrated natural resources management; agroecology; integrated nutrient management; conservation agriculture) aim at creating favorable conditions for plant growth through enhanced soil organic matter and biotic activity, minimizing water losses and nutrient leakage by water harvesting, soil cover or minimum tillage, recycling and balancing of nutrient flows, and enhancing diversity, biological interactions and synergisms in agroecosystems (Altieri 2002; Wu and Ma 2015; FAO 2017a; FAO 2018a; Lal 2020). Agroforestry combines crops or livestock with trees or shrubs, with benefits such as increased yields, greater efficiency in nutrient, light and water use, reduced erosion, and increased biodiversity and soil carbon storage (Nair and Garrity 2012; Arunachalam et al. 2014; FAO 2017b). Agricultural wetland landscapes with crop or fish cultivation integrated in the landscape facilitate cultivation but do not drain wetlands permanently, allowing for continued delivery of regulating wetland ecosystem services like flood control, groundwater recharge or water quality regulation. Paludiculture (production of food, fodder, fuel, pharmaceuticals or construction materials in wet or rewetted peatlands) can contribute to reducing soil degradation and GHG emissions, and can be combined with other practices such as fisheries, aquaculture or livestock (Joosten et al. 2016) but more research, particularly on paludiculture in tropical peatlands, is needed (Tan et al. 2021).

Promoting more integrated food production is attractive but also challenging. Integrated systems were traditionally in use all over the world (Altieri 2002; Prein 2002), but many traditional systems such as the Mexican chinampas (Merlín-Uribe et al. 2013), or the fishpond landscapes in Eastern Europe (Harmáčková and Vačkář 2015) or China (Ruddle and Zhong 1988; Lang et al. 2009) have degraded or disappeared under changing social and economic conditions and as a result of intensification. Attempts to introduce them elsewhere have seen mixed results. Developing conservation agriculture and implementing innovative integrated agriculture-wetland systems requires knowledge not only about technical aspects, but also about socio-economic, cultural and institutional factors that determine the adoption of these technologies (Stevenson et al. 2014; Liebig et al. 2017).

Efficiency can also be increased on the post-harvest and demand side of the food system, by reducing post-harvest losses and food waste. About one third of all food that is produced globally is lost every year (Alexander et al. 2017). A reduction of food waste can be achieved e.g. by educating consumers and producers about the environmental and social impacts of food choices, by reducing overproduction, by improving post-harvest activities and infrastructure in developing countries, and by reusing and recycling of crop residues through composting (FAO 2018c). A reduction in meat consumption (especially beef) can also contribute to reducing the inefficiency of the food system and reducing GHG emissions (Gerber et al. 2013; Clark and Tilman 2017).

Table 3. FAO's principles and actions for Sustainable Food and Agriculture (FAO 2018c; first two columns); and what this means for sustainable wetland-agriculture interactions (third column).

FAO Principles	FAO Actions	Actions towards more sustainable wetland-agriculture interactions
1. Improving efficiency in the use of resources is crucial to sustainable agriculture	1 Facilitate access to productive resources, finance and services 2 Connect smallholders to markets 3 Encourage diversification of production and income 4 Build producers' knowledge and develop their capacities	• Efficient water management • Efficient nutrient and pest management (INM, IPM) • Integration of crop, livestock and fish systems, conservation agriculture • Integrated wetland-agriculture (incl. paludiculture) • Reduce post-harvest losses, food waste, meat consumption
2. Sustainability requires direct action to conserve, protect and enhance natural resources	5 Enhance soil health and restore land 6 Protect water and manage scarcity 7 Mainstream biodiversity conservation and protect ecosystem functions 8 Reduce losses, encourage reuse and recycle	• Protect the ecological character of wetlands • Promote wise use of wetlands • Avoid further loss of wetlands • Restore degraded wetlands, particularly coastal wetlands and peatlands that store carbon
3. Agriculture that fails to protect and improve rural livelihoods, equity and social well-being is unsustainable	9 Empower people and fight inequalities 10 Promote secure tenure rights for men and women 11 Use social protection tools to enhance productivity and income 12 Improve nutrition and promote balanced diets	• Improve access to markets for small farmers • Improve tenure security, access to credit and other support services
4. Enhanced resilience of people, communities and ecosystems is key to sustainable agriculture	13 Prevent and protect against shocks: enhance resilience 14 Prepare for and respond to shocks 15 Address and adapt to climate change 16 Strengthen ecosystem resilience	• Reward landscape stewardship role of farmers • Financial and technical support for sustainable farming methods • Transformation of farming practices and landscape management
5. Sustainable food and agriculture requires responsible and effective governance mechanisms	17 Enhance policy dialogue and coordination 18 Strengthen innovation systems 19 Adapt and improve investment and finance 20 Strengthen enabling environment and reform institutional framework	• Consider trade-offs between wetland ecosystem services and food production to maximize benefits • Dialogue, negotiation, cooperation and conflict resolution between sectors at catchment scale • Adaptive, multi-level governance

