YSLME Project Phase II

Develop regional strategy for using wetlands as nutrient sink project

Review report

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Executive Summary

This report focuses on reviewing the status of coastal wetland and their roles in nutrient removal for the Yellow Sea Coastal area, mainly including the coastal wetland ecosystem distribution in Yellow Sea, the ecosystem services of coastal wetland, the mechanisms of nutrient retention, and the wetland management or restoration in Yellow Sea area. The production of the report has been stimulated by an apparent lack of recognition and focus on coastal wetland ecosystems to develop a regional strategy for using wetlands as nutrient sink, especially for the coastal wetlands.

To construct this report, we had done the extensive literature research about wetlands and using wetlands as nutrient sink, and divided it into four parts, which are the four chapters listed as followed. These resultant chapters were reviewed from the literatures from these scientists who are the core of this research area. These literatures were selected because all of them were relevant with using coastal wetland as nutrient sink, especially for those implement in the Yellow Sea area. There are of course other features of our coastal wetland that already established as good nutrient sink ---the key focus for the initial work has, however, been on those ecosystems where management and restoration intervention can reasonably readily play a role in improving the future state of the given role. Hope this report could expand the range of global options for using wetland as nutrient sink into Yellow Sea coastal wetland, unlocking many possibilities for action and possible financing of new management and restoration measures to protect the important coastal wetlands.

The main chapters and the contents of this report are:

In chapter 1, we have made a summary for the review report using wetland as nutrient sink, which mainly including the wetland ecosystem distribution and status of coastal wetland especially in China and RoK, the ecosystem services can provide of the coastal wetland, the mechanisms of nutrient retention, and wetland management and restoration status in Yellow Sea.

In chapter 2, beside we have done the literature research for the wetland distribution, we have most focused on the changes of coastal wetland from 1950s, which will have great benefit to explain the necessary of wetland management and restoration, this chapter had proved that both in China and RoK, the coastal wetland were undergoing a great decreasing and degradation both distribution area and ecosystem services.

In chapter 3, we focused on the nutrient load in Yellow Sea area according to the two kinds of nutrient load, which were from river and the atmosphere and the results implied that the riverine discharge of nutrient into the Yellow Sea was lower than that from the direct atmospheric deposition.

In chapter 4, we have reviewed the wastewater treatment and nutrient removal in wetland, which mainly including the wastewater and manure dumping, nutrient removal, and enhanced nitrous oxide emission control. This chapter implied that using wetland as nutrient sink was a feasible method but should also pay special attention to the other negative effects such as greenhouse gas emission.

In chapter 5, we have collected several examples for the project implement for using wetland as nutrient sink which both included using natural and constructed wetlands. Although several wetlands was implemented in the inland wetland or in experiment, they use constructed wetlands with horizontal sub-surface flow for various types of wastewater will have great benefit for out further work using wetland as nutrient sink.

This report provides the essential evidence need to motivate discussions and initiatives on how much coastal wetland ecosystems should be incorporated in to nutrient remove in Yellow Sea area. These evidences presented here makes clear why using coastal wetland as nutrient sink. The coastal wetland management and restoration is not only a political imperative for biodiversity conservation, food security, and shoreline protection, but also now for helping mitigate nutrient pollution.

Chapter 1. Background and introduction

1.1 Wetlands and coastal wetlands

The Ramsar Convention defined that wetlands are areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres. They may incorporate riparian and coastal zones adjacent to the wetlands, and islands or bodies of marine water deeper than six metres at low tide lying within the wetlands, especially where these have importance as waterfowl habitat (Ramsar Convention Secretariat 2010). According to the above definition, coastal wetlands mainly include mangroves, salt marshes, seagrass, coral reefs, beaches, estuaries, and coastal water bodies within −6 m depth.

The Ramsar Hanbook listed the general classification system of wetlands (see Table 1.1), which was approved by the 1990 Conference of the contracting Parties (Recommendation 4.7) and subsequently amended. In this classification system, forty-two wetland types are identified in the system, grouped into the categories "coastal/marine", "inland", and "human-made", which provide a broad framework for the rapid identification of the main wetland habitat types represented at each site, with the "dominant wetland type" clearly indicated.

Table 1.1 Classification system for wetland type (adapted from The Ramsar Handbook, 2016)

Type	Code	Subtype and description
Marine/Coastal Wetlands	A	**Permanent shallow marine waters** in most cases less than six metres deep at low tide; includes sea bays and straits.
	B	**Marine subtidal aquatic beds**; includes kelp beds, sea-grass beds, tropical marine meadows.
	C	**Coral reefs**
	D	**Rocky marine shores**; includes rocky offshore islands, sea cliffs.
	E	**Sand, shingle or pebble shores**; includes sand bars, spits and sandy islets; includes dune systems and humid dune slacks.
	F	**Estuarine waters**; permanent water of estuaries and estuarine systems of deltas.
	G	**Intertidal mud, sand or salt flats**.
	H	**Intertidal marshes**; includes salt marshes, salt meadows, saltings, raised salt marshes; includes tidal brackish and freshwater marshes.
	I	**Intertidal forested wetlands**; includes mangrove swamps, nipah swamps and tidal freshwater swamp forests.
	J	**Coastal brackish/saline lagoons**; brackish to saline lagoons with at least one relatively narrow connection to the sea.
	K	**Coastal freshwater lagoons**; includes freshwater delta lagoons.

Type	Code	Subtype and description
	Zk(a)	**Karst and other subterranean hydrological systems**, marine/coastal
Inland Wetlands	L	**Permanent inland deltas.**
	M	**Permanent rivers/streams/creeks**; includes waterfalls.
	N	**Seasonal/intermittent/irregular rivers/streams/creeks.**
	O	**Permanent freshwater lakes** (over 8 ha); includes large oxbow lakes.
	P	**Seasonal/intermittent freshwater lakes** (over 8 ha); includes floodplain lakes.
	Q	**Permanent saline/brackish/alkaline lakes.**
	R	**Seasonal/intermittent saline/brackish/alkaline lakes and flats.**
	Sp	**Permanent saline/brackish/alkaline marshes/pools.**
	Ss	**Seasonal/intermittent saline/brackish/alkaline marshes/pools.**
	Tp	**Permanent freshwater marshes/pools**; ponds (below 8 ha), marshes and swamps on inorganic soils; with emergent vegetation water-logged for at least most of the growing season.
	Ts	**Seasonal/intermittent freshwater marshes/pools** on inorganic soils; includes sloughs, potholes, seasonally flooded meadows, sedge marshes.
	U	**Non-forested peatlands**; includes shrub or open bogs, swamps, fens.
	Va	**Alpine wetlands**; includes alpine meadows, temporary waters from snowmelt.
	Vt	**Tundra wetlands**; includes tundra pools, temporary waters from snowmelt.
	W	**Shrub-dominated wetlands**; shrub swamps, shrub-dominated freshwater marshes, shrub carr, alder thicket on inorganic soils.
	Xf	**Freshwater, tree-dominated wetlands**; includes freshwater swamp forests, seasonally flooded forests, wooded swampson inorganic soils.
	Xp	**Forested peatlands;** peatswamp forests.
	Y	**Freshwater springs**; oases.
	Zg	**Geothermal wetlands**
	Zk(b)	**Karst and other subterranean hydrological systems**, inland
human-made wetlands	1	**Aquaculture** (e.g., fish/shrimp) **ponds**
	2	**Ponds**; includes farm ponds, stock ponds, small tanks; (generally below 8 ha).
	3	**Irrigated land**; includes irrigation channels and rice fields.
	4	**Seasonally flooded agricultural land**(including intensively managed or grazed wet meadow or pasture).
	5	**Salt exploitation sites**; salt pans, salines, etc.
	6	**Water storage areas**; reservoirs/barrages/dams/impoundments (generally over 8 ha).
	7	**Excavations**; gravel/brick/clay pits; borrow pits, mining pools.
	8	**Wastewater treatment areas**; sewage farms, settling ponds, oxidation basins, etc.
	9	**Canals and drainage channels, ditches.**
	Zk(c)	**Karst and other subterranean hydrological systems**, human-made

1.2 Ecosystem services of coastal wetlands

The Millennium Ecosystem Assessment summarized the ecosystem services that coastal wetlands can provide for human. Table 1.2 provides a list of the main services provided by different types of coastal wetland and their general relative magnitude.

Table 1.2 Ecosystem services provided by coastal wetlands
Source: Millennium Ecosystem Assessment (Finlayson et al. 2005).

Services	Comments and Examples	Estuaries and Marshes	Mangroves	Lagoons, Including Salt Ponds	Intertidal Flats, Beaches, and Dunes	Kelp	Rock and Shell Reefs	Seagrass Beds	Coral Reefs
Coastal Wetlands									
Provisioning									
Food	production of fish, algae, and invertebrates	⬤	⬤	•	●	•	●	•	●
Fresh water	storage and retention of water; provision of water for irrigation and for drinking	•		•					
Fiber, timber, fuel	production of timber, fuelwood, peat, fodder, aggregates	⬤	⬤	●					
Biochemical products	extraction of materials from biota	•	•			•			•
Genetic materials	medicine; genes for resistance to plant pathogens, ornamental species, and so on	•	•	•		●			•
Regulating									
Climate regulation	regulation of greenhouse gases, temperature, precipitation, and other climatic processes; chemical composition of the atmosphere	●	●	●	•		•	•	●
Biological regulation (C11.3)	resistance of species invasions; regulating interactions between different trophic levels; preserving functional diversity and interactions	●	⬤	●	•		•		•
Hydrological regimes	groundwater recharge/discharge; storage of water for agriculture or industry	•		•					
Pollution control and detoxification	retention, recovery, and removal of excess nutrients and pollutants	⬤	⬤	●		?	•	•	•
Erosion protection	retention of soils	●	⬤	•				•	•
Natural hazards	flood control; storm protection	⬤	⬤	•	•	•	●	●	⬤
Cultural									
Spiritual and inspirational	personal feelings and well-being	⬤	•	●	⬤	•	•	•	⬤
Recreational	opportunities for tourism and recreational activities	⬤	•	•	⬤	•			⬤
Aesthetic	appreciation of natural features	●	•	●	●				⬤
Educational	opportunities for formal and informal education and training	•	•	•	•		•		•
Supporting									
Biodiversity	habitats for resident or transient species	●	●	•	⬤	•	⬤	•	⬤
Soil formation	sediment retention and accumulation of organic matter	●	●	•	•				
Nutrient cycling	storage, recycling, processing, and acquisition of nutrients	●	●	●	•	•	•		●

Note: Scale is low •, medium ●, to high ⬤; not known = ?; blank cells indicate that the service is not considered applicable to the wetland type. The information in the table represents expert opinion for a global average pattern for wetlands; there will be local and regional differences in relative magnitudes.

Coastal wetlands can provide important services, such as food and bio-materials as direct resources, habitat for wildlife, carbon sequestration, protection against storm surges, and sediment accumulation for land accretion (Barbier *et al.*, 2011, Camacho-Valdez *et al.*, 2013). They also provide water purification, tourism resorts, and other functionalities (Figure 1.1). Although the area of coastal wetlands is rather small compared to many other terrestrial ecosystems, their productivity is comparable to the most productive ecosystems. Moreover, as key habitats for many terrestrial and marine species, vegetated zones and tidal creeks provide diverse shelter and food sources for a large variety of wild animals, resulting in high biodiversity and unique food webs. About two-thirds of marine animals, such as fish, shrimps, crabs, mollusks, and turtles, have to spend some time at coastal wetlands during their life history, and over 90% of marine fisheries are sourced from coastal zones, either through harvesting of wild organisms or mariculture (Hinrichsen, 2008). In the meantime, the coastal wetlands provide food sources and habitats for millions of waterbirds (Figure 1.1).

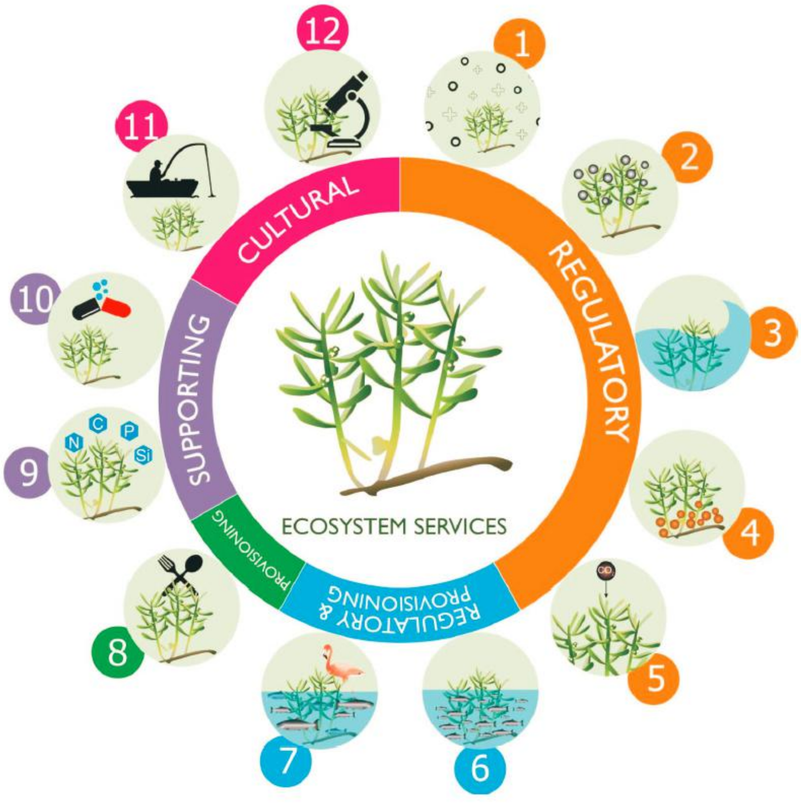

Figure 1.1 Ecosystem services of tidal wetlands: 1. Bio-filter for nutrients; 2. Trap for pollutants; 3. Buffer against waves and storms; 4. Sediment trapping and shoreline stabilization; 5. Carbon sequestration; 6. Nursery for fishes; 7. Breeding and feeding grounds; 8. Raw material and food; 9. Nutrient source for biota; 10. Medicinal use; 11. Recreation; 12. Research.

In summary, coastal habitats provide such ecosystem services essential to people and the environment. These services are valued at billions of dollars, services provided by coastal wetlands including the following aspects: ***Flood Protection & Erosion control***: Coastal wetlands protect upland areas, including valuable residential and commercial property, from flooding due to sea level rise and storms. Coastal wetlands can also prevent coastline erosion due to their ability to absorb the energy created by ocean currents which would otherwise degrade a shoreline and associated development; ***Wildlife Food & Habitat***: Coastal wetlands provide habitat for many federally threatened and endangered species; ***Commercial Fisheries***: Over 50 percent of commercial fish and shellfish species rely on coastal wetlands; ***Water Quality:*** Wetlands filter chemicals and sediment out of water before it is discharged into the ocean; ***Recreation***: Recreational opportunities in coastal wetlands include canoeing, kayaking, wildlife viewing and photography, recreational fishing and hunting; ***Carbon Sequestration***: Certain coastal wetland ecosystems (such as salt marshes and mangroves) can sequester and store large amounts of carbon due to their rapid growth rates and slow decomposition rates. Here, we elaborate these ecosystem services for the main kinds of ecosystems which included salt marshes, mangroves, coral reefs and seagrass beds.

(1) Salt marshes

Salt marshes are intertidal grasslands that form in low-energy, wave-protected shorelines along continental margins (Mitsch & Gosselink, 2008). Extensive salt marshes can establish and grow both behind barrier-island systems and along the wave-protected shorelines of bays and estuaries. Salt marshes are characterized by sharp zonation of plants and low species diversity, but extremely high primary and secondary production. The structure and function of salt marsh plant communities (and thus their services) were long thought to be regulated by physical processes, such as elevation, salinity, flooding, and nutrient availability. Among coastal ecosystems, salt marshes provide a high number of valuable benefits to humans, including raw materials and food, coastal protection, erosion control, water purification, maintenance of fisheries, carbon sequestration, and tourism, recreation, education, and research (Table 1.3).

Table 1.3 Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for salt marshes.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials and food	generates biological productivity and diversity	vegetation type and density, habitat quality, inundation depth, habitat quality, healthy predator populations	£15.27· ha^{-1}·yr^{-1} net income from livestock grazing, UK (King and Lester 1995)	marsh reclamation, vegetation disturbance, climate change, sea level rise, pollution, altered hydrological regimes, biological invasion
Coastal protection	attenuates and/or dissipates waves	tidal height, wave height and length, water depth in or above canopy, marsh area and width, wind climate, marsh species and density, local geomorphology	US\$8236·ha^{-1}·yr^{-1} in reduced hurricane damages, USA (Costanza et al. 2008)	
Erosion control	provides sediment stabilization and soil retention in vegetation root structure	sea level rise, tidal stage, coastal geomorphology, subsidence, fluvial sediment deposition and load, marsh grass species and density, distance from sea edge	estimates unavailable	
Water purification	provides nutrient and pollution uptake, as well as retention, particle deposition	marsh grass species and density, marsh quality and area, nutrient and sediment load, water supply and quality, healthy predator populations	US\$785–15 000/acre capitalized cost savings over traditional waste treatment, USA (Breaux et al. 1995)†	
Maintenance of fisheries	provides suitable reproductive habitat and nursery grounds, sheltered living space	marsh grass species and density, marsh quality and area, primary productivity, healthy predator populations	US\$6471/acre and \$981/acre capitalized value for recreational fishing for the east and west coasts, respectively, of Florida, USA (Bell 1997) and \$0.19–1.89/acre marginal value product in Gulf Coast blue crab fishery, USA (Freeman 1991)†	
Carbon sequestration	generates biogeochemical activity, sedimentation, biological productivity	marsh grass species and density, sediment type, primary productivity, healthy predator populations	US\$30.50·ha^{-1}·yr^{-1}‡	
Tourism, recreation, education, and research	provides unique and aesthetic landscape, suitable habitat for diverse fauna and flora	marsh grass species and density, habitat quality and area, prey species availability, healthy predator populations	£31.60/person for otter habitat creation and £1.20/person for protecting birds, UK (Birl and Cox 2007)	

† One acre = 0.4 ha.
‡ Based on Chumra et al. (2003) estimate of permanent carbon sequestration by global salt marshes of 2.1 Mg C·ha^{-1}·yr^{-1} and 23 September 2009 Carbon Emission Reduction (CER) price of the European Emission Trading System (ETS) of €12.38/Mg, which was converted to US\$2000.

For over 8000 years, humans have relied on salt marshes for direct provisioning of raw materials and food (Davy *et al.*, 2009). Although harvesting of marsh grasses and use of salt marshes as pasture lands has decreased today, these services are still important locally in both developed and developing areas of the world (Gedan *et al.*, 2009).

