Chapter 19

RESTORATION OF SALT MARSHES

Jan P. Bakker
19.1 INTRODUCTION: THE HISTORICAL CONTEXT

This chapter deals with tidal salt marshes adjacent to intertidal flats. Salt marshes and intertidal flats occur along the edges of shallow seas with soft sediment bottoms where the tidal range is considerable, at least a meter or so (van de Kam et al. 2004), but also in the absence of tides, as in the Baltic Sea. Low-lying intertidal areas are inundated at least once a day, and make a place for more irregularly inundated areas of salt marsh higher up. In tropical areas, and even some benign temperate areas such as northernmost New Zealand, the upper parts of intertidal areas may be covered by mangrove forests rather than salt marsh. Such mangroves have the tendency also to cover the regularly inundated parts of intertidal soft sediments, thus reducing the extent of mudflats in many tropical areas. No intertidal deposits or salt marshes occur at high latitudes (farther north than 70–73°N). Here coastlines are either ice-covered for most of the year or disturbed by moving ice too frequently for soft sediment deposits or vegetation to build up. Hence, salt marshes are mainly found in the temperate zone.

Salt marshes and intertidal flats are under complex natural controls. The main external controls for the tidal lands are the sea level and sediment supply regimes. Upward sea level movements and auto-compaction – that is, diminishing of the volume of the sediment – combine to provide accommodation space within which marshes build upwards. Mineralogenic marshes consist of a vegetated platform dissected typically by extensive networks of blind-ended, branching tidal creeks. The flow-resistant surface vegetation both traps and binds tidally introduced mineral sediment, but also contributes an organic component of indigenous origin to the deposit. When the sea level next to mineralogenic marshes becomes stable or falls, however, in response to century- or millennium-scale fluctuations, the organic sediment component becomes dominant and mineralogenic marshes are transformed into organogenic ones. Because peat is such a porous and permeable sediment, and there is little or no tidal inundation, organogenic marshes in north-western Europe typically lack surface channels for intertidal drainage (Allen 2000). At present very little peat marsh occurs in Europe, except for the Baltic Sea (Dijkema 1984). In contrast, the north-eastern coast of North America features large coastal peat deposits (Niering 1997). Along the south-eastern coast of North America, the vertical accretion rate of salt marshes is directly related to the accumulation of organic matter, rather than to inorganic matter (Turner et al. 2000).

Global sea level rise after the last glacial period caused a poorer drainage of the coastal hinterland and a subsequent rise of the groundwater table in the adjacent low-lying inland zones which became marshy, allowing peat formation over the underlying Pleistocene subsurface. As a consequence of increased marine influence, the freshwater marsh transformed into areas of tidal salt marshes and intertidal mudflats, or brackish lagoons. As a result, the basal peat layers were covered by marine sediments, and with continuous sea level rise the area became totally submerged. This transgressive process continued until the mid-Holocene, after which the coastline stabilized more or less at its present position. As a result of the decline in sea level rise, sedimentary processes became increasingly dominant (Esselink 2000).

Coastal regions around the world are not only affected by natural processes. In north-western Europe, the coastal zone became increasingly shaped by human activities undertaken to increase agricultural land area, transport links and urbanization and coastal defence. Most human activities in and exploitation of intertidal flats were relatively unintrusive for a long time, consisting primarily of small-scale fishing and the taking of shellfish by hand. With the advent of industrialization, however, over the twentieth century, and the use of large nets and dredges, human exploitation patterns of intertidal flats have come to influence the natural processes a great deal indeed. It is not entirely clear whether the same can be said for salt marshes, where, embankments aside, grazing by domestic animals has been the main human factor. Loss of extent has had a significant impact on salt marshes, resulting in truncation of the upper zone. Canalization and dredging of estuaries resulted in widening and deepening of channels and loss of the pioneer zone. Both resulted in loss of associated species. It is quite possible that the grazing by domestic animals has replaced the grazing that took place before human times by wild large herbivores. The Baltic and southern European coasts have been less affected by coastal defence works. See Davy et al. (2009) for an overview on embankments and land claim along the European coastline.