Principle 2 (protection of natural resources) has been the remit of the Ramsar Convention on Wetlands, and usually translates into guidance for national governments on the protection of the ecological character of Ramsar wetland sites. This remains important, because further loss of wetlands, including by conversion for agriculture, should be avoided, even more so now that the importance of wetlands for climate change mitigation, and the role of wetlands in protecting people from climate extremes and adapting to climate change is widely recognized (Moomaw et al. 2018). The Convention's goals cover not only protection but the wide range of management measures from protected areas to wise use of wetlands, which includes sustainable use for food production, other forms of livelihoods support, and more generally the contributions of wetlands to achieving sustainable development (Finlayson et al. 2011; Ramsar Convention 2018b; Kumar et al. 2020).

Improving the livelihoods of small-scale farmers and their resilience to change (Principles 3, 4) has been the main goal of agricultural policies in many high income countries from the middle of the 20th century (e.g. US and EU). This has led to a vast improvement of farm productivity and farmer incomes, but also to an increase in farm size and a reduction in the number of farms. Intensification and land use change led to environmental degradation (including loss of wetlands) and biodiversity loss. On the other hand, millions of small farm households in low and middle income countries today still struggle to increase farm productivity and income, with insecure tenure and access to land, a degrading natural environment, poor access to markets for their produce, and a lack of credit and extension support (Dixon et al. 2001; Terlau et al. 2009; Jayne et al. 2010). Low farm productivity in combination with the impacts of insecure rainfall and more extreme events (climate change) can increase the pressure on natural systems such as wetlands. For a long time, a "Green Revolution" scenario was considered the best development option, but this has considerable social and environmental limitations (Pingali 2012). A more attractive alternative is a pathway in which food production and environmental concerns are considered simultaneously. Besides benefits at the farm level, wetland ecosystem services extend to the landscape scale, e.g. by providing water to downstream users, by controlling floods, or by providing habitat for important species. Recognizing the stewardship role of food producers and making use of their traditional knowlege in maintaining these ecosystem services can promote sustainable food production and wise use of wetlands simultaneously. To achieve this, approaches for compensating farmers for landscape management are needed, e.g. through payments for environmental services (PES) or other incentive schemes. It would require fundamental changes in economic policy and in governance, to build bridges between government agencies, the private sector, and farmers and farmer cooperatives, and to create financial mechanisms and valuation methods to ensure fair prices, other forms of compensation, and technical support while avoiding harmful subsidies and other perverse incentives (Wood et al. 2013; Bennett et al. 2018; ten Brink and Russi 2018).

This leads to the innovative transformative approaches to the governance of agriculture and the environment (Principle 5) needed to address the challenges of sustainable food production and wetland degradation with a growing population in a changing climate. Policies in many high-income countries are developing in this direction, for example with the reforms in the EU Common Agricultural

Policy (Pe'er et al. 2019; Heyl et al. 2021). Countries that need strong growth of their food production sectors can move straight in this direction, and avoid the social and environmental risks of a pathway of 'classical' intensification and globalisation. The emphasis on one-sided sectoral management, either for optimum crop yields or for strict wetland protection may transform to an integrated catchment/landscape approach which recognizes the contribution of both wetlands and agriculture to combined biodiversity, livelihoods and development goals. To achieve this, trade-offs within the wider landscape need to be considered to arrive at the mix of ecosystem services that maximizes benefits (Wood and Van Halsema 2008; Tanner et al. 2013; Wood et al. 2013; Freeman et al. 2015; Everard and Wood 2018). Methods and tools for quantification of wetland ecosystem services and trade-offs can support the decision-making process (e.g., Zsuffa et al. 2014; Santos et al. 2017). The competing demands of food production, water resources management, ecosystem services, biodiversity, poverty alleviation, and social and economic development also need coordinated management, that responds and adapts to changing conditions (e.g. climate change) through dialogue, negotiation, cooperation and sometimes conflict resolution between different sectors and levels of jurisdiction. Such adaptive, multi-level governance must be supported by suitable institutional and finance frameworks, new policies and regulation and a commitment to long-term engagement from all stakeholders (e.g., Schallenberg et al. 2017).