Moreover, salt marshes have provided coastal protection from waves and storm surge, as well as from coastal erosion, for humans during thousands of years (Davy *et al.*, 2009). By stabilizing sediment, increasing the intertidal height, and providing baffling vertical structures (grass), salt marshes reduce impacts of incoming waves by reducing their velocity, height, and duration (Morgan *et al.*, 2009). Marshes are also likely to reduce storm surge duration and height by providing extra water uptake and holding capacity in comparison to the sediments of mudflats. This storm protection value can be substantial, as a study of the protection against hurricanes by coastal wetlands along the U.S. Atlantic and Gulf coasts reveals (Costanza *et al.*, 2008). What is more, salt marshes act as natural filters that purify water entering the estuary (Mitsch & Gosselink, 2008). As water passes through marshes, it slows due to the baffling and friction effect of upright grasses (Morgan *et al.*, 2009). Suspended sediments are then deposited on the marsh surface, facilitating nutrient uptake by salt marsh grasses. This water filtration service benefits human health, but also adjacent ecosystems, such as seagrasses, which may be degraded by nutrients and pollutants.

Salt marsh ecosystems serve to maintain fisheries by boosting the production of economically and ecologically important fishery species, such as shrimp, oysters, clams, and fishes (MacKenzie & Bruland, 2012). For example, salt marshes may account for 66% of the shrimp and 25% of the blue crab production in the Gulf of Mexico (Zimmerman *et al.*, 2000). Because of their complex and tightly packed plant structure, marshes provide habitat that is mostly inaccessible to large fishes, thus providing protection and shelter for the increased growth and survival of young fishes, shrimp, and shellfish. As one of the most productive ecosystems in the world (up to 3900 g C m^{-2} yr^{-1}), salt marshes sequester millions of tons of carbon annually (Mitsch & Gosselink, 2008). Because of the anoxic nature of the marsh soils (as in most wetlands), carbon sequestered by salt marsh plants during photosynthesis is often shifted from the short-term carbon cycle (10–100 years) to the long-term carbon cycle (1000 years) as buried, slowly decaying biomass in the form of peat (Mitsch & Gosselink, 2008, Morgan *et al.*, 2009). This cycle shifting capability is unique among many of the world’s ecosystems, where carbon is mostly turned over quickly and does not often move into the long-term carbon cycle.

However, current human threats to salt marshes include biological invasions, eutrophication, climate change and sea level rise, increasing air and sea surface temperatures, increasing CO_2 concentrations, altered hydrologic regimes, marsh reclamation, vegetation disturbance, and pollution (Marsden, 2012). As indicated in Table 1.4, a growing number of valuable marsh services are lost with the destruction of this habitat. Approximately 50% of the original salt marsh ecosystems have been degraded or lost globally (Gedan *et al.*, 2009).

(2) Mangroves

Mangroves are coastal forests that inhabit saline tidal areas along sheltered bays, estuaries, and inlets in the tropics and subtropics throughout the world. In the 1970s, mangroves may have covered as much as 200 000 km^2, or 75% of the world's coastlines (Spalding *et al.*, 1997). But since then, at least 35% of global mangrove area has been lost, and mangroves are currently disappearing at the rate of 1–2% annually (Alongi & Daniel, 2002, IMIS, 2008, Valiela *et al.*, 2001). The worldwide destruction of mangroves is of concern because they provide a number of highly valued ecosystems services, including raw materials and food, coastal protection, erosion control, water purification, maintenance of fisheries, carbon sequestration, and tourism, recreation, education, and research (Table 1.4). For many coastal communities, their traditional use of mangrove resources is often closely connected with the health and functioning of the system, and thus this use is often intimately tied to local culture, heritage, and traditional knowledge (Walters *et al.*, 2009).

Of the ecosystem services listed, three have received most attention in terms of determining their value to coastal populations. These include (1) their use by local coastal communities for a variety of products, such as fuel wood, timber, raw materials, honey and resins, and crabs and shellfish; (2) their role as nursery and breeding habitats for offshore fisheries; and (3) their propensity to serve as natural ‘‘coastal storm barriers’’ to periodic wind and wave or storm surge events, such as tropical storms, coastal floods, typhoons, and tsunamis.

Table 1.4 Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for mangroves.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials and food	generates biological productivity and diversity	vegetation type and density, habitat quality	US\$484–585·ha^{-1}·yr^{-1} capitalized value of collected products, Thailand (Barbier 2007)	mangrove disturbance, degradation, conversion; coastline disturbance; pollution; upstream soil loss; overharvesting of resources
Coastal protection	attenuates and/or dissipates waves and wind energy	tidal height, wave height and length, wind velocity, beach slope, tide height, vegetation type and density, distance from sea edge	US\$8966–10 821/ha capitalized value for storm protection, Thailand (Barbier 2007)	
Erosion control	provides sediment stabilization and soil retention in vegetation root structure	sea level rise, tidal stage, fluvial sediment deposition, subsidence, coastal geomorphology, vegetation type and density, distance from sea edge	US\$3679·ha^{-1}·yr^{-1} annualized replacement cost, Thailand (Sathirathai and Barbier 2001)	
Water purification	provides nutrient and pollution uptake, as well as particle retention and deposition	mangrove root length and density, mangrove quality and area	estimates unavailable	
Maintenance of fisheries	provides suitable reproductive habitat and nursery grounds, sheltered living space	mangrove species and density, habitat quality and area, primary productivity	US\$708-\$987/ha capitalized value of increased offshore fishery production, Thailand (Barbier 2007)	
Carbon sequestration	generates biological productivity, biogeochemical activity, sedimentation	vegetation type and density, fluvial sediment deposition, subsidence, coastal geomorphology	US\$30.50·ha^{-1}·yr^{-1}†	
Tourism, recreation, education, and research	provides unique and aesthetic landscape, suitable habitat for diverse fauna and flora	mangrove species and density, habitat quality and area, prey species availability, healthy predator populations	estimates unavailable	

† Based on Chumra et al. (2003) estimate of permanent carbon sequestration by global salt marshes of 2.1 Mg C·ha^{-1}·$yr^{-1}$and 23 September 2009 Carbon Emission Reduction (CER) price of the European Emission Trading System (ETS) of €12.38/Mg, which was converted to US\$2000.

Since the 2004 Indian Ocean Tsunami, there has been considerable global interest in one particular service of mangroves: their role as natural barriers that protect the lives and properties of coastal communities from periodic storm events and flooding. Eco-hydrological evidence indicates that this protection service is based on the ability of mangroves to attenuate waves and thus reduce storm surge (Walters *et al.*, 2009). Moreover, the ability of mangroves to stabilize sediment and retain soil in their root structure reduces shoreline erosion and deterioration (Barbier & Sathirathai, 2001, Thampanya *et al.*, 2006). Mangroves also serve as barriers in the other direction; their water purification functions protect coral reefs, seagrass beds, and important navigation waters against siltation and pollution. In southern China, field experiments have been conducted to determine the feasibility of using mangrove wetlands for wastewater treatment (Chen *et al.*, 2009). Mangrove roots may also serve as a sensitive bio-indicator for metal pollution in estuarine systems (Macfarlane *et al.*, 2003). The economic value of the pollution control service of mangroves has not been reliably estimated, however. Because mangroves are among the most productive and biogeochemically active ecosystems, they are important sources of global carbon sequestration.

Although many factors contribute to global mangrove deforestation, a major cause is aquaculture expansion in coastal areas, especially the establishment of shrimp farms (Barbier & Cox, 2001). Aquaculture accounts for 52% of mangrove loss globally, with shrimp farming alone accounting for 38%. Forest use, mainly from industrial lumber and woodchip operations, causes 26% of mangrove loss globally. Freshwater diversion accounts for 11% of deforestation, and reclamation of land for other uses causes 5% of decline. The remaining sources of mangrove deforestation consist of herbicide impacts, agriculture, salt ponds, and other coastal developments (Valiela *et al.*, 2001). The extensive and rapid loss of mangroves globally reinforces the importance of measuring the value of such ecological services, and employing these values appropriately in coastal management and planning.

(3) Coral reefs

Coral reefs are structurally complex limestone habitats that form in shallow coastal waters of the tropics. Reefs can form nearshore and extend hundreds of kilometers in shallow offshore environments. Coral reefs are created by sedentary cnidarians (corals) that accrete calcium carbonate and feed on both zooplankton and maintain a mutualistic symbiosis with photosynthetic dinoflagellates. Thus, the majority of the reef structure is dead coral skeleton laid down over millennia, covered by a thin layer of live coral tissue that slowly accretes new limestone. In addition, coralline algae play an important role in stabilizing and cementing the coral reef structure. The community composition of reefs depends on global, regional, and local factors, which interact to produce the wide variety of coral reefs present on earth (Hughes *et al.*, 2005). Coral reefs provide a number of ecosystem services to humans including raw materials, coastal protection, maintenance of fisheries, nutrient cycling, and tourism, recreation, education, and research.

An important ecosystem service provided by coral reefs is coastal protection or the buffering of shorelines from severe weather, thus protecting coastal human populations, property, and economic activities. As indicated in Table 1.5, this service is directly related to the economic processes and functions of attenuating or dissipating waves and facilitating beach and shoreline retention. By altering the physical environment (i.e., reducing waves and currents), corals can engineer the physical environment for entire ecosystems, making it possible for other coastal ecosystems such as seagrass beds and mangroves to develop, which in turn serve their own suite of services to humans.

Coral reefs also serve to maintain fisheries through the enhancement of ecologically and economically important species by providing shelter space and substrate for smaller organisms, and food sources for larger epibenthic and pelagic organisms. Increases in fishing technology and transport have transformed reef fisheries that initially functioned solely for subsistence into commercial operations that serve international markets. Coral reef fisheries consist of reef-associated pelagic fisheries (e.g., tuna, mackerel, mahi-mahi, and sharks), reef fishes (e.g., jacks, snappers, groupers, and parrot fishes), and large invertebrates (e.g., giant clams, conch, lobsters, and crabs). The commercial value of these fisheries can be significant for some economies.

Table 1.5 Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for coral reefs.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials	generates biological productivity and diversity	reef size and depth, coral type, habitat quality	estimates unavailable	climate change, blast or cyanide fishing, lime mining, eutrophication, sedimentation, coastal development, dredging, pollution, biological invasion
Coastal protection	attenuates and/or dissipates waves, sediment retention	wave height and length, water depth above reef crest, reef length and distance from shore, coral species, wind climate	US\$174·ha^{-1}·yr^{-1} for Indian Ocean based on impacts from 1998 bleaching event on property values (Wilkinson et al. 1999)	
Maintenance of fisheries	provides suitable reproductive habitat and nursery grounds, sheltered living space	coral species and density, habitat quality, food sources, hydrodynamic conditions	US\$15–45 000·km^{-2}·yr^{-1} in sustainable fishing for local consumption and \$5–10 000·km^{-2}·yr^{-1} for live-fish export, the Philippines (White et al. 2000)	
Nutrient cycling	provides biogeochemical activity, sedimentation, biological productivity	coral species and density, sediment deposition, subsidence, coastal geomorphology	estimates unavailable	
Tourism, recreation, education, and research	provides unique and aesthetic reefscapes, suitable habitat for diverse fauna and flora	lagoon size, beach area, wave height, habitat quality, coral species and density, diversity	US\$88 000 total consumer surplus for 40 000 tourists to marine parks, Seychelles (Mathieu et al. 2003) and meta-analysis of recreational values (Brander et al. 2007)	

Coral reef ecosystems also perform important services by cycling organic and inorganic nutrients. Despite housing a great deal of inorganic carbon in the limestone skeleton that makes up the structure of the reef, coral reefs may actually be a net source of atmospheric carbon dioxide (Kawahata *et al.*, 1997). Reefs do, however, contribute significantly to the global calcium carbonate ($CaCO_3$) budget, estimated as 26% of coastal marine $CaCO_3$ and 11% of the total $CaCO_3$ precipitation. Reefs additionally transfer excess nitrogen production from cyanobacteria and benthic microbes on the reef to the pelagic (water column) environment (Moberg & Rönnbäck, 2003). Though poorly quantified, the sequestering of $CaCO_3$ to form the foundation or habitat of the reef is the primary reason for such high abundance and diversity of organisms. In addition to tourism and recreation, reefs also provide substantial services through research opportunities for scientists, work that is essential to basic and applied science (Benjaminj & Johnm, 2008).

Despite the numerous economic benefits and ecosystem services coral reefs provide, reef ecosystems are under threat of irrevocable decline worldwide from a suite of anthropogenic stressors. Localized stressors (i.e., within reefs or archipelagos) include overfishing, dynamite or cyanide fishing, pollution, mining, eutrophication, coastal development, dredging, sedimentation, and biological invasion. A variety of reef ecosystem services may be affected by coral degradation.

(4) Seagrass beds

Seagrasses are flowering plants that colonize shallow marine and estuarine habitats. With only one exception (the genus Phyllospadix), seagrasses colonize soft substrates (e.g., mud, sand, cobble) and grow to depths where 11% of surface light reaches the bottom. Seagrasses prefer wave-sheltered conditions as sediments disturbed by currents and/or waves lead to patchy beds or their absence (Koch *et al.*, 2006). Despite being among the most productive ecosystems on the planet, fulfilling a key role in the coastal zone (Duarte, 2002) and being lost at an alarming rate (Orth *et al.*, 2006), seagrasses receive little attention when compared to other coastal ecosystems (Duarte *et al.*, 2008).

Seagrass beds provide a wide range of ecosystem services, including raw materials and food, coastal protection, erosion control, water purification, maintenance of fisheries, carbon sequestration, and tourism, recreation, education, and research. Although in the past seagrasses were highly valued as raw materials and food, modern direct uses of seagrasses are rather limited. Coastal protection and erosion control are often listed as important ecosystem services provided by seagrasses (Koch *et al.*, 2009). Seagrasses can attenuate waves and, as a result, smaller waves reach the adjacent shoreline. Coastal protection is highest when the plants occupy the entire water column, such as at low tide, or when plants produce long reproductive stems (Koch *et al.*, 2006). When small seagrasses colonize deeper waters, their contribution to wave attenuation and coastal protection is more limited. Sediment stabilization by seagrass roots and rhizomes, as well as by their beach-casted debris is important for controlling coastal erosion.

Table 1.6 Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for seagrasses.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials and food	generates biological productivity and diversity	vegetation type and density, habitat quality	estimates unavailable	eutrophication, overharvesting, coastal development, vegetation disturbance, dredging, aquaculture, climate change, sea level rise
Coastal protection	attenuates and/or dissipates waves	wave height and length, water depth above canopy, seagrass bed size and distance from shore, wind climate, beach slope, seagrass species and density, reproductive stage	estimates unavailable	
Erosion control	provides sediment stabilization and soil retention in vegetation root structure	sea level rise, subsidence, tidal stage, wave climate, coastal geomorphology, seagrass species and density	estimates unavailable	
Water purification	provides nutrient and pollution uptake, as well as retention, particle deposition	seagrass species and density, nutrient load, water residence time, hydrodynamic conditions, light availability	estimates unavailable	
Maintenance of fisheries	provides suitable reproductive habitat and nursery grounds, sheltered living space	seagrass species and density, habitat quality, food sources, hydrodynamic conditions	loss of 12 700 ha of seagrasses in Australia; associated with lost fishery production of AU$235 000 (McArthur and Boland 2006)	
Carbon sequestration	generates biogeochemical activity, sedimentation, biological productivity	seagrass species and density, water depth, light availability, burial rates, biomass export	estimates unavailable	
Tourism, recreation, education, and research	provides unique and aesthetic submerged vegetated landscape, suitable habitat for diverse flora and fauna	biological productivity, storm events, habitat quality, seagrass species and density, diversity	estimates unavailable	

Water purification, or the increase in water clarity, by seagrasses occurs via two processes: nutrient uptake and suspended particle deposition. Seagrasses not only remove nutrients from the sediments and water column, but also their leaves are colonized by algae (epiphytes), which further remove nutrients from the water column (Cornelisen & Thomas, 2006). The nutrients incorporated into the tissue of seagrasses and algae are slowly released back into the water column once the plants decompose or are removed from the nutrient cycle when buried in the sediment (Romero *et al.*, 2006). In addition to reducing nutrients, seagrass beds also decrease the concentration of suspended particles (e.g., sediment and microalgae) from the water. Leaves in the water column provide an obstruction to water flow and, as a result, currents and waves are reduced within seagrass canopies causing particles to be deposited (Koch *et al.*, 2006). This water purification effect can be quite dramatic with clearer water in vegetated areas compared to those without vegetation. Seagrasses also generate value as habitat for ecologically and economically important species such as scallops, shrimp, crabs, and juvenile fish. Seagrasses protect these species from predators and provide food in the form of leaves, detritus, and epiphytes (Barbier, 2011).

1.3 Status and changes of coastal wetland in Yellow Sea

In the Yellow Sea region of East Asia, tidal wetlands are the frontline ecosystem protecting a coastal population of more than 60 million people from storms and sea-level rise. According to historical topographic maps, tidal wetlands in the Yellow Sea occupied 1.12 and 0.55 million ha in the mid-1950s and 1980s, respectively, only 0.39 million ha remained in 2000s (Murray *et al.*, 2014). In the late 2000s, the tidal wetland distributed in China, North Korea, and South Korea along with Yellow Sea coastline is 161 066, 107765, and 120 472 ha, respectively (Figure 1.2, Table 1.7).

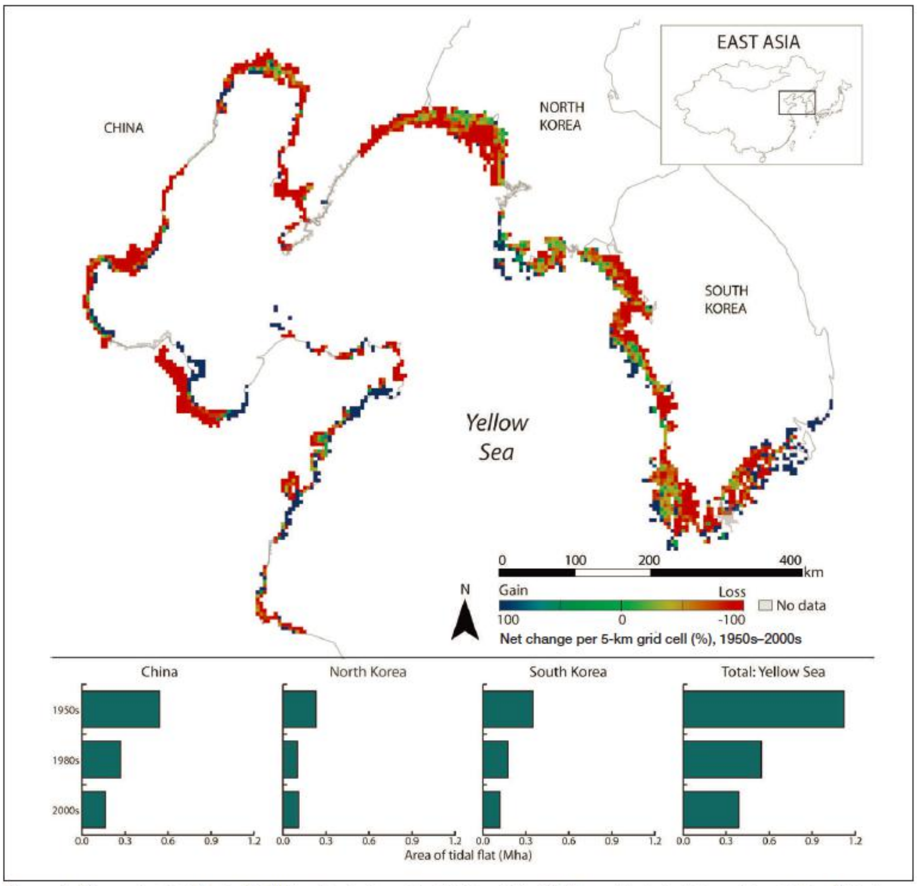

Figure 1. Change in tidal flats in the Yellow Sea between the 1950s and the 2000s, mapped at a 5-km grid resolution. Net change between the two time periods is shown on a color ramp from blue (total gain) to red (total loss).