In Europe, the concepts of ‘natural’ and ‘seminatural’ salt marshes have been defined for the international Wadden Sea (Esselink et al. 2009). Natural salt
marshes feature undisturbed geomorphological conditions and have no history of direct management. They show a natural drainage system with meandering creeks and levees with higher elevation than the adjacent depressions and no agricultural exploitation (Plate 19.1). Grazing can occur with natural grazers such as geese and hares. Natural or pristine salt marshes are very rare, and occur in Europe in sandy back-barrier conditions or in parts of former Wadden Sea bays along the mainland coast. Two groups of seminatural marshes can be considered. Some seminatural marshes are similar to natural marshes with natural draining creeks, but include livestock grazing or mowing (Plate 19.1). An other group of seminatural marshes features artificial ditches, sedimentation fields, defences against erosion and often land use. The sedimentation fields are areas surrounded by 200 m × 200 m brushwood groynes that reduce water velocity, with subsequently higher sedimentation of suspended material. These marsh types are affected in their geomorphological conditions by artificial drainage and/or by measures to enhance livestock grazing or cutting (Plate 19.2). Seminatural marshes are found on barrier islands, in foreland clay marshes and in marshes with sedimentation fields and an artificial drainage system (i.e. ditches) (Table 19.1). Characteristic salt marsh plant species can be present in all three salt marsh types. However, their abundance in typical salt marsh communities and their spatial arrangement in the vegetation structure can be affected by land use. Intensive livestock grazing results in homogeneous short sward, whereas abandoned salt marshes are often characterized by uniform tall vegetation. Moderate grazing features most variation in vegetation structure by combining short and tall canopy.

Understanding the historical context of the development of tidal salt marshes along intertidal flats is a prerequisite for any fruitful discussion about the perspectives of nature conservation and restoration in these systems. In this chapter, I focus on concepts, which makes it easier to understand salt marsh systems and the possibilities for restoration than by presenting many examples of restoration from all over the world. Because of my familiarity with western Europe, many cases will be from that continent.

19.2 CHARACTERISTICS OF SALT MARSHES

Abiotic conditions, especially elevation, affect the duration and frequency of inundation by sea water. Hence, they affect the zonation along the elevational gradient of salt marsh ecosystems. The succession of the communities on estuarine marshes is driven by vertical accretion. However, the successions of the communities on sandy marshes is nutrient-driven (de Leeuw et al. 1993; Olff et al. 1997). However, salt marsh vegetation is governed not only by bottom-up physical factors, such as tidal inundations, salinity and soil nutrient concentrations, but also by top-down processes such as grazing animals.

19.2.1 Abiotic conditions, zonation and succession

The driving bottom-up control in salt marsh development is the tidal amplitude, causing inundation and subsequent sedimentation of silt. The mean spring-tidal range in Europe varies from 12.3 m in estuaries to 1.6 m in the Wadden Sea (Allen 2000), to nearly zero in the Baltic Sea. Transplant experiments in Alaska have demonstrated that the abiotic conditions drive plant communities at low elevations and higher inundation frequency and duration, whereas at higher elevation interspecific competition drives plant communities (Snow & Vince 1984). Therefore, grazing may strongly affect plant communities at higher elevation by removal of biomass and subsequent spreading of communities of the low marsh towards higher on the elevational gradient (Bakker 1985).