In conclusion, there is no intrinsic contradiction between food production and wetland integrity. Food production depends on intact, healthy ecosystems, and humans have always relied on harvesting food from natural ecosystems or modifying ecosystems to produce aquatic or terrestrial plants and animals. Throughout history, this has led to impacts on the natural environment, but with the intensification and globalisation of food systems these impacts have now grown to unsustainable levels (Rockström et al. 2009; Campbell et al. 2017). Increasingly the connectedness of humans and the Earth environment is re-discovered, e.g. in the thinking about social-ecological systems and sustainability science that seeks to overcome human–nature dualism and recognizes the rights of Nature (Díaz et al. 2015; Finlayson et al. 2022). Wetlands are part of this, and although the interactions between food production and wetlands have often been approached from their separate positions and discourses (agriculture or aquaculture versus wetland conservation), it is now time to consider food production in a smarter, more integrated manner with recognition of the diversity in both wetlands and farming systems and their interactions. Food production and healthy wetlands can go together in a socially and environmentally sustainable way, ensuring the biodiversity and ecosystem services of wetlands, the livelihoods for people living in or near wetlands, and food sovereignty for countries harbouring valuable wetland resources. This approach reflects the ambition and foresight of those who founded the Ramsar Convention on Wetlands more than 50 years ago (Stroud et al.

2021) and of those who in subsequent years have striven to support such efforts (Gell et al. 2023) through transformation of agricultural systems alongside wetland conservation and wise use.

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Supplementary materials

S1. Description of the 8 farming system types, and of integrated systems

Rainfed systems are often located in high (upland) or dry (arid) zones, but are also found in many sub-tropical and tropical lowland areas. In dry areas, cereal crops like maize, millet and sorghum are produced. In high areas, a variety of crops can be found. *Extensive rainfed systems* are often smallholder farms, many of which exist in developing countries, producing mainly or entirely for subsistence. Input levels (fertilizer, pesticides, improved seeds, compound feeds) are generally low, and often farms are in areas with suboptimal rainfall or have soils that need more fertilizer inputs to become more productive. Farms are often mixed, with multiple crops and also livestock, and farm households engage in diverse activities to generate income, including other livelihoods activities like fishing, production of handicrafts, and off-farm jobs. *Intensive rainfed systems* are most common in climate zones with sufficient rainfall, good and well-managed soils, fertilizer and agro-chemicals inputs, and farm mechanization. They are common in temperate zones like Europe, North America or New Zealand and can also occur in sub-tropical and tropical areas in South Africa, Brazil, Eastern China, and parts of India. They often produce for the national or international markets. In temperate zones, crops would consist of monocrops of wheat, maize (also for feeding animals), barley, soybeans, rapeseed, sugar beet or potatoes. In the sub-tropics, besides wheat there could be fruits and oil crops (e.g. soybeans). In the tropics maize, sugarcane and soybeans are important. Intensive rainfed systems can also be found in combination with intensive livestock farming. Most monocultures of trees and cash crops (plantation agriculture) are also a form of intensive rainfed systems, some of them on drained wetlands such as palm oil, *Eucalyptus* and *Acacia* plantations in southeast Asia on drained peat swamp forests, or sugarcane cultivation in floodplains.