Figure 1.2 Wetland ecosystem distribution and changes in Yellow Sea between the 1950s and the 2000s.

However, with the region forecast to be a global hotspot of urban expansion, extensive losses of the region's principal coastal ecosystem-tidal flats-associated with urban, industrial, and agricultural land reclamations. The research about coastal wetland area changes in Yellow Sea area had proved that 28% of tidal flats existing in the 1980s had disappeared by the late 2000s, at the mean rate of -1.2% (Murray *et al.*, 2014). Moreover, reference to historical maps suggests that up to 65% of tidal flats were lost over the past five decades (Table 1.7). At the country level, China lost more tidal flat area and at a faster rate (39.8%, -1.8% yr^{-1}) than South Korea (32.2%, -1.6% yr^{-1}), on the contrary, the minor gains of tidal flats occurred in North Korea (8.5%, 0.3% yr^{-1}). Previous results proved that up to two-thirds of the tidal flats existing around the Yellow Sea in the 1950s have since vanished, with losses in China and South Korea accounting for most of the decline (Figure 1.2, Table 1.7)(Murray *et al.*, 2014).

Table 1.7 Tidal flat area and rates of change by country, 1950s-2000s.

	Estimated area of tidal flat (ha)			*% change*			*Continuous rate of change (% yr^{-1})*		
	1950s	*1980s*	*2000s*	*1950s–1980s*	*1980s–2000s*	*1950s–2000s*	*1950s–1980s*	*1980s–2000s*	*1950s–2000s*
China	539 794	267 751	161 066	–50.4	–39.8	–70.2	–2.7	–1.8	–2.2
North Korea	231 813	99 333	107 765	–57.1	8.5	–53.5	–4.9	0.3	–1.6
South Korea	350 331	177 729	120 472	–49.3	–32.2	–65.6	–2.4	–1.6	–2.0
Yellow Sea	1 121 938	544 812	389 303	–51.4	–28.0	–65.3	–3.0	–1.2	–2.0

Notes: Area estimates should be considered minima for the Yellow Sea, because 12.1% of the coastline could not be mapped owing to the presence of cloud or ice cover in satellite imagery obtained at suitable tide heights (Figure 1).

In general, losses of tidal flats in Yellow Sea were spatially pervasive, occurring throughout heavily populated and rapidly developing coastal areas and increased in extent only occurred in a few isolated locations (Figure 1.2). Much of the Yellow Sea coastline is under intense pressure from reclamation for agriculture, aquaculture, and industrial development. For example, Caofeidian port development in Tianjin, China and Saemangeum port development in South Korea are among the largest reclamation projects on East Asia, which were 31 000 ha and 40 100 ha, respectively (Mackinnon *et al.*, 2012). Similarly, the conversion of tidal flats to aquaculture ponds is widespread in the Yellow Sea and, with Asia currently supplying

89% of global aquaculture production, further reclamation of tidal flats will be required to meet increasing demand (Naylor *et al.*, 2000). The data indicates that tidal flats along the Yellow Sea are declining at a rate comparable to many other at risk ecosystems, such as tropical forests (Achard *et al.*, 2002). Degradation and reclamation of coastal wetlands are worldwide phenomena (Hassan *et al.*, 2005) and are likely to intensify, owing to the increasing scarcity of land in coastal areas and the low cost and rapid pace at which these areas can be developed (Mackinnon *et al.*, 2012).

1.4 Wetland management and restoration in Yellow sea area

Coastal wetland loss and degradation was considered as a global problem in the present day, and they faced a global loss of about 50% in the last decades, most of which was transformed into mariculture ponds (Valiela *et al.*, 2009). Land reclamation has been the main reason of coastal wetland loss in China in the last decades. Surrounding the Bohai Sea, Northern China, more than 2000 km^2 were reclaimed between 1980 and 2012 (Li *et al.*, 2018). Using new technology for land reclamation, muddy water can be pumped into the levee and new land can be created within a few months. This has happened along almost all of the Chinese coast wherever muddy subsurface is available, driven by rapid economic development. Loss of wetland area means the loss of its corresponding ecological services for human beings, with economic gain only for a special group of people making profits from aquaculture or other land use forms (Worm *et al.*, 2006). Profound changes have been caused by conversion of coastal wetlands into other land use forms with damage from flooding or other indirect influences (Worm *et al.*, 2006) (Figure 1.3).

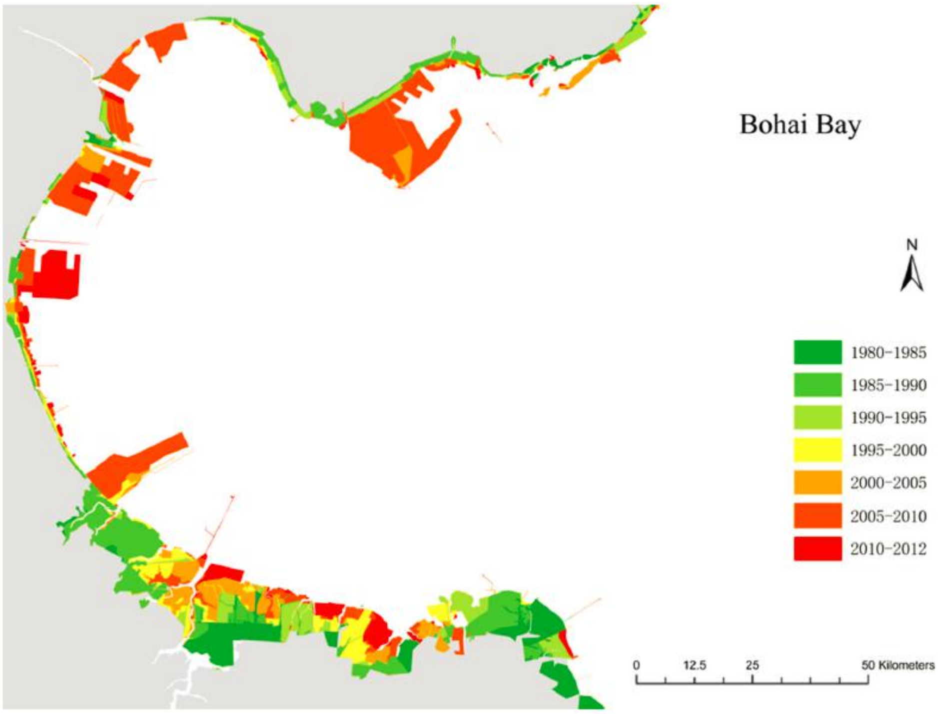

Figure 1.3. Land reclamation around the Bohai Bay, China, between 1980 and 2012.

While coastal wetlands are facing loss, many countries have started taking measures to rebuild marshes with dredged sediments, or divert the river channels to elevate ground surface. Some new concepts, such as "building with nature" (Vriend *et al.*, 2015), "living shore lines" (Rose *et al.*, 2015), or "blue forest", have also been proposed and implemented in different parts of the world to support practices for coastal wetland restoration.

However, there was still a long way for the coastal wetland restoration, because lacking of comprehensive understanding of coastal ecosystems. Knowledge gaps still exist for successful coastal wetland restoration. Geomorphic units where coastal wetlands can develop are often complicated, from high tidal zones to low lands, lagoons, and tidal creeks. They are ever-shifting due to human activities and natural processes, and there is insufficient real-time monitoring for critical processes, such as water and sediment redistribution, subsidence, or ecosystem dynamics. Sediment budgets and ratios of mud, sand, and organic matter in the soil of deltaic plain are generally unknown, yet are crucial for preventing drowning (Giosan *et al.*, 2014).

From an academic point of view, the challenge for coastal wetlands restoration is also a great opportunity for landscape ecologists to transfer their knowledge into practice. For example, how to compromise between the different functionalities of coastal wetlands, such as *S. alterniflora*? Its functions for coastal protection and sediment trapping are considered to be positive along the Chinese coastline (Yang, 1998, Yang *et al.*, 2008), but as an invasive species, it has negative effects on the native organisms (He *et al.*, 2012), especially in southern China (Shu *et al.*, 2014). In contrast, in the middle Atlantic coast of the US, the invasive *Phragmites australis* proved to be more effective in combating sea level rise with higher mineral and organic sediment trapping ability than the local *S. alterniflora* (Rooth & Stevenson, 2000).

Also, lack of knowledge about site-specific bio-morphological interactions is also a great challenge for the coastal wetland restoration. For vegetation restoration in the tidal zone, it is necessary to consider how the ecosystem will interact with the physical environment. Waves and sediments will be redistributed and attenuated differently by different species, also depending on their density and biomass, thus changing the landforms, which, in turn, will affect the vegetation diversity and distribution (Leonardi & Fagherazzi, 2015). Yet the knowledge for the mechanism of bio-morphological interaction is rather limited, and is often site-specific in terms of tidal ranges, wave energy, salinity gradients, suspended sediment contents, morphological conditions, and species structure. General interpretations of marsh mechanisms obtained at large scale also need site-dependent data input to support successful rehabilitation (Marani *et al.*, 2011).

Coastal wetland restoration practice should also be “site-specific”. The location, species, size, and spatial orientation of the wetland must be carefully considered according to the tide and substrate conditions. Successful restoration requires both semi-natural vegetation structure and a high diversity of fauna groups, to ensure multi-functionality of the restored ecosystem. More importantly, when restoring the ecosystem occupied by aquaculture, the income of local people should not be reduced. New benefits for those people must be explored, such as ecotourism, apiculture, and horticulture. Incorporating ecosystem services into coastal planning will achieve greater returns from coastal protection and tourism than from achieving conservation or development goals only (Arkema *et al.*, 2015). Moreover, while faced with quick economic development along the world coast zones, we also need to make room for potential sea-level rise. By combining conventional engineering with ecosystem-based engineering, we may mitigate potential big flooding risks in the long run (Temmerman & Kirwan, 2015, Tessler *et al.*, 2015), and provide important habitats for numerous wild and commercial species (Cui *et al.*, 2009, Jiang *et al.*, 2015, Li *et al.*, 2018).

Chapter 2. Coastal wetlands distribution in Yellow Sea

2.1 Coastal wetlands in China

Healthy coastal wetland ecosystems play an important role in guaranteeing the territory ecological security and the sustainable development of coastal zone in China. China has approximately 5.80×10^6 ha coastal wetlands by estimated in 2014, accounting for 10.82% of the total area of natural wetlands. According to the second national wetland resources survey conducted during 2009–2013, the total area of natural wetlands were as followed: 21.73×10^6 ha are marshes and swamps, 8.59×10^6 ha are lakes, 10.55×10^6 ha are rivers, and 5.80×10^6 ha are coastal wetlands, which account for 40.68%, 16.09%, 19.75% and 10.85% of the total wetlands area, respectively (Figure 2.1)(Mou *et al.*, 2015, Sun *et al.*, 2015, Wang *et al.*, 2014). The coastal wetlands are mainly distributed in 11 provinces of Liaoning, Hebei, Tianjin, Shandong, Jiangsu, Shanghai, Zhejiang, Fujian, Guangdong, Guangxi and Hainan, in which the area of coastal wetlands of Shandong and Guangdong provinces accounts for 37.55% of the total coastal wetlands area (Figure 2.1). Coastal wetlands in China generally can be divided into two groupings. One part is located to the north of the Hangzhou Bay. In this part, the Bohai Sea coast and the Jiangsu coast have sandy or silty wetlands while the Liaodong Peninsula and Shandong Peninsula have rocky beaches. The other part is located to the south of the Hangzhou Bay. In this part, the coasts are mainly rocky, including the major river deltas, such as the Yangtze River Delta, the Qiantang River–Hangzhou Bay, the Jin River estuary–Quanzhou Bay, the Pearl River Delta and the North (Beibu) Gulf (Niu *et al.*, 2009).

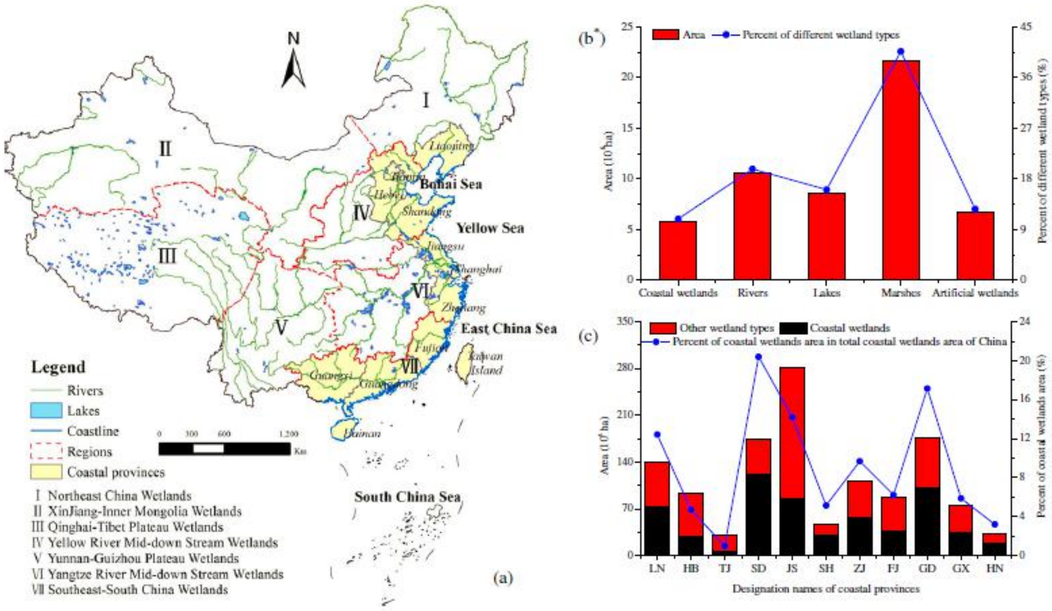

Figure 2.1 Distribution of China's coastal wetlands and the area and the percent of coastal wetland across the country or in different coastal provinces.

Over the past 60 years, China's coastal wetlands have suffered tremendous loss due to the increased threats and pressures on wetlands for the reason of the increasing population and rapidly developing economy. From 1950 to 2014, the country lost 8.01 × 10^6 ha coastal wetlands, with a total loss rate of 58.0%. Numerous factors endanger the existence of coastal wetlands, such as land demands by a large population, a lack of understanding of coastal wetland values, a misguided reclamation policy and a lack of environmental laws and regulations. Among them, reclamation and infrastructure construction were the primary causes, which account for 70–82% of the total loss (Niu et al., 2011; An et al., 2007). Approximately 3.86 × 10^6 ha coastal lands, including swamp, salt marsh, estuary, gulf and mangroves, were reclaimed or greatly destroyed in the past 20 years (1991–2014), which approximated the lost area during 1950–1991 (4.15 × 10^6 ha) (Figure 2.1). Because the statistical calibers of coastal wetlands were different in the two national wetland resources surveys (over 100 ha in the first national wetland resources survey and over 8 ha in the second national wetland resources survey), the approximate area of coastal wetlands in the two periods, to some extent, covered the actual loss status of coastal wetlands in the past decade.

It was found that the coastal wetlands in the Luan River estuary (Hebei), Binhai New Area

(Tianjin), Laizhou Bay (Shandong), Yancheng (Jiangsu), Hangzhou Bay (Zhejiang), Min River estuary (Fujian), Pearl River Delta (Guangdong) and Beibu Gulf (Guangxi) showed significant decrease status,while those in the Qilihai (Hebei), Xiamen Bay (Fujian), Jiaozhou Bay and Yellow River Delta (Shandong) presented slow descending tendency. Differently, the coastal wetlands in the Liao River Delta (Liaoning), Yangtze River estuary (Shanghai) and Dongzhaigang (Hainan) decreased greatly before 2007 and increased slightly during 2007–2013 due to the implementation of large-scale restoration projects (Sun *et al.*, 2015). The mangroves, once widely distributed in southeast China, lost approximately 3.34×10^4 ha during 1950–1997 due to the unreasonable utilization and reclamation, with a total loss rate of 69.15%. Especially, the loss rates in Guangdong, Hainan, Fujian and Guangxi provinces over the same period reached 82.1%, 51.6%, 50.0% and 43.5%, respectively (He and Fan, 1995; Zhang et al., 1997). From 1997 to 2008, although the Chinese government gradually realized the importance of mangrove conservation and took a series of effective actions, the area of mangroves increased slightly. Since then, with the implementation of large-scale mangrove restoration projects, the area of mangroves has already been recovered to the level in 1980.

Nonetheless, several major issues recently emerged in China's coastal wetland conservation are evidently existed, including (1) the increasing threats of pollution and human activities, (2) the increasing adverse effects of threaten factors on ecosystem function, (3) the increasing threats of coastal erosion and sea-level rising, (4) the insufficient funding for coastal wetlands conservation, (5)the imperfect legal and management system for coastal wetlands, and (6) the insufficient education, research and international cooperation. Although the threats and pressures on coastal wetlands conservation are still apparent, the future of China's coastal wetlands looks promising since the Chinese government understands that the sustainable development in coastal zone requires new attitudes, sound policies and concerted efforts at all levels. The major strategies for future improvement of China's coastal wetland conservation include: (1) exploring effective measures in response to major threaten factors; (2) improving the conservation and compensation system for coastal wetlands; (3) strengthening coastal wetland legislation and management; (4) increasing funds for coastal wetland conservation and research; and (5) strengthening coastal wetland education and international cooperation.