The pioneer zone of salt marshes consists of annual plant species, and they do not trap sediment. The perennial grasses Spartina anglica and Puccinellia maritima at the low salt marsh catch sediment, whereas erosion can take place of unvegetated soil (Langlois et al.
In the *Festuca rubra* zone, higher up the salt marsh with less inundation, the rate of sedimentation is lower than in the *P. maritima* zone (Andresen et al. 1990). A typical salt marsh zonation along the east coast of the United States shows from the pioneer zone towards higher elevation: *Spartina alterniflora*, *S. patens*, *Juncus gerardi* and *Iva frutescens* (Bertness et al. 2009). Sedimentation patterns show spatial variation. Over comparatively wide marshes, a landward decrease of sedimentation was found in seminatural mainland marshes in Sussex, United Kingdom (Reed 1988), along the Westerschelde, the Netherlands (Temmerman 2003), in a back barrier marsh at Skallingen, Denmark (Bartholdy 1997) and in seminatural marshes with sedimentation fields in Germany (Schröder et al. 2002) and in the Dollard, the Netherlands (Esselink et al. 1998). Superimposed on the large-scale differences from low to high marsh, the rate of sedimentation also declines away from creeks and ditches (Figure 19.1). Moreover, higher rates were

![Figure 19.1](image_url)

**Figure 19.1** (a) Levee development near a minor creek at a distance of 750 m from intertidal mudflats and (b) annual vertical accretion rate (means ± S.E.M.) at different distances from a main creek as a function of the distance from the intertidal mudflats. In (a): •, 1984; ○, 1991. Points in the left-hand panel at higher elevations were located on the neighbouring levee of the next minor creek. MHT, mean high tide. (Modified from Esselink et al. 1998. Reproduced by permission of *Journal of Coastal Research.*
found on ungrazed than on heavily grazed seminatural marshes with sedimentation fields in the Dollard (Esselink et al. 1998). Dense, tall vegetation positively affects the rate of sedimentation (Leonard et al. 1995). Detailed measurements have revealed vertical accretion during winter periods, but shrinkage during the dry summer period (Erchinger et al. 1994).

The vertical accretion results in succession from pioneer communities, via low-salt marsh communities to high-salt marsh communities. This successional sequence, derived from aerial photographs, is mirrored in the spatial zonation from pioneer to low and high marsh in mainland estuarine marshes in the southwestern Netherlands (de Leeuw et al. 1993). In contrast, long-term permanent plot studies revealed different successional patterns at the low and high marsh of the barrier island of Terschelling, the Netherlands, where low-marsh plant communities did not transform into high-marsh communities (Leendertse et al. 1997). Hence, the zonation does not mirror succession in back barrier salt marshes, but reflects the underlying geomorphology of the sandy base elevation (de Leeuw et al. 1993; Olff et al. 1997). At many older ungrazed marshes in north-western Europe, the tall grass *Elytrigia atherica* is taking over gradually. With increasing age of the salt marsh, *E. atherica* is spreading downwards along the elevational gradient. It appears that the grass can cope with the salinity stress at lower elevations when more nutrients are available (Olff et al. 1997). Salt marshes are nitrogen-limited systems (see references in Davy et al. 2009). Nitrogen supply can affect the competitive relations of marsh plants and hence has important consequences for the abundance and distribution across marsh landscapes (Levine et al. 1998). The lower tidal boundaries of marsh plant distributions are generally set by physical stress, whereas the upper boundaries of plants are set by competitive exclusion (Snow & Vince 1984).

The influence of fresh water in mainland estuaries, discharged by rivers from the hinterland, creates a gradient of decreasing salinity inland. The vegetation features *Scripus maritimus* and especially *Phragmites australis*, as in estuarine marshes of the Wadden Sea (Esselink 2000) and the Baltic Sea in Europe (Dijkema 1990), and the Atlantic coast in the United States (Bertness 1999).

In tropical regions, mangrove forests and salt marshes compete for space along a latitudinal gradient. In north-east Queensland, Australia, mangroves may be up to 30 m, but in southern Victoria they take the form of low shrubs about 1 m tall (Thomson et al. 2009). Salt marshes can occur where mangrove growth is inhibited, which often occurs by cutting the trees. Mangrove cutting results in increased soil salinity and hence bare intertidal flats or herbaceous vegetation (Costa et al. 2009).