Irrigated systems currently occupy a quarter of the total cropped area in the world. About 70% of this area is in Asia (especially south and southeast but also west Asia), but north Africa and the dryer parts of Australia, North America and Europe are also important irrigation areas. About two thirds of this area is supplied with surface water, and one third with groundwater but the share of groundwater has been increasing (Siebert et al. 2010; Wada and Bierkens 2014). This can lead to competition for water with domestic or industrial use, and in areas where water is over-exploited to water shortages. Irrigation is an important factor in intensification and productivity increase. It allows the effective use of better crop varieties and higher input levels (fertilizers, pesticides), which leads to higher yields (generally at least double compared to similar rainfed systems) and often allows harvesting more than one crop per year. Crop types can include a wide variety, including cereals (rice, maize, wheat) and cash crops and fruits (cotton, almonds, palm oil). In Asia, 70-85% of irrigated land is used for the production of cereals, particularly rice. In Africa and the Americas, this is 40-50% and in Europe and Oceania even lower (30% or less). *Horticulture* is a high-precision form of intensive irrigated systems which can be operated outdoors but also in indoor systems (including glass houses) where fertilizer use can be optimized and water can be recycled. Most important horticulture products are fruits, vegetables and ornamental plants (including flowers).

Extensive livestock systems are grazing-based systems in areas unsuitable for cultivated crops because of insufficient or too variable rainfall, low temperatures, or infertile, stony soils and unsuitable terrain (e.g. steep slopes). Important grazing areas can be found in Central and East Asia, the semi-arid and arid savannahs of Central and East Africa, and the highlands of Europe, the Middle East, North Africa and South America. Animals in these systems are ruminants: traditional cattle breeds, sheep and goats, all capable of digesting high-fibre vegetation. Traditionally, people whose livelihoods are based on these systems are nomadic, migrating with the herds in search of grazing areas. Wetlands can be important because of the seasonal productivity peaks in wetland vegetation. In many parts of the world, extensive livestock systems are under pressure of decreasing grazing areas, increasing livestock densities or conversion to crop land (UNCCD 2017). A somewhat more intensive form of livestock production are mixed crop-livestock farms. Animals in these farming systems are diverse, can be small in number, and are sometimes kept for financial security. They can be ruminants, but also mono-gastric animals like pigs or poultry, and are often (depending on species) fed with fodder collected on-farm or off-farm, food and crop wastes, supplemented with grazing on the farm and roaming in homestead areas (Herrero et al. 2009; McDermott et al. 2010; Udo et al. 2011).

Intensive livestock systems include more intensive mixed crop-livestock systems, with intensively managed pasture (mainly for cattle), or landless industrial livestock production systems (mainly for pigs and poultry). Intensive grazing systems (animal density >100 km^{-2}; IPCC 2019) occur in areas with sufficient rainfall for grassland to be productive under more intensive management, including fertilization. These systems produce meat, dairy and other products and are common in North and South America, Australia and Europe. Grazing can be supplemented with concentrated feeds, and other inputs include genetically improved breeds and veterinary support. The major part of global milk production (~ 80%) comes from Europe, North America and Asia (mostly India, Pakistan, China). New Zealand and Australia, Brazil, Argentina and Russia are also significant dairy producing countries (Tivy 1990; FAO 2019). Landless systems are used for the production of monogastric animals like pigs (meat) and poultry (meat, eggs). More than half of all pigs and almost 75% of all poultry are produced in such systems. The animals are fed entirely with formulated feeds based on cereal grains as the main source of carbohydrates, and fishmeal and soybeans as protein sources. Landless systems can be established where supply channels for feeds and feed ingredients are available, and the waste (faeces and urine) can be disposed of without high costs. Feeds are often imported and a considerable area of agricultural land is needed for their production (UNCCD 2017; IPCC 2019). Concerns have been raised over the impact of cultivation of animal feed crops on natural ecosystems, e.g. through deforestation (Gibbs et al. 2015).

Aquaculture is the production of aquatic animals or plants under captivity with a clear ownership arrangement (as opposed to capture fisheries, which deal with the harvesting of wild stocks. In broad terms, four production systems can be distinguished: ponds, enclosure systems (floating cages and pens), tank systems, and coastal aquaculture systems. Most of the freshwater fish and marine shrimps are produced in *freshwater or brackish fishponds* in Asia, especially China and other countries in Asia (Verdegem and Bosma 2009). Marine ponds are used for growing species like milkfish, mullets, seabass, a variety of other marine species, and for shrimps of the genus *Penaeus*. Freshwater pond systems are by far the most important production systems in terms of quantity, producing an estimated 33 million tonnes per year (FAO 2018b). The majority of ponds are excavated earthen ponds that can function partly as natural ecosystems. Extensive ponds rely on primary production and the natural pond foodweb to produce feed for the fish. In more intensive ponds, fertilization, supplemental feeding, and aeration can be used to achieve higher production. In principle, freshwater ponds are stagnant and only require water for initial stocking and for compensating evaporation and seepage losses. As the intensity of production increases, water use can increase. Aquaculture pond systems around the world are going through a process of intensification and increasingly depend on external feed inputs to achieve high yields, leading to higher discharge of sediment and nutrients to surface water (Edwards 2015). Further intensification can lead to production in concrete or fiberglass *tank systems* which allow even more control of water quality and feeding regimes. Intensive tank systems can also remove excreta by filtration, after which the water can be recycled to the fish tanks. Recirculating systems can produce fish in very high densities and have a much-reduced water use, however they are more energy-intensive (Martins et al. 2010).