2.2 Coastal wetlands in RoK

The Korean coastline is > 11,000 km long, including > 6000 km of mainland and > 5000 km of island coast (Sato & Koh, 2004). When compared with other islands or peninsular countries, Korea has a relatively high proportion of coastline to land surface area. The character of the Korean coast varies depending on location. The southwestern coast is diverse and intricate; the eastern coast has a relatively simple alignment and near-shore deep water. Tidal wetlands are primarily associated with the more diverse southwestern coast whereas the eastern coast mainly has sandy beaches. The tidal wetland system of the Yellow Sea is among the most extensive and ecologically important in the world. The coastal landscapes in southwestern Korea include a diverse array of tidal wetlands and salt marshes. These coastal zones link the ecological functions of marine tidal wetlands and freshwater ecosystems with terrestrial ecosystems. They are rich in biological diversity and play important roles in sustaining ecological health and processing environmental pollutants. Korean tidal wetlands are particularly important as nurseries for economically important fishes and habitats for migratory birds.

The area of Korean tidal wetlands was reduced from an estimated 2800 km^2 in 1987 to 2400 km^2 in 1997 (Sato & Koh, 2004). More than 26% (>1,600 km) of this coastline has been altered by construction of seawalls and other flood or erosion control facilities in recent decades. The loss was due to both urbanization and agricultural conversion. Although there have been several studies addressing these decreasing trends, none has provided a comprehensive overview or fully considered the ecological and socioeconomic consequences of this phenomenon. Diking, draining, tourism, and conversion to agricultural and urban uses have adversely affected Korean tidal wetlands. Recent large development projects have contributed to further losses. Environmental impact assessments conducted for projects affecting tidal wetlands and their surrounding landscapes should be customized for application to these special settings. Adequate environmental impact assessments will include classification of hydro-geomorphic units and consideration of their responses to biological and environmental stressors. As is true worldwide, Korean laws and regulations are changing to be more favorable to the conservation and protection of tidal wetlands. More public education needs to be done at the local level to build support for tidal wetland conservation. Some key public education points include the role of tidal wetlands in maintaining healthy fish populations and reducing impacts of nonpoint source pollution. There is also a need to develop procedures for integrating economic and environmental objectives within the overall context of sustainable management and land uses.

The population density of Korea is among the highest in the world. The South Korean government has also pursued aggressive economic growth policies. Consequently, there have been many instances of adverse environmental impact and degradation. One specific case in point concerns coastal ecosystems and tidal wetlands. Tidal wetlands in Korea have been drained and converted to urban uses at an alarming pace over the past decade (Koh, 1997). These areas have also experienced an increased flux of nonpoint source pollutants. All of this has been to the detriment of the diverse flora and fauna associated with the tidal wetlands. Tidal wetlands are both ecotones between marine and terrestrial ecosystems as well as ecosystems in and of themselves. It is their dual nature that makes them so important. Korean tidal wetlands have high potential biological productivity. They provide favorable habitat for resident organisms and migratory birds (*Egretta eulophotes* and *Platalea minor*) traveling from Siberia to Australia.

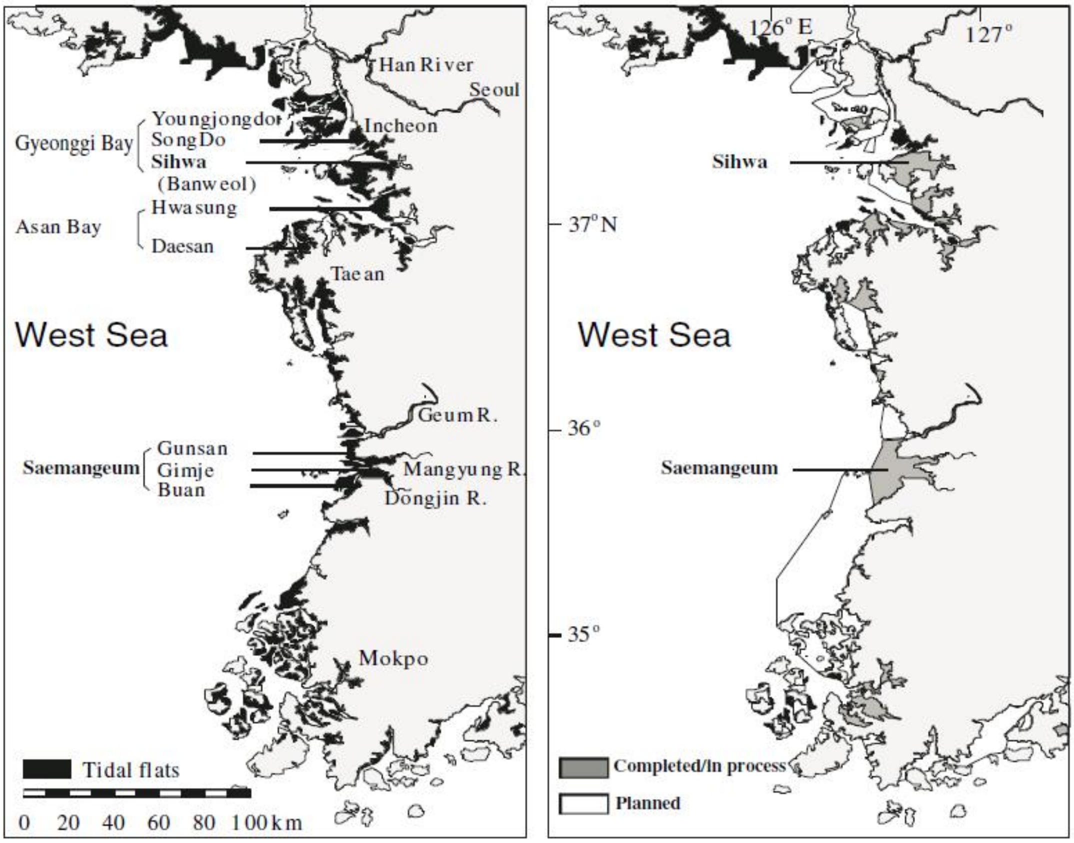

Figure 2.2. Detailed locations of tidal flat wetlands in western Korea and reclamation project planned.

Moreover, Korean tidal wetlands are mainly deltaic and associated with rivers. Consequently, they receive inputs of fresh water, nutrients, and sediments from upland watersheds as well as being tidally influenced. Estuarine tidal wetlands in Korea are dominated by Phragmites and halophytic plants, depending on their exposure to tidal influence. Tidal wetlands interspersed with shallow water and lagoons are particularly diverse. Where the substrate is sandy, the benthic community is well developed (MH *et al.*, 2000). Because benthic organisms facilitate the transfer of oxygen to subsurface sediments, they contribute to favorable growing conditions for plants. Microalgae (i.e., diatoms) may create a surface mat on tidal wetlands during low tide. In addition to being a food resource for wildlife, these mats provide habitat for marine invertebrates that are consumed by resident and migratory birds. Maintaining and enhancing the hydrologic and biological linkages between terrestrial and marine ecosystems inherent to tidal wetlands requires a landscape or regional management perspective. In Korea, this perspective is compatible with the current shift from a concentration on management of individual species to an emphasis on habitat ecosystem functions.

As of 1996, more than 25% of Korea's population lived within 6227 km of the terrestrial shoreline of South Korea. Approximately 810 km^2 of tidal wetland have been lost over the last 10 years as a result of projects such as the Shihwa-Lake Tidal Power Plant Construction Project and Saemangeum Reclamation Project. The population density in the developed coastal area is roughly 400 people/km^2, which is somewhat lower than the average for the country as a whole. People living on the coast generate a disproportionate share of the country's gross national product (nearly 42%). Economically important uses on the coast include fishing, agriculture, marine transportation, and aquaculture.

In their natural state, coastal landscapes and their tidal wetlands are dynamic, expanding and contracting over time in relation to natural disturbances. Changes in tidal influence due to floodwalls, diking, and draining curtail the basic process that sustains these wetland ecosystems (Walmsley, 2002). Reclamation projects and urban development have reduced the size and diversity of the coastal landscape and limited its potential for recovery. The ecological services of tidal wetlands that benefit humans such as retention and processing of pollutants, production of vegetable and animal human foods, and dampening of extreme weather events are accordingly reduced. In essence, adverse impacts on tidal wetlands have a cascading effect through the ecosystem that ultimately reaches the human population. Although the public interest in tidal wetlands has increased of late, tourism and other uses continue to cause damage to the remaining wetlands. In addition to these direct impacts, there are also indirect impacts associated with changes in drainage due to reclamation, thermal pollution from power plant discharges, harvesting of sand, and nonpoint source pollution from agricultural and urban runoff.

There is international recognition that tidal wetlands are ecologically, hydrologically, and socially essential. In Europe, this resulted in the Joint Declaration on the Protection of the Wadden Sea by Netherlands, Germany, and Denmark. There has been recent interest in restoring the tidal wetlands of Tokyo-Yokohama Bay. In the United States, protection and restoration of tidal wetlands have been mounting on both the Atlantic and Pacific Coasts and along the Gulf of Mexico (Turner *et al.*, 2004). The effects of Hurricane Katrina on the Gulf Coast and the lessons learned about wetland losses have raised awareness well beyond the borders of the United States. In 1999, Korea established the Wetland Conservation Act. In addition, the government has begun to designate Marine Protected Areas (MPAs) in order to effectively protect and manage the resources and culture of coastal and marine ecosystems. There is some concern, however, that the management and uses of these MPAs may not always be ecologically sound or conform to international standards. The ambiguous nature of Korea's legal, political, and managerial decision systems has emerged as an obstacle to the establishment of a comprehensive coastal management system. Decision-making is fragmented among several administrative agencies. There is a need for a more integrated environmental management system that incorporates the best of the related laws and institutions and precludes decisions contrary to ecological sustainability.

Improved management and protection of Korean tidal wetlands does not just depend on legal and institutional measures. There is a critical need to educate local governments, coastal residents, and the Korean populace as a whole about the importance of tidal wetlands. One strategy to reverse a pattern of degradation in tidal wetlands would be to assign social value to tidal wetlands. For example, preservation of indigenous sustainable industries on the coast, especially commercial fishing, can provide the impetus to protect natural processes in tidal wetlands. The most important commercial species depend upon these wetlands to survive. There is evidence that Korean society at large would like to preserve its fishing heritage. Korean citizens also need to understand that tidal wetlands perform extremely important functions as sinks and processors of nonpoint source pollutants. They act as buffers not only against the actions of the sea but also against the damage from urban runoff. More information should be provided to urban Koreans and decision-makers about these benefits. Creating public enthusiasm for the protection of tidal wetlands is a necessary first step. The legal mechanisms for preservation exist, for example, in the Korean Natural Environment Conservation Act. There does need to be a system of prioritizing competing areas for protection and/or restoration. The two socially relevant functions presented above—commercial fish nurseries and water purification— could be used as criteria for ranking and selecting sites or areas to preserve.

Chapter 3. Nutrient load in Yellow Sea

Previous research proved that the nutrient load into the sea mainly including the load from river and the atmosphere. Here we can descript the detail nutrient load from the two aspects.

3.1 Nutrient loading from river discharge

Based on monthly monitoring data of unfiltered water, the nutrient discharges of the eight main rivers flowing into the coastal waters of China were calculated. In 2012, the total load of NH_3-N (calculated in nitrogen), total nitrogen (TN, calculated in nitrogen) and total phosphorus (TP, calculated in phosphorus) was 5.1×10^5, 3.1×10^6 and 2.8×10^5 tons, respectively (Tong *et al.*, 2015). The nutrient loading from the eight major rivers into the coastal waters peaked in summer and autumn, probably due to the large water discharge in the wet season. The Yangtze River was the largest riverine nutrient source for the coastal waters, contributing 48% of the NH_3-N discharges, 66% of the TN discharges and 84% of the TP discharges of the eight major rivers in 2012. The East China Sea received the majority of the nutrient discharges, i.e. 50% of NH_3-N (2.7×10^5 tons), 70% of TN (2.2×10^6 tons) and 87% of TP (2.5×10^5 tons) in 2012 (Tong *et al.*, 2015). The riverine discharge of TN into the Yellow Sea and Bohai Sea was lower than that from the direct atmospheric deposition, while for the East China Sea, the riverine TN input was larger.

“Dead zones” in coastal areas have spread exponentially since the 1960s and have caused serious consequences for ecosystem functioning. One important cause for the dead zones is coastal eutrophication. Due to population growth, rapid industrialization and urbanization, numerous previously pristine, unimpacted coastal waters have undergone a transformation to more mesotrophic and eutrophic conditions. At present, eutrophication offshore has become a global concern, and also one of the most prominent environmental problems in China. In 2012, the area affected by eutrophication was estimated to be 9.8×10^4 km^2 in China, increasing by 2.4×10^4 km^2 from 2011. The areas with the most severe eutrophication always occurred near the estuaries of the main rivers. Coastal eutrophication could be attributed to the enrichment of nutrients in the water, such as nitrogen and phosphorus, and the nutrient enrichment could increase the productivity of phytoplankton, ultimately leading to harmful algal blooms

(HABs). It has been reported that red tides occurred 73 times offshore from China, and the coastal area affected by red tides could be as high as 7971 km^2 in 2012.

The sources of nutrients flowing into the seas could be generally divided into non-point sources (such as agriculture diffuses) and point sources (such as industrial and sewage sources). Nowadays, more grains had be planted in China, the grain production in 2005 was 70% higher than in the 1980s. However, it has been estimated only 20–35% of the nitrogenous fertilizer used could be assimilated by the crops, while the majority was discharged into the environment. Moreover, as a result of improving living standards in China, many people have shifted their dietary preferences toward animal-derived products. These dietary changes were also associated with increased nutrient inputs for agriculture. Wastewater and other industrial emissions caused by the urbanization and industrialization also increased nutrient inputs to the rivers and coastal waters. In 2012, the wastewater produced in China was 684.6×10^8 tons. Compared with the rapid growth in wastewater amounts, wastewater treatment facilities could not keep up with the urbanization progress. From 2000 to 2005, only 30–45% of the emitted wastewater was treated before being discharged into waters. After 2006, in order to control the pollution of point sources, the Chinese government began to emphasize the construction of wastewater treatment plants. In 2010, the percentages of treated sewage in urban and rural in China were up to 82% and 60%, respectively.

The nutrients emitted from both point and non-point sources are ultimately discharged to the coastal water through the rivers. Currently, most statistical studies for Chinese cases only focus on changes in nutrients concentrations or transport in small-scale watersheds. For instance, Chen *et al.* (2000) found seasonal variances of nitrogen content varied with watersheds in the Yangtze River system, and the difference of nitrogen contamination level was related to the regional population and economic development (Chen *et al.*, 2000). Li *et al.* (2014) concluded that increased nutrient loads from the Yangtze River had led to increased Harmful Algal Blooms (Li *et al.*, 2014). However, although the modeling research has been carried out recently, the knowledge about the riverine nutrient discharges into the seas in

China is still limited. Here we can give some details about the nutrient discharge in China.

Concentrations of NH_3-N, TN and TP

The yearly NH_3-N concentrations in the four large rivers discharge into Yellow Sea are presented in Figure 4.1. In 2012, the highest NH_3-N concentration was found in the Haihe River, with an average of 3.7 mg/L, and the lowest was found in the Huanghe River, with an average of 0.5 mg/L. Generally, the NH_3-N concentrations have decreased since 2006. For example, the average NH_3-N concentration for the Haihe River in 2006 was 14.0 mg/L (the monthly values ranging from 7.0 to 24.3 mg/L), decreasing to 3.7 mg/L (ranging from 0.3 to 8.8 mg/L) in 2012. For the Liaohe River, the average NH_3-N concentration was 3.7 mg/L (ranging from 0.4 to 9.0 mg/L) in 2006, decreasing to 1.0 mg/L (ranging from 0.2 to 2.4 mg/L) in 2012. For the Yangtze River, the average concentrations were 0.7 mg/L (ranging from 0.2 to 2.7 mg/L) in 2006 and decreased to 0.3 mg/L in 2012. In 2012, the average monthly NH_3-N concentrations of the four rivers in northern China (Liaohe, Huanghe, Haihe and Huaihe Rivers) were significantly higher than those of the rivers in the south (such asYangtze, Qiantangjiang, Minjiang and Zhujiang Rivers) (Tong *et al.*, 2015).

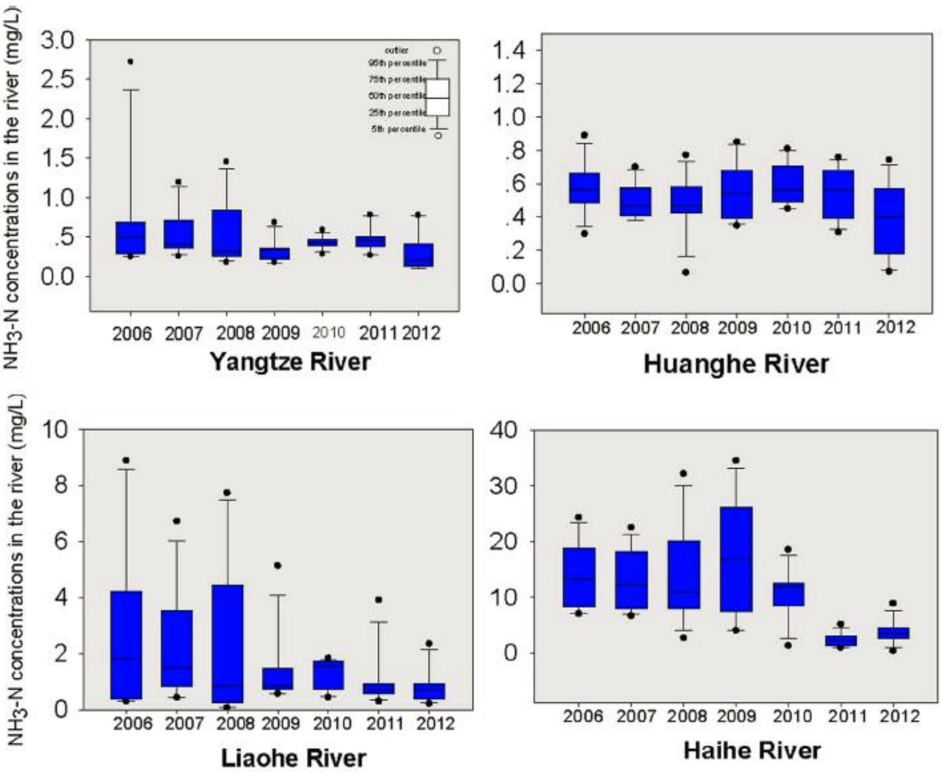

Figure 3.1. Yearly NH_3-N concentrations in the four large rivers discharge into Yellow Sea.