### 19.2.2 Plant-animal interactions

The barrier island of Schiermonnikoog, the Netherlands, extends eastward, thus featuring a chronosequence from east to west. This phenomenon provides an opportunity to study salt marsh development over a period of some 200 years. A positive correlation is found between the thickness of the clay layer and both the nitrogen pool (Olff et al. 1997) and the availability of nitrogen for plants (Bakker et al. 2005). Hence, the chronosequence represents a productivity gradient featuring low-statured plants in early stages and tall grass in later successional stages with thicker clay layer, and a decrease in the number of plant species (Bakker et al. 2002b). As a result, the forage quantity for natural herbivores such as spring-staging geese and resident hares increases, but the quality, expressed as the leaf-stem ratio, decreases (van de Koppel et al. 1996). Hence, geese and hares are ‘evicted’ by vegetation succession, namely dominance of the tall forb *Atriplex portulacoides* and the tall grass *Elytrigia atherica*. Exclusion experiments revealed that geese can forage on the salt marsh only when hares occur and eat the tall plants, hence hare facilitate for geese, and they can retard the succession with a few decades (Kuijper & Bakker 2005). Indeed, back barrier island marshes without hares – such as on Mellum, Germany – appeared to be completely dominated by tall stands of vegetation (Kuijper & Bakker 2003).

Salt marsh ecosystems have long been considered as being controlled by physical bottom-up factors such as salinity, tidal inundations and soil nutrient concentrations. The above-mentioned studies with natural herbivores stress the importance of consumers and food web characteristics of salt marshes. Hence, top-down control is an essential element in understanding salt marsh ecosystems (Silliman et al. 2009), including human exploitation of salt marshes by livestock grazing. The role of livestock grazing is further discussed in section 19.3.
19.3 THREATS AND DISTURBANCES

Embarkment is the most definitive threat for the occurrence of salt marshes, and occurs worldwide. Other threats are erosion, reduction of the tidal amplitude and decrease in salinity of inundating water. Intensive livestock grazing is considered a threat in many parts of the world. Overgrazing results in short uniform swards without tall vegetation for plant and animal species to survive. Also, the potential for sediment capture and wave attenuation has reduced. Moreover, evaporation and salinization take place. However, in Europe most salt marshes have always been exploited for either grazing or cutting. As most salt marshes are agriculturally exploited, the cessation of exploitation is a sudden change in long-term management and thus causes disturbance. After abandonment, a single plant species can become dominant with subsequent losses of characteristic halobiontic species (i.e. groups or organisms tolerating saline conditions). However, some exceptions will be discussed below.

19.3.1 Embankments

In Europe, the first seawalls were constructed against the increased risk of flooding in the tenth century in the northern Netherlands (Oost & de Boer 1994), and during the eleventh century in adjacent Germany (Behre 1995). The entire North Sea coastline of the Netherlands and Germany was protected in the thirteenth century. The first seawalls were constructed in the salt marshes above the level of mean high tide, and hence, in most places, a strip of unprotected salt marsh remained in front of the seawalls. New marshes developed after the construction of the first seawalls, especially in sheltered bays (Oost & de Boer 1994). The new marshes, which originated during the twelfth-fourteenth centuries, may have evolved without human intervention. However, human intervention was very likely from at least the seventeenth century onwards. Several techniques have been applied to promote both vertical accretion and horizontal expansion of salt marshes. At present, the majority of mainland salt marshes are seminatural, resulting from sedimentation fields with intensively engineered ditching for drainage, such as those currently found in Denmark, Germany, the Netherlands (Dijkema 1984) and the United States (Pye 2000). Back barrier marshes developed in the shelter of dunes, but also in the shelter of artificial sand dikes during the twentieth century (Dijkema 1987). These marshes always had a natural drainage pattern with creeks.