Enclosure systems can be floating cages (fully mesh-enclosed systems) that do not permit the fish to be in contact with anything but the water; or pen systems which enclose a shallow area of water and allow the fish access to the bottom sediment. The majority of enclosure systems are used for rearing finfish. Most cage culture relies on the use of formulated feeds (see intensive aquaculture below). Most freshwater cage production takes place in China, Vietnam, Thailand, Bangladesh and

Indonesia while most marine cage culture (salmonids) is in temperate regions. In recent years, freshwater cage culture has developed strongly on some African lakes like Lake Volta and Lake Victoria. Important pen systems are found in Asia, e.g. the culture of milkfish and other species in Laguna de Bay, Philippines. Production in most enclosure systems depends on external feed inputs.

Coastal aquaculture includes the production of finfish, shrimps, molluscs, and aquatic plants. Marine fish and shrimp can be produced in ponds or enclosure systems which generally involves providing feeds (see intensive aquaculture below). Molluscs (bivalves such as oysters, mussels, cockles, clams; and gastropods such as abalone and sea snails) are produced in a variety of systems, usually involving ropes or trays to which the animals can attach. Production of aquatic plants (mostly seaweeds, including kelp) traditionally takes place in shallow coastal water. Often, floating lines tied to stakes in the bottom are used. Recently, methods for culture in deeper water (6-7 m) have been developed (Zhang et al. 2022). More than 60% of global seaweed production is used for non-food purposes, like the production of carrageenan and in feeds for aquaculture (Edwards et al. 2019). In the absence of feed or fertilizer inputs, the impact of mollusc and plant aquaculture in terms of water consumption and eutrophication is limited compared to other forms of aquaculture. Bivalves and aquatic plant culture share their use of the natural productivity of the coastal environment: bivalves filter phytoplankton and particles, and seaweeds utilize nutrients from the water to produce biomass.

Integration is the linking of farm components in which an output from one component, which may otherwise go to waste, becomes an input to another component resulting in reduced nutrient imports and a greater production efficiency for the same land/water area (Edwards et al. 1988; Edwards 1998; Prein 2002; Costa-Pierce et al. 2010). Integration can also be realized beyond the farm borders to include the use of off-farm resources, for example agro-industrial (by-)products to feed livestock, urban wastewater as a source of nutrients, or heat recovery from effluents for increasing the water temperature in aquaculture. Integration can be physical, where a sub-system becomes part of another sub-system through spatial integration (e.g. intercropping in agriculture, polyculture in pond aquaculture, rice-fish culture, or integrated irrigation-aquaculture; Halwart and Van Dam 2006). Sub-systems can also be linked through flows of nutrients, water and by-products (e.g. the use of livestock manure for fertilizing crops and fishponds, fishpond sediment for growing crops, or crop residues to feed livestock; Prein 2002). In marine aquaculture, integrated multi-trophic aquaculture (IMTA) systems combine fed aquaculture of fish or shrimp with extractive aquaculture of seaweeds or shellfish, the latter absorbing the excess nutrients discharged by the former (Troell et al. 2009; Chopin et al. 2012). Integrated systems often foster a higher diversity, have relatively low external inputs and maximize the recycling of nutrients, thereby minimizing the need for fertilizers. While in many traditional integrated farming systems a drive for higher production and intensification has led to de-integration (Edwards 2015), Europe and North America are seeing a renewed interest in crop-livestock integration as part of a movement towards more sustainable or circular agriculture (Eyhorn et al. 2019).