The yearly TN concentrations in the seven large rivers (excluding the Liaohe River) are provided in Figure 4.2. During the period between 2006 and 2012, the TN concentrations were generally lower than 5 mg/L for a majority of the rivers. However, the TN concentrations showed a decreasing trend from 2006 to 2012. For the Huaihe River, a remarkable decreasing trend was observed from 2007 to 2008. The average TN concentration in the Huaihe River in 2007 was 2.7 mg/L (ranging from 1.2 to 4.5 mg/L), and decreased to 1.0 mg/L (ranging from 0.9 to 1.8 mg/L) in 2008. For the Huanghe River, TN concentrations were quite stable from 2006 to 2012, staying near 1.0 mg/L.

The yearly TP concentrations in the eight large rivers are provided in Figure 4.3. In 2012, the highest TP concentration was observed in the Haihe River, with an average concentration of 0.6 mg/L (ranging from 0.3 to 0.7 mg/L), and the lowest TP concentration occurred in the Huanghe River, with an average concentration of 0.04 mg/L (ranging from 0.04 to 0.06 mg/L). For the Huaihe River, the average TP concentration decreased from 0.11 mg/L (ranging from 0.07 to 0.20 mg/L) in 2006 to 0.07 mg/L (ranging from 0.03 to 0.10 mg/L) in 2012.

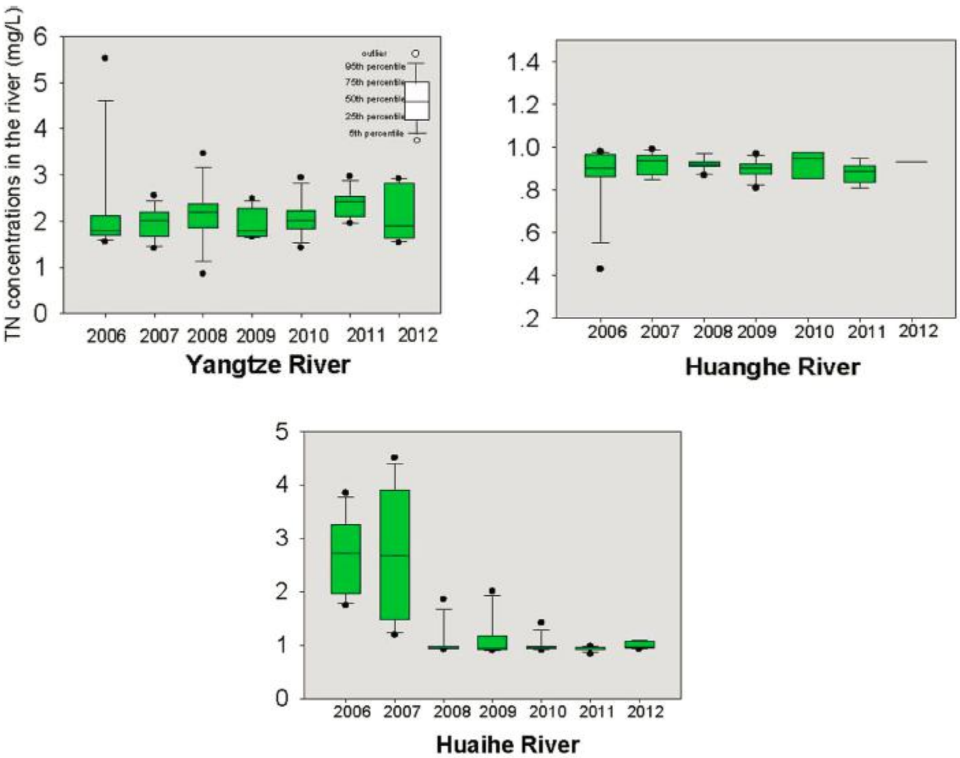

Figure 3.2. Yearly TN concentrations in the Yangtze, Huanghe, and Huaihe rivers.

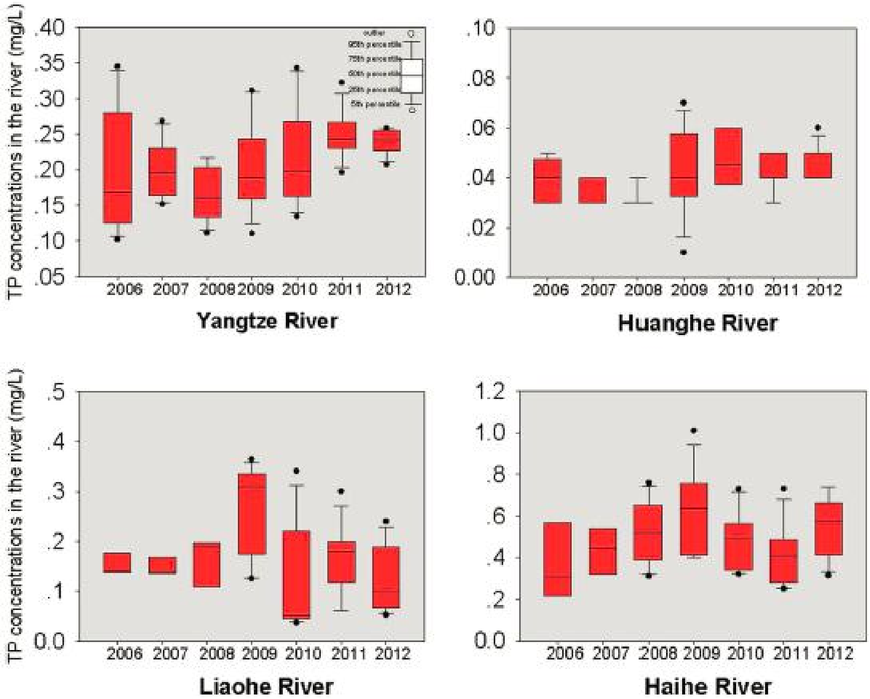

Figure 3.3. Yearly TP concentrations in the Yangtze, Huanghe, and Huaihe rivers.

Nutrient loads

The yearly loads of NH_3-N, TN and TP of the four rivers flowing into the coastal waters of China from 2006 to 2012 were calculated and are presented in Table 4.1. In 2012, the total load of NH_3-N, TN and TP of the selected rivers was 5.1×10^5, 3.1×10^6 and 2.8×10^5 tons, respectively. In 2006, the nutrient load was 7.4×10^5, 2.2×10^6 and 1.6×10^5 tons for NH_3-N, TN and TP, respectively. The Yangtze River was the largest riverine nutrient source for the coastal waters. In 2012, Yangtze River contributed 48% of the NH_3-N discharges, 66% of the TN discharges and 84% of the TP discharges of the four major rivers. The Huanghe River is the second longest river in China, but the nutrient discharge to coastal waters was much lower than that of the Yangtze Rivers. The small nutrient discharge could possibly be attributed the low water discharge of the Huanghe River, which was caused by excessive water use and climate change in the Huanghe River Basin.

The nutrient loading varied a lot among different seas. The East China Sea received the majority of the nutrient discharges. In 2012, about 50% of the NH_3-N (2.7×10^5 tons), 70% of the TN (2.2×10^6 tons), 87% of the TP (2.5×10^5 tons) loads were discharged into the East China Sea, while only 3% of the NH_3-N loads and less than 1% of TN and TP loads flowed into the Bohai Sea (Table 2). The South China Sea is also an important destination for nutrients. In 2012, about 2.2×10^5 tons of NH_3-N, 8.0×10^5 tons of TN and 3.5×10^5 tons of TP were discharged into the South China Sea.

Table 3.1 Yearly nutrient loads of NH_3-N, TN and TP into coastal seas during 2006-2012 (the unit was in tons).

	Yangtze	Huanghe	Liaohe	Haihe	Huaihe	Qiantangjiang	Minjiang	Zhujiang	Total
NH_3-N									
2006	4.37E+05	1.19E+04	4.48E+03	2.05E+00	2.50E+04	—*	1.72E+04	2.44E+05	7.40E+05
2007	3.66E+05	9.36E+03	2.25E+03	2.66E+00	3.51E+04	4.89E+03	1.05E+04	2.13E+05	6.42E+05
2008	3.48E+05	6.71E+03	3.96E+03	3.77E+00	1.48E+04	1.02E+04	1.36E+04	2.89E+05	6.86E+05
2009	2.56E+05	7.18E+03	1.71E+03	6.79E+00	8.45E+03	4.87E+03	1.17E+04	2.00E+05	4.89E+05
2010	4.28E+05	1.07E+04	9.75E+03	2.86E+00	1.88E+04	1.03E+04	2.37E+04	2.88E+05	7.89E+05
2011	2.98E+05	1.02E+04	2.52E+03	1.16E+00	6.17E+03	3.96E+03	7.79E+03	1.39E+05	4.68E+05
2012	2.44E+05	1.25E+04	2.03E+03	2.23E+00	7.28E+03	9.63E+03	1.64E+04	2.19E+05	5.11E+05
TN									
2006	1.35E+06	1.61E+04	—	2.47E+03	7.14E+04	—	—	7.74E+05	2.22E+06
2007	1.53E+06	1.81E+04	—	3.06E+03	1.02E+05	2.33E+04	—	7.05E+05	2.38E+06
2008	1.72E+06	1.36E+04	—	5.80E+03	2.90E+04	4.18E+04	7.04E+04	1.02E+06	2.90E+06
2009	1.49E+06	1.21E+04	—	7.88E+03	1.41E+04	3.17E+04	5.83E+04	7.52E+05	2.37E+06
2010	2.09E+06	1.37E+04	—	2.67E+03	3.12E+04	6.59E+04	1.34E+05	9.52E+05	3.29E+06
2011	1.61E+06	2.07E+04	—	—	9.22E+03	4.58E+04	4.94E+04	4.96E+05	2.24E+06
2012	2.04E+06	2.15E+04	—	4.72E+03	1.08E+04	7.52E+04	1.23E+05	8.02E+05	3.08E+06
TP									
2006	1.22E+05	7.91E+02	2.29E+02	4.73E+01	3.22E+03	—	—	3.34E+04	1.60E+05
2007	1.56E+05	6.64E+02	1.54E+02	1.06E+02	5.78E+03	1.13E+03	2.08E+03	2.52E+04	1.91E+05
2008	1.43E+05	4.59E+02	5.32E+02	1.98E+02	2.67E+03	1.34E+03	2.06E+03	4.27E+04	1.93E+05
2009	1.48E+05	6.08E+02	3.51E+02	2.56E+02	1.62E+03	9.51E+02	2.25E+03	2.00E+04	1.74E+05
2010	2.02E+05	8.69E+02	4.72E+02	1.45E+02	3.34E+03	2.75E+03	5.61E+03	3.67E+04	2.52E+05
2011	1.67E+05	1.01E+03	4.00E+02	2.09E+02	8.29E+02	1.30E+03	1.71E+03	2.17E+04	1.94E+05
2012	2.38E+05	1.26E+03	2.78E+02	4.24E+02	9.07E+02	2.60E+03	5.34E+03	3.47E+04	2.83E+05

3.2 Nutrient loading from the atmosphere

Nitrogen limitation characterizes large segments of the world's oceanic, coastal, and estuarine water. The rate of biologically available nitrogen supply to these waters is a key control of primary production and resultant trophic state. Anthropogenically generated inputs of new N contribute from 25% to >50% of new coastal primary production and are therefore believed to play a key role in the geographically skewed distribution of total oceanic primary production. Among the most rapidly growing sources of anthropogenic new N loading is atmospheric deposition. It was estimated that about 20-40% of new N inputs into coastal waters are of atmospheric origin, virtually all of them attributable to growing agricultural, urban, and industrial emissions. Atmosphere deposition alone contributes from 300-1000 mg N m^{-2} yr^{-1} in coastal water. The relative contribution of atmosphere deposition nitrogen to coastal N budgets could increase substantially during the next decades. Globally, atmosphere deposition N is a highly significant contributor to oceanic new N inputs, accounting for ~35 Tg N yr^{-1}, compared to 30 Tg N yr^{-1} from runoff and riverine discharge. In numerous regions, atmosphere deposition nitrogen is the single most important source of new N currently impacting the coastal zone.

Moreover, the research quantify outputs in riverine export, crop uptake, denitrification, volatilization, runoff, sedimentation and sea water exchange and implied that all of the nitrogen budgets were positive, with N inputs exceeding outputs. The excess N inputs gave rise to increases in N storage in landfills and in groundwater. Annual accumulation of N in the Yellow sea, including inputs from South Korea and other drainage areas, was 1229 kt yr^{-1} with a residence time for N of approximately 1.5 years, thus doubling N content in marine waters every 3 years during 1994–1997. The human derived N inputs leads to excessive eutrophication and pollution of the Yellow Sea.

Inputs from the application of mineral fertilizers averaged 226 kg ha^{-1} yr^{-1} in South Korean agriculture, which was the maximum input source term in the N budget for the region. Nitrogen inputs from biological fixation (nonsymbiotic only, since the area under symbiotically fixed crops was very small in ROK) was carried out using data from the Environmental Statistics Yearbook 1998; Cleveland et al. 1999; Zhu et al. 1997, as follows. In agricultural lands, the rate of fixation in the land area in rice plantations (1009560 ha) was taken to be 45 kg ha^{-1} yr^{-1}, yielding an annual flux of 45430 tons. Fixation rates in other cropland areas (966280 ha) were assumed to be 15 kg ha^{-1} yr^{-1}, yielding an annual flux of 14494 tons. In forested lands (5072600 ha) the assumed fixation rate (l kg ha^{-1} yr^{-1}) yielded an annual flux of 5072 tons. Total N inputs to agricultural and forest lands from fixation, therefore, equaled 64996 tons yr^{-1}.

N losses due to denitrification were calculated using data from Environmental Statistics data.

Denitrification loss in the agricultural areas of rice plantation (1009560 ha) was calculated as 32% of the fertilizer use rate, yielding an annual flux of 73011 tons. Denitrification loss in upland crop areas (966280 ha) was calculated as 15% of the fertilizer use rate, yielding an annual flux of 31887 tons. Denitrification loss from manure was calculated as 13% of the manure N application rate, yielding an annual flux of 20528 tons. Denitrification losses from soils were assumed to be 3 kg ha^{-1} yr^{-1}, yielding an annual flux of 5916 tons. Agricultural recycled N was considered for regional biogeochemical budget in South Korea agroecosystems as organic fertilizer N. The values of organic fertilizer N were assessed using the statistical data on human and animal/poultry population and rates of N in excreta (Table 2). Losses from anthropogenic NH_3 emissions were estimated in an earlier study (Park 1998). The modified European calculation factors were applied. The average total value was 142123 ton and NH_3 emission from fertilizers was predominant (35% from the total value).

In addition to the input/output items for agroecosystems, the N fluxes with river runoff for calculating the N budget for the whole South Korean area. The mean annual water discharge was 61.6×10^{12} L. In accordance with statistical data, about half of wastewater was untreated in ROK in 1994–1997. As a consequence, the content of reduced N in surface waters was almost the same as the content of oxidized N. Nitrite-N was also monitored in South Korea rivers and its mean content was 0.045 mg/L.

Chapter 4. Wastewater treatment and nutrient removal in Yellow Sea wetland

Wetlands are in use worldwide to reduce concentrations of nutrients in through-flowing water. Many studies at the site scale have demonstrated that wetlands have a high and long-term capacity to improve water quality and this evidence has resulted in many initiatives to restore or even create wetlands for this particular purpose. The most 'human-controlled' examples are the so-called 'treatment wetlands', which are constructed, planted and hydraulically controlled for the purpose of removing pollutants from wastewater. Apart from these constructed wetlands, (semi)natural wetlands in landscapes, such as riparian (stream-side) wetlands, also reduce the nutrient load of through-flowing water by removing nitrate and phosphorus from surface and subsurface runoff. With water quality in streams severely deteriorating in densely populated areas and intensively farmed regions worldwide, the interest from natural resource managers in the purifying functionality of wetlands in river catchments is increasing. In Europe, future water-quality standards will become stricter as a result of the implementation of the EU Water Framework Directive. In Canada and the USA, there is also pressure to tighten water-quality standards. Manuals to restore riparian wetlands have been produced and there have been studies to determine the most optimal location or spatial arrangement of wetlands in agricultural catchments. The use of riparian zones and other types of wetlands for water-quality enhancement has been advocated as a mitigation procedure for ever-more intensive land-use practices, involving denser livestock rearing and increased fertilizer application.

Water quality in many stream catchments and river basins is severely impacted by nutrient enrichment as a result of agriculture. Water-resource managers worldwide are considering the potential role of riparian zones and floodplain wetlands in improving stream-water quality, as there is evidence at the site scale that such wetlands are efficient at removing nutrients from through-flowing water. The water purification function of wetlands at the site and catchment scale and suggest ways in which these disadvantages could be overcome. Nutrient loading of terrestrial, freshwater and coastal ecosystems occurs as a result of human waste disposal and agriculture at a global scale. Although technological purification plants are the best option to reduce the nutrient fluxes to the environment, nutrient loading owing to intensive agricultural practices typically occurs through diffuse or 'non-point' sources, which are hard to tackle technologically. Measures to reduce diffuse sources of nutrient loading are: (i) the reduction of fertilizer application; (ii) the use of nitrogen-fixing crops; and (iii) the restoration or creation (hydrological connections to) of wetlands in the landscape.

In reality, we should critically evaluate the practice of using wetlands to manage water quality in catchments dominated by agriculture. Firstly, we should first consider this practice from a functional perspective and evaluate under what conditions of loading intensity and relative wetland surface area in the catchment wetlands are effective. We pay special attention to undesired effects such as enhanced greenhouse gas production and breakthrough of nutrients stored earlier in the system as a result of overloading. And then, we evaluate the ecological consequences of nutrient loading on the species composition and structure of wetlands. This issue has been ignored in most catchment water-quality enhancement initiatives, where wetlands are considered simply as systems with a high potential for nutrient retention.

4.1 Wastewater and manure dumping

In many cases, the conservation or restoration of habitat for plant and animal species is seen as an additional benefit of such initiatives. And the main question is using wetlands to improve water quality: how does it work? River catchments in which the dominant land-use type is agricultural often have lower-order stream subcatchments that are strongly influenced by runoff from fields or grasslands. In intensively farmed areas, nutrient loading is often so high that large quantities of nitrate leach into the groundwater, which discharges into streams as seepage or subsurface runoff. In intensively farmed catchments, phosphorus (P) and nitrogen (N) are also transported to streams in surface runoff. Between agricultural fields and streams, one often finds riparian areas that can influence surface and subsurface runoff before it reaches streams.

There is a large body of literature based on studies at individual sites that indicates that riparian habitats remove nutrients from the water flowing through them on its way from the agricultural land to the stream. The most frequently documented function is the removal of nitrate from subsurface run-off in wetland zones with anaerobic soil conditions. Denitrification is generally the most important process for nitrate removal, whereby dead organic matter is decomposed by bacteria in the absence of oxygen, using nitrate as an electron acceptor. Nitrate is converted to nitrous oxide (N_2O) and, subsequently, to atmospheric nitrogen (N_2), which is emitted by the wetland. Nutrient uptake in vegetation as water passes through the riparian zone is also important and results in long-term nitrogen storage. However, its removal from the system only occurs if the vegetation is harvested as part of the management of that system. Phosphorous removal in riparian habitats has also been reported, with sedimentation, soil adsorption and plant uptake being the most important mechanisms.

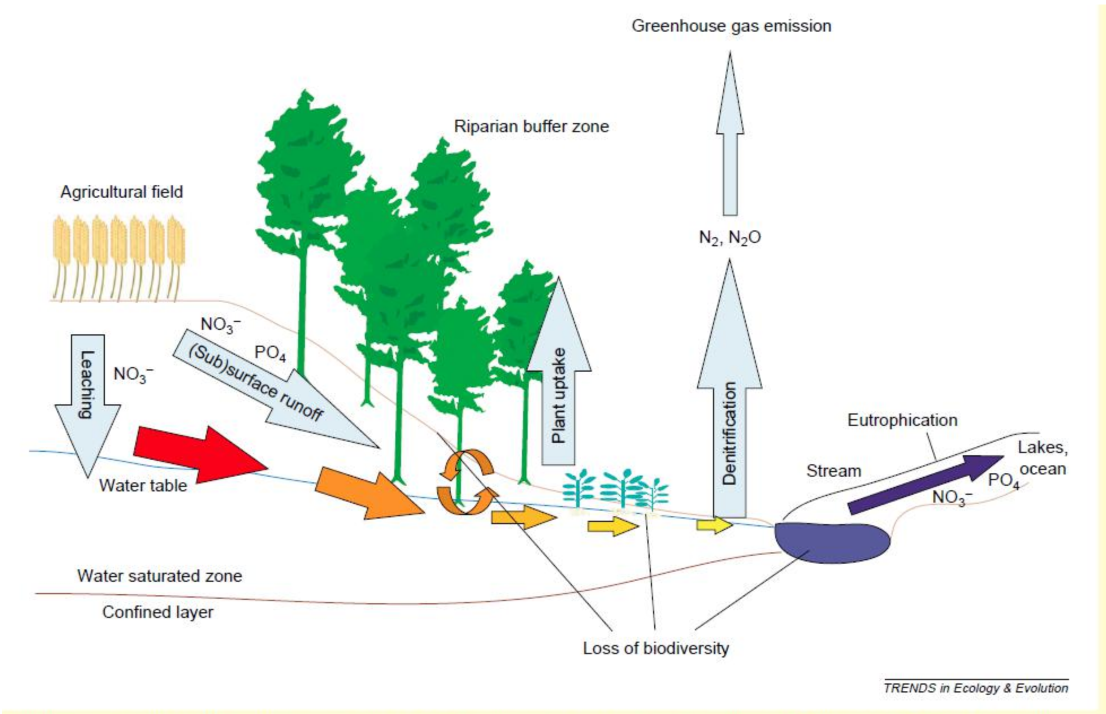

Figure 4.1. Cross-section of riparian wetland showing hydrological fluxes, nutrient processes and environmental impacts of nutrient loading.

Research on the nutrient removal capacity of wetlands in the temperate zone has revealed that the maximum potential rate of nitrogen and phosphorous removal typically ranges from 1000 to 3000 kg N ha^{-1} yr^{-1} and from 60 to 100 kg P ha^{-1} yr^{-1}. These are high values, if one considers that they are an order of magnitude higher than fertilizer applications in intensively farmed areas. The capacity of riparian wetlands to remove nitrogen and phosphorous is, however, only important if it can be demonstrated that there has been a significant reduction of the load of nitrogen and phosphorous that reaches stream ecosystems. In some cases, it has been argued that riparian wetlands contribute in a 'significant' way if they remove at least 30% of the total nitrogen and phosphorous load. Another approach is to investigate whether nitrogen and phosphorous removal by riparian wetlands contributes to meeting the water-quality standards in the receiving surface water. These standards are, however, often pragmatic compromises and subject to change. For example, water-quality standards differ regionally in the USA, and have requirements in terms of the presence of indicator species rather than strict limits for nitrogen and phosphorous concentrations in Europe ('ecological quality', EU Water Framework Directive). For practical reasons, therefore, we use the first approach by adopting a standard of 30% removal of nitrogen and phosphorous loading as the boundary between 'significant' and 'insignificant' reductions.

Studies evaluating the removal of nitrogen and phosphorous by wetlands at the catchment level have been carried out at different scales. For the whole of the Mississippi basin, Mitsch et al. calculated that 20–50% of the total nitrogen load carried by the river into the Gulf of Mexico could be removed by restoring a major proportion of all riparian zones and wetlands associated with the small, lower-order streams, together covering 1–2% of the total catchment area. An additional 20–50% of this total nitrogen load to the Gulf of Mexico could be removed if bottomland hardwood forests associated with the river floodplains were restored to the point that they would cover 3–7% of the total Mississippi basin. Similar estimates were made in catchments in southern Sweden, which are a major source of nitrate enrichment of the Baltic Sea. In the Ronnea catchment, restoration of 148 wetlands covering 0.6% of the catchment area failed to improve river nitrate concentrations. A modeling study revealed that 40% nitrogen removal would require a wetland area covering 5% of the total catchment. The traditional Chinese 'multipond systems' indicate that effective nitrogen and phosphorous retention occurs when the water in a catchment is directed through a system of ditches and created wetlands covering around 5% of the catchment area, preventing high nutrient loading of the streams and rivers and associated eutrophication problems. These examples from the USA, Sweden and China all suggest a global rule that wetlands can contribute significantly to water-quality improvement at the catchment level if they account for at least 2–7% of the catchment area. Given that many wetlands in agricultural catchments are continually enriched with nutrients, it is relevant to consider the consequences for the species composition and functioning of the wetland ecosystem. Enriched wetland systems can lose species and can show drastic changes in nutrient cycling rates.

4.2 Nutrient removal

Nutrient inputs to wetland ecosystems have increased over the past century in many parts of the world. The resulting nutrient enrichment often has significant effects, including increased productivity, higher rates of nutrient leaching and shifts in the dominance, and composition, of species. Most ecosystems can incorporate higher loading rates, which have only minor effects as long as they do not surpass a certain limit. However, when nutrient loading rates surpass this critical level, species composition and ecosystem functioning change dramatically over short periods of time and the systems often move to a different stable state. Among the best-known examples are shallow lakes that shift from water with low turbidity and abundant submerged macrophyte vegetation to a turbid state with prolonged phytoplankton dominance. Wetlands that are characterized by low productivity and high plant diversity dominated by slowgrowing, nutrient-conserving species shift to systems dominated by large, fast-growing helophytes following a strong increase in nutrient-loading rates. The degree to which the species composition changes depends on the natural nutrient richness of the system. Naturally nutrient-poor (i.e. oligotrophic and mesotrophic) systems react more drastically than do naturally nutrient-rich (eutrophic) systems. Nutrient-poor systems show a complete shift in plant species composition as well as a drastic change in nutrient dynamics, whereas nutrient-rich systems might show only further increased productivity. All systems, however, show a characteristic breakdown of the nutrient retention function after prolonged high nutrient loading. Species-rich communities dominated by sawgrass Cladium jamaicense have been replaced by tall species-poor cattail Typha domingensis stands in areas enriched with nutrients. In the same areas, the rate of phosphorus cycling has increased as a result of higher decomposition rates. This reinforces the nutrient richness of the system and creates a situation that is difficult to reverse to mesotrophic conditions.

Such shifts in structure (i.e. species composition) and functioning (nutrient cycling and retention) have spurred scientists to propose critical loads of nitrogen and phosphorous for ecosystems. A critical load of a nutrient is defined as the loading rate below which the system remains all but unchanged, but beyond which it exhibits sudden, drastic changes, including a shift in species dominance and species composition and a major change in ecosystem functioning, in terms of carbon (C) and nutrient outputs, trophic interactions and/or nutrient cycling rates. Such situations of nutrient overloads cannot only be detected as a major shift in species composition or structure, but also as a distinct increase in the nutrient concentrations of water that is exported from the ecosystem (e.g. in wetland outflow). A drastic increase in output nutrient concentrations is another indicator that can be used to establish critical loads for ecosystems. For wetlands, a critical loading rate of 10 kg P ha^{-1} yr^{-1} has been proposed, based on the analysis of a large database of wetlands enriched with nutrients. For nitrogen, much research has been carried out along gradients of atmospheric nitrogen deposition in Europe. Increased levels of atmospheric nitrogen deposition occur as a result of air pollution owing to fossil fuel combustion (NOx) and agriculture (NH_3). In Western Europe during the 1980s, levels were higher than 45 kg N ha^{-1} yr^{-1}, which is ten times the background value. Metadata sets on the effects of increased deposition have proposed a critical nitrogen load of 25 kg N ha^{-1} yr^{-1} for wetlands, but there is little published information available about differences among different wetland types in this respect.

It is striking that the proposed critical loading rates of nitrogen and phosphorous for wetlands are several orders of magnitude lower than the typical loading rates in natural or constructed wetlands used for improving water quality. Only in a few cases, loading rates of nitrogen and phosphorous have been close to these critical values. This implies that critical nutrient loads are easily surpassed in many natural wetlands and that, depending on their original trophic status, shifts in species composition and/or increases in nutrient concentrations in the outflow will occur, in spite of rates of nutrient retention remaining high. Many of them lost their original plant diversity decades ago, but continue to retain high quantities of nitrate, even though the surface and subsurface water discharged from them still have high concentrations of nitrate. Hence, wetlands in agricultural catchments can contribute significantly to water-quality improvement, but their loading rates often surpass critical values.

4.3 Enhanced nitrous oxide emissions

The major process responsible for nitrate removal in wetlands is denitrification. However, in situations where the reduction of the nitrate to N_2 is incomplete, the denitrification process can also be a major source of the greenhouse gas N_2O, which has a global warming potential 310 times that of CO_2. N_2O accounts for 6% of the total greenhouse effect and also has an important role in the destruction of the stratospheric ozone. N_2O emission in wetlands is generally promoted under conditions that are suboptimal for denitrification, such as low pH or soil moisture. However, N_2O production is also promoted by high nitrate availability, because it is energetically favorable for denitrifyers to reduce nitrate instead of N_2O. There is much recent information about the increase in N_2O emission after nitrogen addition to agricultural soils. Much less data exist on the indirect effect of nitrogen addition by agriculture on N_2O emissions from riparian wetlands.

Recent publications strongly suggest that nitrogen transformations in buffer zones receiving high levels of nitrate result in a significant increase in N_2O emissions. For example, riparian buffer zones in Twente, the Netherlands, showed high N_2O emissions (20 kg N ha^{-1} yr^{-1}) at sites with high nitrate loads. This implies that the nutrient retention benefit of riparian zones comes at an environmental cost, in particular when nitrate-loading rates are also high. This potential negative consequence of loading wetlands with nitrate is often ignored or downplayed. Thus, there is a great need for additional information about the risk of N_2O emissions from nitrogen-loaded wetlands and about management options to balance that risk against the environmental benefit of water-quality improvement.

Table 4.1 Characteristic nitrogen and phosphorous loading rates of wetlands in agricultural catchments in relation to relevant loading thresholds.

Catchment	Location	Wetland type	Origin	N load (kg ha^{-1} y^{-1})	P load (kg ha^{-1} y^{-1})	Refs
Liuchahe	P.R. China	Multipond	Constructed	>500	>50	[35–37]
Regge, Twente	Netherlands	Riparian	Natural	200–1140		[13,63]
Everglades	USA	Marsh	Natural	N/A[a]	2–40	[38,43]
Mississippi	USA	Forested	Natural	19–39	2–9	[49]
Various	USA	Riparian	Natural	20–155	N/A	[32,49]
Treatment wetlands in USA and Europe			Constructed	500–9000	100–2000	[2,32]
Maximum load[b]				1000	60	[2,20,32]
Critical load[c]				25	10	[44,45,47,48]

[a]No data available.
[b]Beyond this limit, the wetland will show substantial leaching and associated high concentrations in the outflow.
[c]Beyond this limit, wetlands with a species-rich vegetation will show a dramatic increase of productivity and associated change in species composition.

Wetlands in nutrient-loaded agricultural catchments have a significant role in improving water quality. Our analysis shows that this potential is only realized in catchments with a minimum area of wetlands relative to total catchment size. Measurements from different regions around the world indicate that at least 2–7% of the total catchment needs to be in wetland habitat to see a significant increase of water quality at the catchment scale, a remarkably narrow range. This minimum value has already raised much dispute among policy makers as to the practicality of restoring or creating such a large area of wetlands as a management tool, particularly in many European catchments, where the proportion of wetlands has often become close to zero.

The implications for land-use management in areas under intensive agricultural use are that only a combination of measures will result in acceptable environmental quality, that is, (i) fertilizer levels have to be reduced significantly; (ii) riparian zones and floodplain wetlands should be rehabilitated; and (iii) their location in the landscape should be carefully selected on the basis of hydrological studies. This will only be possible at great economic cost, which will delay implementation. Another clear outcome of our discussion is that many wetlands in agricultural catchments are loaded beyond the ‘critical’ limits, and some even beyond the ‘maximum’ limits. This means that many riparian zones will lose plant species or have already done so. Riparian wetlands in such catchments will all converge towards a narrow set of ecosystem types characterized by high levels of nutrient richness, a high primary productivity and low species density. The example of wetlands used for water treatment in the Mississippi Delta shows that the nutrient retention function can be combined successfully with conservation of the original status of the ecosystem and its biodiversity under the condition that loading rates are low and not surpassing critical limits.

Land-use planners and environmental resource managers are facing many challenges to ensure good surface-water quality in agricultural catchments. In their development of sustainable water-quality management options, they must give consideration to the existence of loading limits as discussed here. Because of their high potential for nutrient retention, it is still a good idea to use wetlands in catchment water resources management for water-quality improvement. Restoring wetlands and their hydrological connections to the upland and the stream can be a rewarding activity in this respect. Most importantly, however, the combination of water quality improvement with wetland biodiversity conservation requires loading rates of wetlands to remain below critical thresholds. In many agricultural catchments, these limits have been surpassed and biodiversity restoration would require a decrease in the loading through decreased fertilizer applications or the use of larger areas of wetlands for nutrient retention. In catchments with intensive land use, loading rates could surpass another, much higher critical limit, beyond which the wetland ecosystem no longer performs its retention function properly but releases nutrients or emits the greenhouse gas N_2O. In such catchments, the only feasible measure is to decrease fertilizer levels.

Chapter 5. Mechanisms of using wetland as nutrient sinks

Wetlands are being considered increasingly important for wastewater treatment because of the ability of many wetland plants to absorb large amounts of nutrient and a variety of toxic substances. The studies highlight the physical, chemical and biological processes which contribute to the improvement of water quality, and the distinction between natural and constructed wetlands. The impacts of long-term wastewater disposal on the biotic changes, reduction in treatment efficiency, and wetland processes such as production of trace gases. Constraints in using wetlands, for wastewater treatment, such as poor understanding of the natural wetland functions and responses of native plants and animals to wastewater, particularly in developing countries, are briefly discussed. It is suggested that while the possibilities for using constructed wetlands based on native species for small communities are explored, greater emphasis should be laid on the restoration of lost and degraded wetlands, especially the river floodplains, lake littorals and coastal wetlands, which can help check pollution from non-point sources.

Wetlands include a large spectrum of habitats: from temporary shallow waterbodies such as play as or billabongs to marshes and swamps, from lake margins (littorals) to large river floodplains, from coastal beaches and salt marshes to mangroves, from peat bogs and fens to coral reefs and beds of marine algae or seagrasses. What brings all these diverse kinds of habitats together is that the land is so wet (or under water) for a part or whole of the year that the vegetation is quite distinct from that of the adjacent upland areas. Differences in the hydrological regime of various habitats, coupled with other climatic and edaphic factors, create the diversity of wetlands with different structural and functional characteristics (Mitsch & Gosselink, 2008). These values were recognized only recently by the developed world which considered them so far as wastelands. However, it is important to recognize that not all wetlands are similar in their function.

Nutrient transformation is one of the major wetland functions which is translated into their value for improving the quality of wastewater. Efforts are made to use wetlands for disposal of wastewaters from various sources and with different characteristics. Wetlands are now being constructed worldwide, designed especially for wastewater treatment at secondary and tertiary level (Haberl, 1997). An erroneous impression is often conveyed that natural wetlands can be used for disposal of domestic wastewater and industrial effluents, and that the constructed wetlands offer a cost-effective alternative to conventional wastewater treatment, particularly for the developing countries. Many studies show that the use of either natural or constructed wetlands is not without its limitations. It is necessary that the constraints of using wetlands for wastewater treatment and the problems likely to arise from such use are also highlighted and properly evaluated. Richardson and Nichols (1985) discussed the ecological considerations in the use of natural wetlands for wastewater treatment. Some limitations of constructed wetlands have also been pointed out earlier.

5.1 Mechanisms of nutrient retention for coastal wetland

In reality, coastal wetlands can be created to fulfil specific ecological services, such as providing wildlife habitat, retaining pollutants and treating wastewater (Guittonny-philippe *et al.*, 2013), in our report, we focused more on the mechanisms of nutrient retention for coastal wetlands. Wetlands, especially for the natural coastal wetlands, are complex ecological systems that incorporate physical, biological and chemical processes. They play an important role in protecting freshwater and marine ecosystems from excessive inputs of nutrients, pathogens, silt, oxygen demand, metals, organics and suspended solids, as well as providing a buffer against storms, soil stabilization and wildlife habitat (Sierszen *et al.*, 2012).

Generally speaking, water purification by wetland occurs via two processes: nutrient uptake and suspended particle deposition. Previous researches about wetlands include surface flow wetlands or subsurface flow wetlands where the flow passes through a media bed in which plants are established. In surface flow wetlands, long detention times of typically 5–14 days and a large surface area promote removal of particulate and organic matter (Ghermandi *et al.*, 2007). Microbial processes, including oxidation of organic matter and transformation of nutrients, occur through the plant biomass, the sediment and the decomposing plant matter on the bed surface. In subsurface flow wetlands, the detention times are typically shorter (i.e. 1–2 days) and as illustrated in Fig. 5.1, functional microorganisms are associated with the surfaces of the substrate and with the root systems (i.e. rhizosphere) of plants established in the substrate . The porous substrate also acts as a filter for reducing levels of suspended solids. In both the surface flow and the subsurface flow wetlands, plants function to oxygenate the surface layers of the sediments and thereby provide an aerobic environment for microbial activity (Valipour & Ahn, 2016). In some cases, plants can accumulate and sequester nutrients (e.g. phosphorus) and pollutants (e.g. metals) from the surrounding substrate or from the water (Fig. 5.1). Recently, hybrid systems that include both vertical flow and subsurface flow systems have been shown to achieve superior pollutant removals (Vymazal, 2013). A variety of other constructed wetland designs have been proposed to enhance removals of pollutants (Wu *et al.*, 2015).