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S2. Diagrams showing the hydrological pathways in a catchment and the impacts of different food production systems on water and nutrient flows

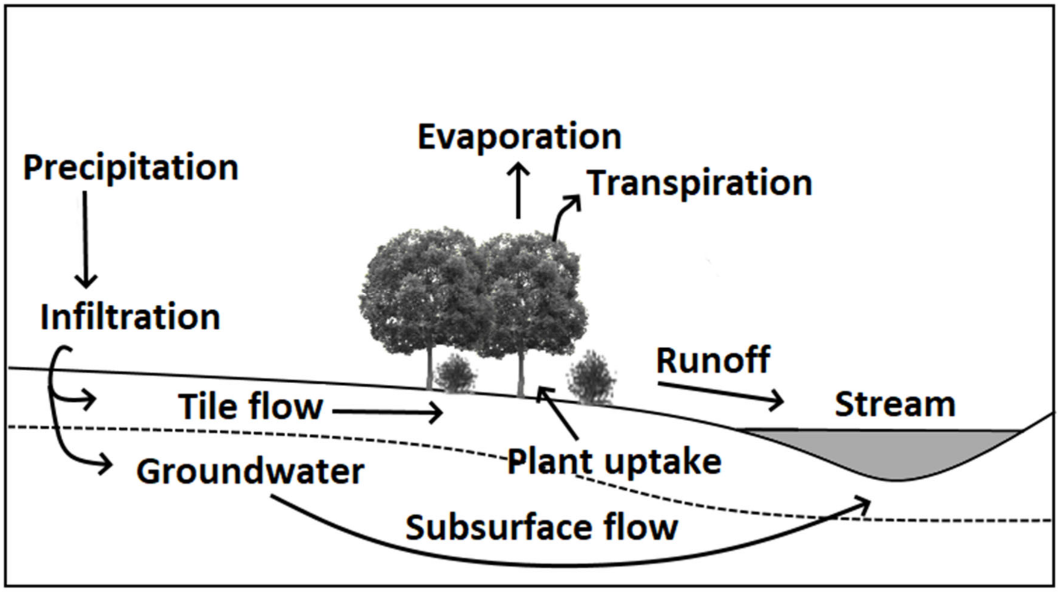

Figure S1. Main hydrological pathways in a catchment.

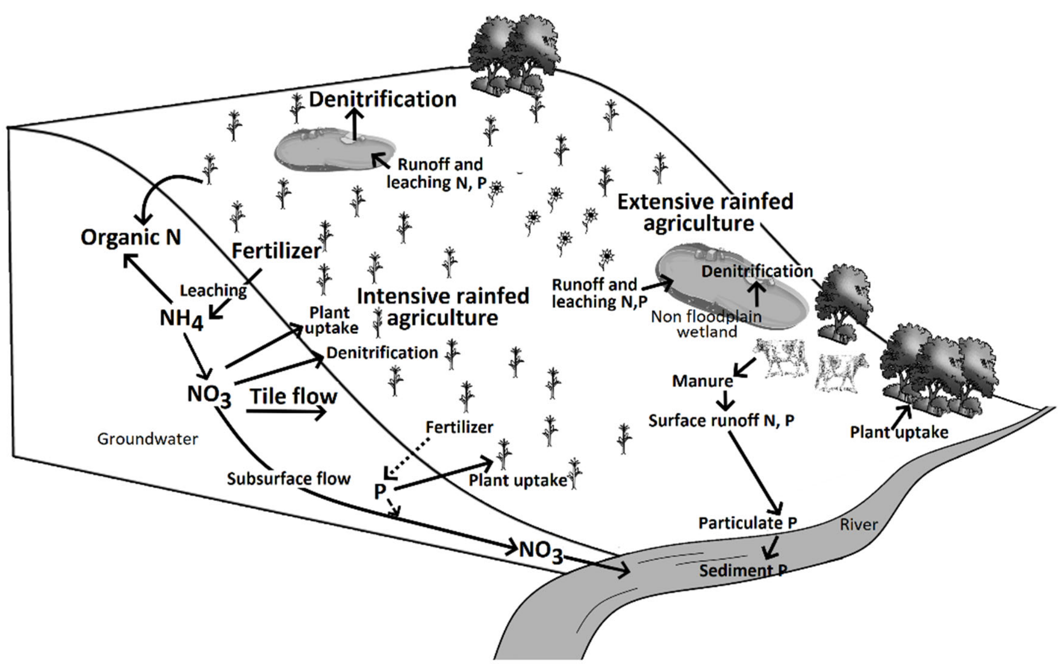

Figure S2. Impacts of rainfed agricultural systems on catchment water and nutrient flows.