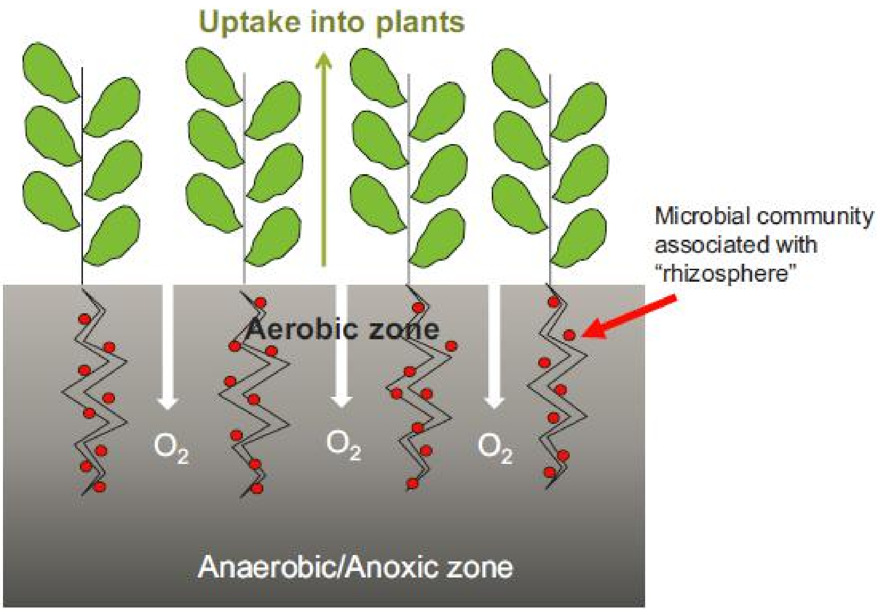

Figure 5.1 Illustration of the functions of plant and microbial communities in natural and constructed wetlands with subsurface flow.

Moreover, constructed wetlands also have been increasingly used for water reuse projects on a small scale, such as residential use for toilet flushing and gardening, or on a larger scale for irrigation of agricultural crops, golf courses and public parks, or to replenish natural wetlands and groundwater (Rousseau *et al.*, 2008). The benefits of constructed wetlands are both social and environmental and need to be considered in the design of the wetland. The social benefits of wetlands include education and recreation, and so wetlands are often designed with interpretive signage, walking and bike paths and green space (Birtles & Cunningham, 2013). These wetlands are often designed as part of an integrated urban design system with other features such as swales, grasslands and forest/shrub areas.

While the benefits of constructed wetlands are many-fold, there are still challenges to their implementation, including availability of land, community support, maintenance and monitoring. Given the relatively long hydraulic detention times and large surface areas required for constructed wetlands, there are often challenges in finding suitable land, especially in urban areas. Poor performance of constructed wetlands can occur where there is poor design or where the wetland is poorly maintained. Although maintenance and operation of constructed wetlands is less demanding than conventional wastewater treatment systems, they still require regular maintenance and monitoring. This includes ensuring even flow distribution, managing water levels, weed control, plant health, animal control (e.g. mosquitos, rodents, nutria) and removal of accumulated solids. It is valuable to have community support for these projects as the community is often relied upon to help with the maintenance of the wetland.

5.2 Natural wetlands for wastewater treatment

Wastewaters have been discharged into natural wetlands such as littoral marshes and floodplains for centuries without realizing their specific role. Since the high nutrient absorption potential of several macrophytes, especially water hyacinth was reported in 1960s, much attention has been paid worldwide to the study of uptake of various nutrients, trace elements and other toxic substances by various macrophytes under different laboratory and field conditions. Luxury consumption occurs in many plants which are able to concentrate nutrients from the sediments and/or water by a factor of several thousands. Previous research often use the concentrations of nutrients and other pollutants in plant tissues observed under experimental conditions over a few days or weeks to scaled up and interpret the field potential of these plants to remove pollutants from wastewaters throughout the year (e.g. Boyd, 1970). A mass balance approach has rarely been used to show the actual amounts removed by the plants when supplied with wastewater under field conditions. Under field conditions, which are suboptimal for growth, a part of the nutrients is returned readily to the system through death of plant parts. As the nutrient loading increases, the nutrient removal by uptake decreases sharply. Peterson and Teal (1996) have reported that the plants accounted for only 1% of the nitrogen removed from a marsh loaded with 5.23 g N/m^2/day in a pilot-scale study (Peterson & Teal, 1996). The total N loss from the system, however, was 38% of the total loading. At a higher loading (15.6 g N/m^2 /day), plants accounted for only 4% removal against the total loss of 68%.

All the nutrients removed by the plants from the wastewater (though some of them come from the sediments) cannot be removed from the system by harvesting the above-ground parts. The below-ground organs of emergents such as *Phragmies*, *Typha* and *Scirpus* species account for 35-50% of the total growth in first year of establishment (one year old stands) and up to 75% or more in older stands. The large proportion of nutrients held as reserves is internally recycled for fresh growth without requiring uptake. Repeated harvests at shorter intervals can help remove more nutrients from the system but deplete the below-ground organs of their nutrients and photosynthesis reserves, resulting in poor growth. The decomposition of below-ground organs in situ is generally slow resulting in a build up of organic matter in the soil. Consequently, the wetland system soon becomes saturated with nutrients, and the treatment efficiency declines sharply. Many field studies on natural wetlands have reported a reduction in their efficiency for N and P removal with time and increased levels of nutrient loading. There are also reports of reduction in the bacterial population of the effluents passing through the wetland. This is related to the reduction in the organic matter content, and not to the actual destruction of the bacterial load. Since Seidel (1964) demonstrated the reduction of pathogenic bacteria by *Scirpus lacustris*, there is no other direct evidence yet if the macrophytes can indeed kill the pathogens. On the other hand, it is also likely that the macrophytes promote the growth of pathogens by providing substrate and contribute to their perpetuation in the system.

The discharge of wastewaters into natural wetlands invariably alters the hydrological regime and brings in organic matter, nutrients and toxic substances. When the wastewater is discharged during the dry season, it alters the duration, timing and depth of flooding. At other times, the change in depth may be small or large depending upon the kind of wetland. Wetland plants are adapted to specific hydrologic regimes. Species which are adapted to a low water or dry period during their growth are adversely affected. Hydrological changes alter the species composition of plant communities, and also the growth and production of various species. The responses of Phragmites and Typha species, which are highly favoured for constructed wetlands, to hydrology and other environmental factors are too well known to be reiterated here. The impacts of water quality are often less understood despite numerous publications on the effects of pollutants on both plants and animals. Whereas the additional supply of nutrient may increase the growth, and hence net nutrient uptake by the plants, the increased loading of ions and toxic substances gradually shifts the community to one composed of more pollution tolerant species. Besides nutrients, other chemical changes such as in pH and salinity also cause a shift in the biotic community. Peat bogs receiving nutrient-rich surface waters develop into fens. Many studies confirm that eutrophication is one of the major/primary causes of reed decline in many European wetlands/lake littorals (Kubín & Melzer, 1997). Osborne and Totome (1994) reported the loss of submerged and floating-leaved species from Lake Waigani due to inflowing sewage effluents. Accumulation of toxic substances in benthic invertebrates and fish and through them into the waterfowl is well known. With respect to natural wetlands, some more points need serious consideration. First, throughout the world, there is great concern about the "eutrophication" and pollution of wetlands. Disposal of domestic and industrial wastes and increased nutrient supply into the wetlands (with runoff from catchments) are recognized as major factors responsible for wetland degradation worldwide. Is it not ironical that we should ever consider the possibility of using natural wetlands as alternate systems for wastewater treatment.

5.3 Constructed Wetlands for Wastewater Treatment

Since Seidel (1966) demonstrated the role of bulrushes *(Scirpus lacustris)* in wastewater treatment, all kinds of plants have been tried and wetland construction is fast becoming a business. It offers many advantages over the traditional oxidation ponds. Not only the aquatic plants contribute to the processes involved in the treatment, the large amount of organic matter produced by them can be put to economic use. One of the main considerations in promoting the use of constructed wetlands is their relatively low cost for construction, operation and maintenance, relative to the conventional treatment systems. Duckweeds have a very low biomass but very high growth rate, and therefore require frequent harvesting (every week or even twice a week). Their major role is through creating anaerobic conditions because they can remove nutrients from a thin layer (1-2 cm) of water only. Water hyacinth has also to be harvested frequently but has greater potential for nutrient removal and reducing suspended particulates. On the other hand, the cattails, reeds and sedges need to be harvested only once a year. All these plants can be used variously. Difficulties in harvesting and using water hyacinth have checked its place in constructed wetlands only to pilot plants. The majority of the constructed wetlands is now based on reeds or cattails (Vymazal, 1998).

However, the constructed wetlands also have also several constraints on their use and several factors need to be considered at length before recommending their use in the developing countries. First and foremost constraint is the large requirement of land per unit volume of wastewater to be treated. Estimates of the land requirement for treating per capita domestic waste based on the nutrient loading rates vary from 10 to 20 persons per hectare. Based on the hydraulic loading rate, the land requirement estimates are generally lower but the actual requirements will depend upon the nutrient concentrations in the wastewater and the retention period. Little attention has been paid to the seasonal variations in the concentration of the wastewaters, their organic and nutrient load and the seasonal climate wherein the monsoon rains can flood the wetlands and make their functioning impossible. Instead, they may become a source of pollution of surface- and ground-waters. It is often argued that the wetland systems are more suitable for the tropical/subtropical regions because of high temperature permitting growth throughout the year. If subfreezing winter temperatures are a constraint in the temperate regions, the high seasonal variability in precipitation with prolonged dry summer and heavy rains in the monsoon season is a constraint in the tropics/subtropics.

The source and quality of the wastewater are an important factors which should be seriously considered when recommending constructed wetlands in developing countries. Although experimental studies have been made on the use of constructed wetlands for treatment of industrial wastes, the majority of the studies refer to domestic or organic wastes. Wetlands are often recommended for tertiary (advanced) treatment because of the relatively high effluent standards adopted in the developed countries. If the raw municipal wastes are to be treated, a pre-treatment facility somewhat similar to a settling basin/oxidation pond is used before releasing the wastewater into the wetland. In developing countries, where even the sewage collection systems are not yet in place, and secondary treatment is not available even in large metropolises for all the sewage generated, tertiary treatment is a far-cry. The constructed wetlands have to be used for secondary treatment. Almost invariably, the municipal sewerage system also carries the industrial effluents, even if partly treated. This means that the municipal wastewater is also generally rich in inorganic and organic toxic pollutants which would affect the microbial processes as well as the wetland biota. In the absence of segregation of different kinds of wastewater during disposal, the constructed wetlands may not help provide the desired solution.

Wetlands have gained much importance in recent years. There is great concern at the rapid loss of natural wetlands due to drainage and reclamation for various purposes. International efforts are being made to conserve and protect them. The created (or constructed) wetlands are also seen as an effort in the direction of mitigating the wetland loss. The conservation and management of wetlands is linked to many global issues which need to be considered also in the context of their use for wastewater treatment. The impacts of wastewater on biodiversity in wetlands were mentioned above. Our knowledge of biodiversity in constructed wetlands is too meagre to enable comparisons but the differences are obvious in the absence of natural regulatory mechanisms. The problem of redwing blackbird (which could damage com fields) roosting on Typha stands in a constructed wetland in Ontario was pointed out by Wile *et al.* (1985). Weaver birds nested in large numbers on young *Typha angustata* stand in Keoladeo Ghana National Park in India though there were none in older stands.

5.4 The experiment using constructed wetland to remove nutrient

The first experiments using wetland macrophytes for wastewater treatment were carried out in

Germany in the early 1950s. Since then, the constructed wetlands have evolved into a reliable wastewater treatment technology for various types of wastewater. The classification of constructed wetlands is based on: the vegetation type (emergent, submerged, floating leaved, free-floating); hydrology (free water surface and subsurface flow); and subsurface flow wetlands can be further classified according to the flow direction (vertical or horizontal). In order to achieve better treatment performance, namely for nitrogen, various types of constructed wetlands could be combined into hybrid systems.

Constructed wetlands are engineered systems that have been designed and constructed to utilize the natural processes involving wetland vegetation, soils, and the associated microbial assemblages to assist in treating wastewaters. They are designed to take advantage of many of the same processes that occur in natural wetlands, but do so within a more controlled environment. Constructed wetlands for wastewater treatment may be classified according to the life form of the dominating macrophyte, into systems with free-floating, floating leaved, rooted emergent and submerged macrophytes. Further division could be made according to the wetland hydrology (free water surface and subsurface systems) and subsurface flow constructed wetlands could be classified according to the flow direction (horizontal and vertical). A simple scheme for various types of constructed wetlands is shown in Figure 5.2.

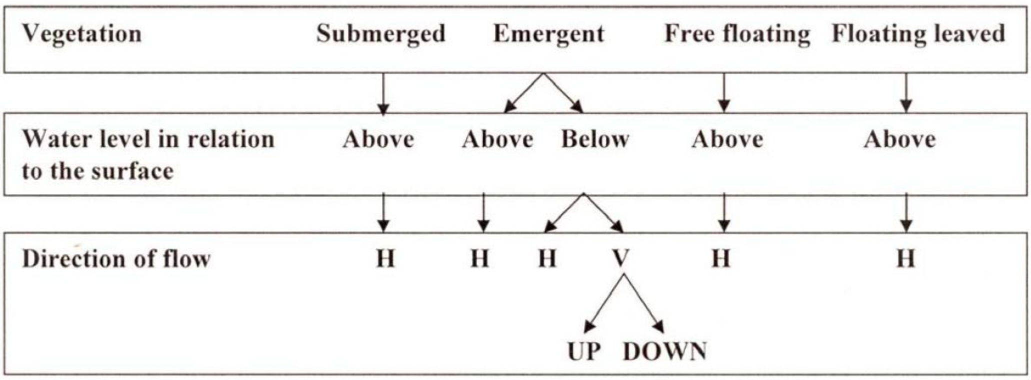

Figure 5.2 The major characteristics of various types of constructed wetlands for wastewater treatment.

The first experiments aimed at the possibility of wastewater treatment by wetland plants were undertaken by Käthe Seidel in Germany in the early 1950s at the Max Planck Institute in Plön. Seidel then carried out numerous experiments aimed at the use of wetland plants for treatment of various types of wastewater, including phenol wastewaters, dairy wastewaters or livestock wastewater. Most of her experiments were carried out in constructed wetlands with either horizontal (HF CWs) or vertical (VF CWs) subsurface flow, but the first fully constructed wetland was built with free water surface (FWS) in the Netherlands in 1967. However, FWS CWs did not spread substantially in Europe where subsurface flow constructed wetlands prevailed in the 1980s and 1990s. In North America, FWS CWs started with the ecological engineering of natural wetlands for wastewater treatment at the end of the 1960s and beginning of the 1970s. This treatment technology was adopted in North America not only for municipal wastewaters but all kinds of wastewaters. Subsurface flow technology spread more slowly in North America but, at present, thousands of CWs of this type are in operation. Various types of constructed wetlands may be combined in order to achieve higher treatment effect, especially for nitrogen. Hybrid systems comprise most frequently VF and HF systems arranged in a staged manner but, in general, all types of constructed wetlands could be combined in order to achieve more complex treatment efficiency.

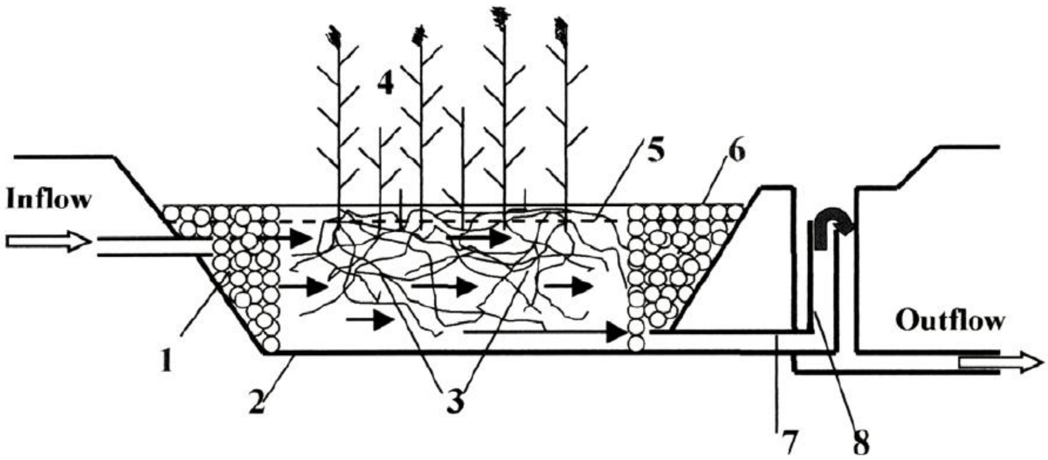

Figure 5.3 Schematic layout of a constructed wetland with horizontal subsurface flow. 1 inflow distribution zone filled with large stones; 2 impermeable layer; 3 filtration material; 4 vegetation; 5 water level in the bed; 6 outflow collection zone; 7 drainage pipe; 8 outflow structure with water level adjustment.

5.4 The use constructed wetlands with horizontal sub-surface flow for various types of wastewater

Vertical flow constructed wetlands (VF CWs) (Figure 6.3) were originally introduced by Seidel to oxygenate anaerobic septic tank effluents. However, the VF CWs did not spread as quickly as HF CWs probably because of the higher operation and maintenance requirements due to the necessity to pump the wastewater intermittently on the wetland surface. The water is fed in large batches and then the water percolates down through the sand medium. The new batch is fed only after all the water percolates and the bed is free of water. This enables diffusion of oxygen from the air into the bed. As a result, VF CWs are far more aerobic than HF CWs and provide suitable conditions for nitrification. On the other hand, VF CWs do not provide any denitrification. VF CWs are also very effective in removing organics and suspended solids. Removal of phosphorus is low unless media with high sorption capacity are used. As compared to HF CWs, vertical flow systems require less land, usually 1–3 m^2 PE−1. The early VF CWs were composed of several stages with beds in the first stage fed in rotation. At present, VF CWs are usually built with one bed and the system is called "compact" VF CWs. VF CWs are very often used to treat domestic and municipal wastewater and especially when discharge limits are set for ammonia-nitrogen. However, in the literature, numerous reports have been published on the use of VF CWS for various types of wastewater such as refinery effluent, composting leachate, airport runoff, dairy or cheese production effluent.

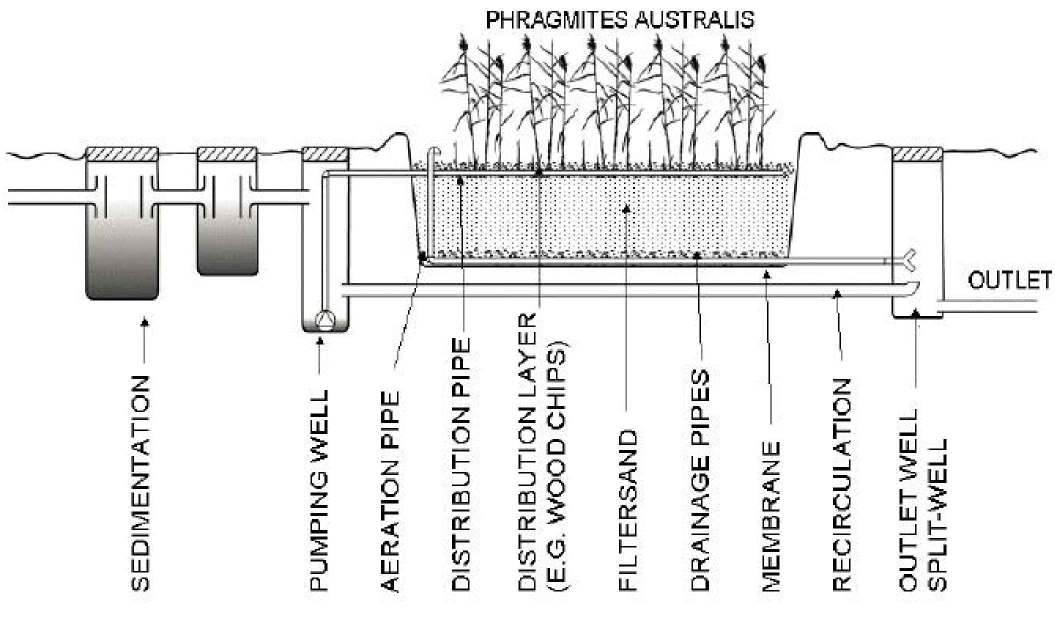

Figure 5.4. Layout of vertical flow constructed wetland system for a single household.

In upflow vertical CWs, the wastewater is fed on the bottom of the wetland. The water percolates upward and then it is collected either near the surface or on the surface of the wetland bed. These systems are commonly used, for example, in Brazil. Recently, the “fill and drain” or “tidal” CWs have been developed. In tidal flow systems the wastewater percolates upwards until the surface is flooded. When the surface is completely flooded, the feeding is stopped, the wastewater is then held in the bed and, at a set time later, the wastewater is drained downwards. After the water has drained from the filtration bed, the treatment cycle is complete and air can diffuse into the voids in the filtration material.

Constructed wetlands with horizontal sub-surface flow (HF CWs) have been used for wastewater treatment for more than 30 years. Most HF CWs have been designed to treat municipal or domestic wastewater. Nowadays, municipal HFCWs focus not only oncommon pollutants but also on special parameters such as pharmaceuticals, endocrine disruptive chemicals or linear alkylbenzen sulfonates (LAS). At present, HF CWs are used to treat many other types of wastewater. Industrial applications include wastewaters from oil refineries, chemical factories, pulp and paper production, tannery and textile industries, abattoir, distillery and winery industries. In particular, the use of HF CWs is becoming very common for treatment of food-processing wastewaters (e.g., production and processing of milk, cheese, potatoes, sugar). HF constructed wetlands are also successfully used to treat wastewaters from agriculture (e.g., pig and dairy farms, fish farm effluents) and various runoff waters (agriculture, airports, highway, greenhouses, plant nurseries). HF CWs have also effectively been used to treat landfill leachate. Besides the use as a single unit, HF CWs are also used in combination with other types of constructed wetlands in hybrid systems.

The technology of wastewater treatment by means of constructed wetlands with horizontal sub-surface flow (HFCWs) was started in Germany based on research by Käthe Seidel commencing in the 1960s and by Reinhold Kickuth in the 1970s. In these systems the wastewater is fed in at the inlet and flows slowly through the porous medium under the surface of the bed in a more or less horizontal path until it reaches the outlet zone where it is collected before leaving via level control arrangement at the outlet (Fig. 1). During this passage the wastewater will come into contact with a network of aerobic, anoxic and anaerobic zones. The aerobic zones occur around roots and rhizomes that leak oxygen into the substrate. HF constructed wetlands have long been used primarily for treatment of municipal or domestic wastewaters. However, at present, constructed wetlands are used for a wide variety of pollution, including agricultural and industrial wastewaters, various runoff waters and landfill leachate. The objective of this paper is to evaluate the use of constructed wetlands with horizontal sub-surface flow for various types of wastewater.

References

Achard F, Eva HD, Stibig H, Mayaux P, Gallego J, Richards T, Malingreau JP (2002) Determination of deforestation rates of the world's humid tropical forests. Science*,* **297**, 999-1002.

Alongi, Daniel M (2002) Present state and future of the world's mangrove forests. Environmental Conservation*,* **29**, 331-349.

Arkema KK, Verutes GM, Wood SA *et al.* (2015) Embedding ecosystem services in coastal planning leads to better outcomes for people and nature. Proc Natl Acad Sci U S A*,* **112**, 7390.

Barbier EB (2011) Coastal Wetland Restoration and the Deepwater Horizon Oil Spill. Vanderbilt Law Review*,* **64**, 1821-1852.

Barbier EB, Cox M (2001) Does Economic Development Lead to Mangrove Loss?: A Cross-Country Analysis. In: *Session on Ecosystem, Conservation and the Environment, Western Economic Association International Meeting.* pp Page.

Barbier EB, Hacker SD, Kennedy C, Koch EW, Stier AC, Silliman BR (2011) The value of estuarine and coastal ecosystem services. Ecological Monographs*,* **81**, 169-193.

Barbier EB, Sathirathai S (2001) *Valuing Mangrove Conservation in Southern Thailand.*

Benjaminj G, Johnm P (2008) Escaping the heat: range shifts of reef coral taxa in coastal Western Australia. Global Change Biology*,* **14**, 513–528.

Birtles P, Cunningham D (2013) The benefits of constructing an undersized wetland. In: *International Water Sensitive Urban Design Conference: Wsud.* pp Page.

Camacho-Valdez V, Ruiz-Luna A, Ghermandi A, Nunes PaLD (2013) Valuation of ecosystem services provided by coastal wetlands in northwest Mexico. Ocean & Coastal Management*,* **78**, 1-11.

Chen J, Xuemin G, Dawei H, Xinghui X (2000) Nitrogen contamination in the Yangtze river system, China. Journal of Hazardous Materials*,* **73**, 107-113.

Chen LZ, Wang YH, Lin GH (2009) Recent progresses in mangrove conservation, restoration and research in China. Journal of Plant Ecology*,* **2**, 45-54.

Cornelisen CD, Thomas F (2006) Water flow enhances ammonium and nitrate uptake in a seagrass community. Marine Ecology Progress*,* **312**, 1-13.

Costanza R, Perez-Maqueo O, Martinez ML, Sutton P, Anderson SJ, Mulder K (2008) The value of coastal wetlands for hurricane protection. Ambio*,* **37**, 241-248.

Cui B, Yang Q, Yang Z, Zhang K (2009) Evaluating the ecological performance of wetland restoration in the Yellow River Delta, China. Ecological Engineering, **35**, 1090-1103.

Davy AJ, Bakker JP, Figueroa ME (2009) Human modification of European salt marshes. University of California Press Berkeley Ca Usa, 311-335.

Duarte CM (2002) The future of seagrass meadows. Environmental Conservation, **29**, 192-206.

Duarte CM, Dennison WC, Orth RJW, Carruthers TJB (2008) The Charisma of Coastal Ecosystems: Addressing the Imbalance. Estuaries & Coasts, **31**, 233-238.

Gedan KB, Silliman BR, Bertness MD (2009) Centuries of human-driven change in salt marsh ecosystems. Ann Rev Mar Sci, **1**, 117-141.

Ghermandi A, Bixio D, Traverso P, Cersosimo I, Thoeye C (2007) The removal of pathogens in surface-flow constructed wetlands and its implications for water reuse. Water Science & Technology A Journal of the International Association on Water Pollution Research, **56**, 207.

Giosan L, Syvitski J, Constantinescu S, Day J (2014) Climate change: protect the world's deltas. Nature, **516**, 31-33.

Guittonny-Philippe A, Masotti V, Höhener P, Boudenne JL, Viglione J, Laffontschwob I (2013) Constructed wetlands to reduce metal pollution from industrial catchments in aquatic Mediterranean ecosystems: a review to overcome obstacles and suggest potential solutions. Environ Int, **64C**, 1.

Haberl H (1997) Human Appropriation of Net Primary Production as an Environmental Indicator: Implications for Sustainable Development. Ambio, **26**, 143-146.

Hassan R, Scholes R, Ash N (2005) Ecosystems and human well-being: current state and trends. Journal of Bacteriology, **1**, 1387-1404.

He Y, Li X, Guo W, Ma Z (2012) Division of labor in rhizomatous species: Comparative performance of native and invasive species in the tidal marshes of the Yangtze River estuary, China. Journal of Experimental Marine Biology & Ecology, **422-423**, 122-128.

Hinrichsen E (2008) Introduction to Aquaculture in the Eastern Cape.

Hughes TP, Bellwood DR, Folke C, Steneck RS, Wilson J (2005) New paradigms for supporting the resilience of marine ecosystems. Trends in ecology & evolution, **20**, 380-386.

Imis V (2008) The World's Mangroves 1980–2005. Fao Forestry Paper, 703-704.

Jiang T-T, Pan J-F, Pu X-M, Wang B, Pan J-J (2015) Current status of coastal wetlands in China:

Degradation, restoration, and future management. Estuarine, Coastal and Shelf Science, **164**, 265-275.

Kawahata H, Suzuki A, Goto K (1997) Coral reef ecosystems as a source of atmospheric CO2: evidence from PCO2 measurements of surface waters. Coral Reefs, **16**, 261-266.

Koch EW, Ackerman JD, Verduin J, Keulen MV (2006) Fluid Dynamics in Seagrass Ecology—from Molecules to Ecosystems. Seagrasses Biology Ecology & Conservation, 193-225.

Koch EW, Barbier EB, Silliman BR *et al.* (2009) Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. Frontiers in Ecology & the Environment, **7**, 29-37.

Koh CH (1997) Korean megatidal environments and tidal power projects: Korean tidal flats - biology, ecology and land uses by reclamations and other feasibilities. La Houille Blanche, **52**, 66-78.

Kubín P, Melzer A (1997) Chronological relationship between eutrophication and reed decline in three lakes of Southern Germany. Folia Geobotanica & Phytotaxonomica, **32**, 443-443.

Leonardi N, Fagherazzi S (2015) Effect of local variability in erosional resistance on large‐scale morphodynamic response of salt marshes to wind waves and extreme events. Geophysical Research Letters, **42**, 5872-5879.

Li HM, Tang HJ, Shi XY, Zhang CS, Wang XL (2014) Increased nutrient loads from the Changjiang (Yangtze) River have led to increased Harmful Algal Blooms. Harmful Algae, **39**, 92-101.

Li X, Bellerby R, Craft C, Widney SE (2018) Coastal wetland loss, consequences, and challenges for restoration. Anthropocene Coasts, **1**, 1-15.

Macfarlane GR, Pulkownik A, Burchett MD (2003) Accumulation and distribution of heavy metals in the grey mangrove, Avicennia marina (Forsk)Vierh: biological indication potential. Environ Pollut, **123**, 139-151.

Mackenzie RA, Bruland GL (2012) Nekton Communities in Hawaiian Coastal Wetlands: The Distribution and Abundance of Introduced Fish Species. Estuaries and Coasts, **35**, 212-226.

Mackinnon J, Verkuil YI, Murray NJ (2012) IUCN situation analysis on East and Southeast Asian intertidal habitats, with particular reference to the Yellow Sea (including the Bohai Sea).

Marani M, D'alpaos A, Lanzoni S, Santalucia M (2011) Understanding and predicting wave erosion of marsh edges. Geophysical Research Letters, **38**, 759-775.

Marsden I (2012) Human Impacts on Salt Marshes: A Global Perspective. Journal of Experimental Marine Biology & Ecology, **412**, 141-141.

Mh J, Ap J, Hm R (2000) Influence of environmental factors on the growth and interactions between salt marsh plants: Effects of salinity, sediment and waterlogging. J Ecol, **88**, 492-505.

Mitsch WJ, Gosselink JG (2008) *Wetlands,* New York USA, Van Nostrand Reinhold.

Moberg F, Rönnbäck P (2003) Ecosystem services of the tropical seascape: interactions, substitutions and restoration. Ocean & Coastal Management, **46**, 27-46.

Morgan PA, Burdick DM, Short FT (2009) The Functions and Values of Fringing Salt Marshes in Northern New England, USA. Estuaries & Coasts, **32**, 483-495.

Mou X, Liu X, Yan B, Cui B (2015) Classification system of coastal wetlands in China. Wetland Science, **13**, 19-26.

Murray NJ, Clemens RS, Phinn SR, Possingham HP, Fuller RA (2014) Tracking the rapid loss of tidal wetlands in the Yellow Sea. Frontiers in Ecology and the Environment, **12**, 267-272.

Naylor RL, Goldburg RJ, Primavera JH *et al.* (2000) Effect of aquaculture on world fish supplies. Nature, **405**, 1017-1024.

Niu ZG, Gong P, Cheng X *et al.* (2009) Geographical characteristics of China's wetlands derived from remotely sensed data. Science in China, **52**, 723-738.

Orth RJ, Carruthers TJB, Dennison WC *et al.* (2006) A Global Crisis for Seagrass Ecosystems. Bioscience, **56**, 987-996.

Peterson SB, Teal JM (1996) The role of plants in ecologically engineered wastewater treatment systems. Ecological Engineering, **6**, 137-148.

Romero J, Lee KS, Pérez M (2006) Nutrient Dynamics in Seagrass Ecosystems. 227-254.

Rooth JE, Stevenson JC (2000) Sediment deposition patterns in Phragmites australis communities: Implications for coastal areas threatened by rising sea-level. Wetlands Ecology & Management, **8**, 173-183.

Rose JM, Bricker SB, Ferreira JG (2015) Comparative analysis of modeled nitrogen removal by shellfish farms. Mar Pollut Bull, **91**, 185-190.

Rousseau DPL, Lesage E, Story A, Vanrolleghem PA, Pauw ND (2008) Constructed wetlands for water reclamation. Desalination, **218**, 181-189.

Sato M, Koh CH (2004) *Biological Richness Of The Asian Tidal Flats And Its Crisis By Human Impacts*, Springer Netherlands.

Shu G, Du YF, Xie WJ, Gao WH, Wang DD, Wu XD (2014) Environment-ecosystem dynamic processes of Spartina alterniflora salt-marshes along the eastern China coastlines. Science China Earth Sciences, **57**, 2567-2586.

Sierszen ME, Morrice JA, Trebitz AS, Hoffman JC (2012) A review of selected ecosystem services provided by coastal wetlands of the Laurentian Great Lakes. Aquatic Ecosystem Health & Management, **15**, 92-106.

Spalding M, Blasco F, Field C (1997) World mangrove atlas. Okinawa Japan International Society for Mangrove Ecosystems.

Sun Z, Sun W, Tong C, Zeng C, Yu X, Mou X (2015) China's coastal wetlands: conservation history, implementation efforts, existing issues and strategies for future improvement. Environ Int, **79**, 25-41.

Temmerman S, Kirwan ML (2015) Building land with a rising sea. Science, **349**, págs. 588-589.

Tessler ZD, Grossberg M, Gladkova I, Aizenman H, Syvitski JP, Foufoula-Georgiou E (2015) Profiling risk and sustainability in coastal deltas of the world. Science, **349**, 638-643.

Thampanya U, Vermaat JE, Sinsakul S, Panapitukkul N (2006) Coastal erosion and mangrove progradation of Southern Thailand. Estuarine Coastal & Shelf Science, **68**, 75-85.

Tong Y, Zhao Y, Zhen G *et al.* (2015) Nutrient Loads Flowing into Coastal Waters from the Main Rivers of China (2006–2012). Scientific Reports, **5**, 16678.

Turner RE, Swenson EM, Milan CS, Lee JM, Oswald TA (2004) Below-ground biomass in healthy and impaired salt marshes. Ecological Research, **19**, 29-35.

Valiela I, Bowen JL, York JK (2001) Mangrove forests: one of the world's threatened major tropical environments. Bioscience, **51**, 807-815.

Valiela I, Kinney E, Culbertson J, Peacock E, Smith S (2009) *Global losses of mangroves and salt marshes*.

Valipour A, Ahn YH (2016) Constructed wetlands as sustainable ecotechnologies in decentralization practices: a review. Environmental Science & Pollution Research, **23**, 1-18.

Vriend HJD, Koningsveld MV, Aarninkhof SGJ, Vries MBD, Baptist MJ (2015) Sustainable hydraulic engineering through building with nature. Journal of Hydro-environment Research, **9**, 159-171.

Vymazal J (1998) Removal mechanisms and types of constructed wetlands. Constructed Wetlands for wastewater treatment in Europe, 17-66.

Vymazal J (2013) Emergent plants used in free water surface constructed wetlands: A review. Ecological Engineering, **61**, 582-592.

Walmsley JJ (2002) Framework for Measuring Sustainable Development in Catchment Systems. Environmental Management, **29**, 195-206.

Walters BB, Rönnbäck P, Kovacs JM *et al.* (2009) Ethnobiology, socio-economics and management of mangrove forests: A review. Aquatic Botany, **89**, 220-236.

Wang W, Liu H, Li Y, Su J (2014) Development and management of land reclamation in China. Ocean & Coastal Management, **102**, 415-425.

Worm B, Barbier EB, Beaumont N *et al.* (2006) Impacts of Biodiversity Loss on Ocean Ecosystem Services. Science, **314**, 787-790.

Wu H, Zhang J, Ngo HH *et al.* (2015) A review on the sustainability of constructed wetlands for wastewater treatment: Design and operation. Bioresource Technology, **175**, 594-601.

Yang SL (1998) The Role of Scirpus Marsh in Attenuation of Hydrodynamics and Retention of Fine Sediment in the Yangtze Estuary. Estuarine Coastal & Shelf Science, **47**, 227-233.

Yang SL, Li H, Ysebaert T *et al.* (2008) Spatial and temporal variations in sediment grain size in tidal wetlands, Yangtze Delta: On the role of physical and biotic controls. Estuarine Coastal & Shelf Science, **77**, 657-671.

Zimmerman RJ, Minello TJ, Rozas LP (2000) *Salt Marsh Linkages to Productivity of Penaeid Shrimps and Blue Crabs in the Northern Gulf of Mexico*, Springer Netherlands.