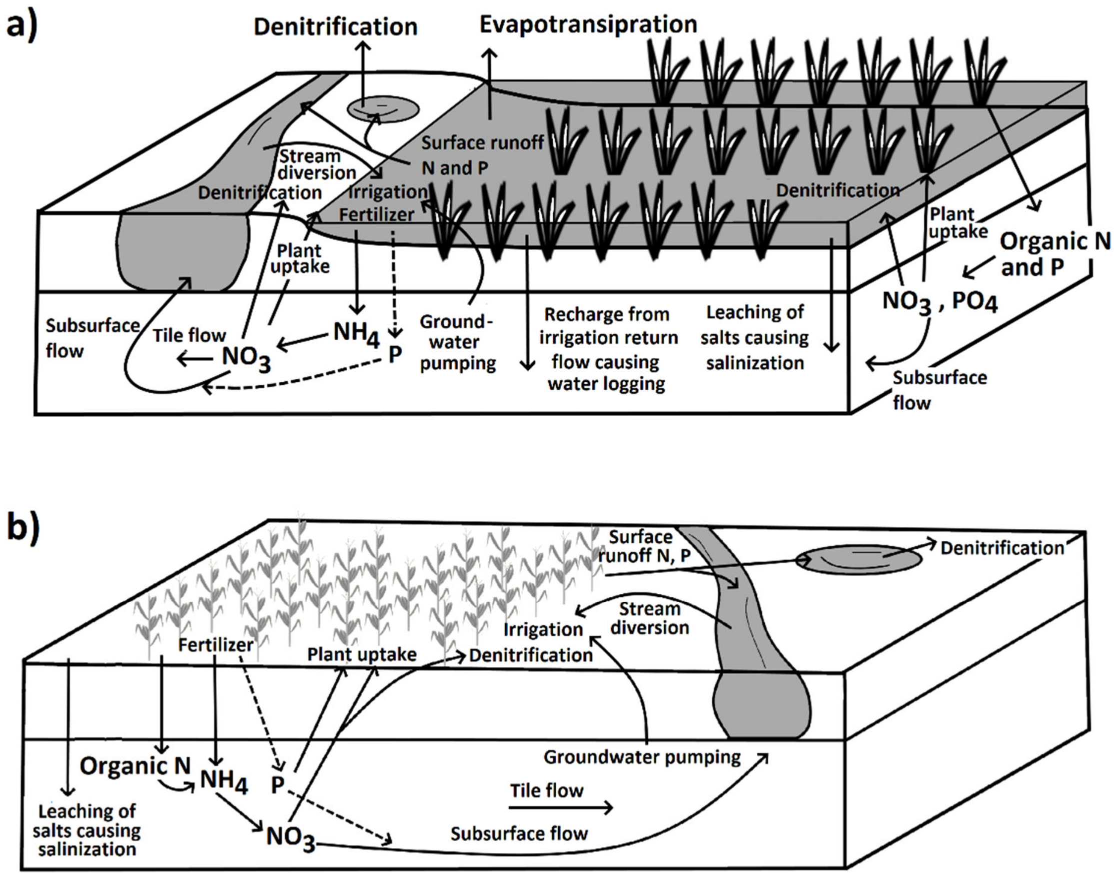

Figure S3. Impacts of irrigated agriculture on catchment water and nutrient flows. a) lowland rice paddy systems; b) irrigation in arid regions.

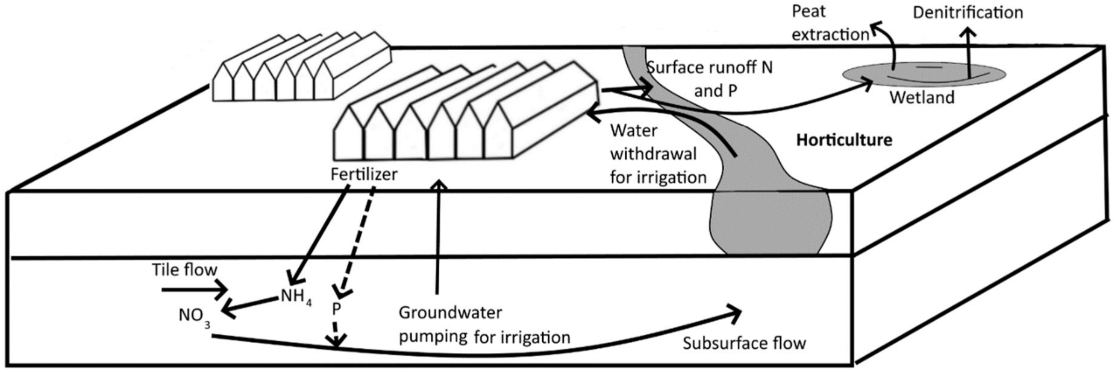

Figure S4. Impacts of horticulture on catchment water and nutrient flows.

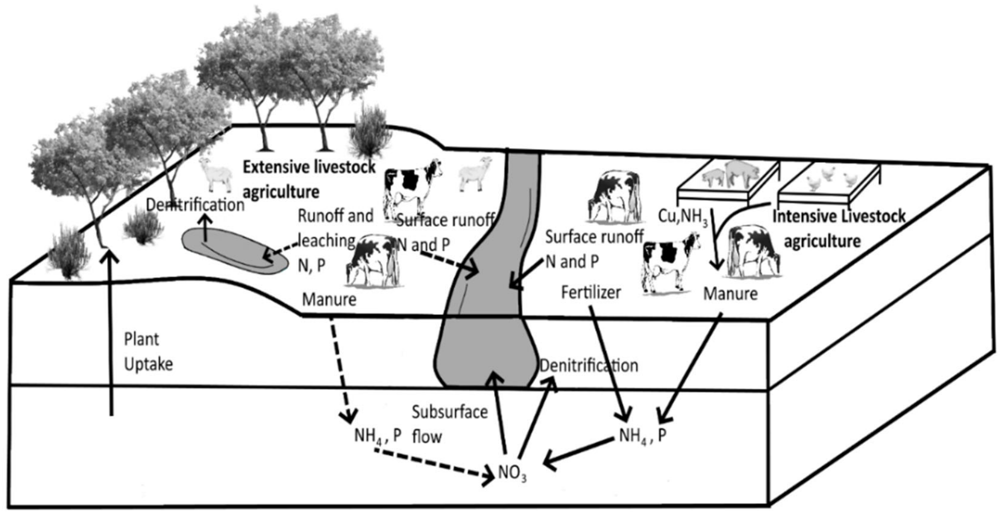

Figure S5. Impact of livestock systems on catchment water and nutrient flows.

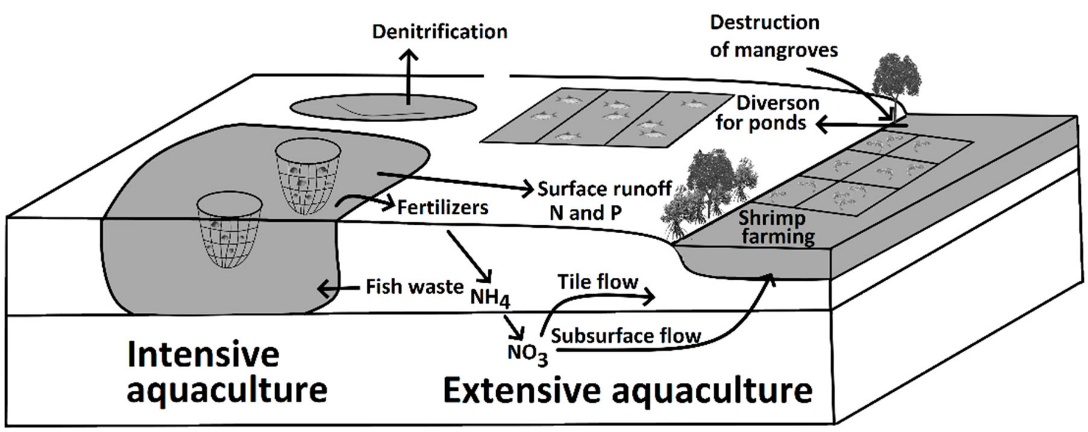

Figure S6. Impacts of aquaculture on catchment water and nutrient flows.