19.3.2 Erosion, reduction of salinity and tidal amplitude

Salt marshes have been eroding rapidly in southwestern England during the past 150 years, and particularly in the past few decades. The mechanisms of erosion include landward recession of the marsh edge, wave erosion of the marsh surface, internal dissection due to enlargement and coalescence of tidal creeks and mud basins and direct removal due to human activities. Increased wind and wave energy is supposed to contribute most strongly to erosion. Increased mean sea level and tidal range are underlying factors leading to coastal ‘squeezing’ of salt marshes between the sea and seawalls (Pye 2000). Establishing seawalls on the intertidal flats makes them more vulnerable to erosion. In seminatural marshes with sedimentation fields, the abandonment of accretion works results in retreating of the marsh edge (Esselink 2000).

As one result of future rising sea levels caused by global climate warming, widespread salt marsh erosion is predicted. A long-term experiment was carried out on this subject, at the Wadden Sea island of Ameland, the Netherlands, in a seminatural marsh system with drainage assured by natural creeks. As a result of gas extraction, soil subsidence of 10 cm was observed over a 15-year period, affecting both the low- and high-salt marsh (Dijkema 1997). The net elevation of low-marsh plots did not change, indicating that sedimentation kept pace with subsidence. In contrast, net elevation of the high-marsh plots decreased by 10 cm, indicating that no extra sedimentation took place. Neither in the lower nor in the upper plots of the salt marsh did vegetation change (Dijkema 1997).

Another important cause of losses of salt marshes is coastal protection by shortening the coastline of estuarine coasts. Desalinization causes the transformation of salt marsh communities into communities adapted to freshwater conditions. Continued grazing by livestock retards the losses of halophytic plant species (Westhoff & Sykora 1979). Moreover, grazing benefits short turfs that are favoured by winter-staging geese, as in the former salt marshes. Undisturbed succession results in scrub and forest with characteristic bird species (van Wieren 1998).
254 Restoration ecology

Not all estuaries are dammed. In 1986 a sluice–gate barrier was completed in the mouth of the Oosterschelde estuary, the Netherlands. It can be closed during storm surges. Although the barrier allows tidal exchange, the tidal flow has been restricted. This caused a 26 cm decrease of the mean high tide and hence a decreased inundation frequency of the marsh. Most plant species had moved down along the elevational gradient (de Leeuw et al. 1994).

More small-scale processes took place in north-eastern United States. During the twentieth century, about 2000 ha (30%) of Connecticut’s tidal marshes were degraded or lost through coastal development. Tidal flow to many marshes was restricted by the construction of impoundments, producing microtidal environments in which the non-native haplotype of Phragmites australis or, less frequently, Typha angustifolia became established at the expense of typical tidal marsh communities. In addition to these human influences, P. australis has also invaded brackish tidal marshes in the lower Connecticut River system where salinity levels are often reduced by freshwater inputs (Fell et al. 2000).

19.3.3 Exploitation

Salt production is one of the most ancient estuarine industries. The oldest indication is a Bronze Age salt-evaporating hearth (1400–1130 BCE) in Essex, United Kingdom, and there were many Iron Age, Roman and medieval salterns around the English coast (Fleming 2004). The first colonists on salt marshes in the north-west European mainland settled on the highest parts of the marsh, on levees along watercourses, in the seventh century CE. Farmsteads were initially built on the marsh bed. In response to increased risk of flooding, people started to build their dwellings on artificial mounds. The number of mounds along the northern coast of the Netherlands suggests intensive exploitation of the salt marshes (Figure 19.2). When salt marshes extended seaward, new settlements were built.

![Figure 19.2](image-url)
on the younger marshes until the entire coastline was protected by seawalls in the thirteenth century. Initially arable crops were grown on the levees, but ditching and the construction of embankments, dated from the first century BCE to the second century CE, allowed crops to be grown on the salt marshes. The majority of the marshes were exploited for livestock grazing from the early settlements onwards, and haymaking from the first–third centuries onwards. The high frequency of subfossils of Juncus gerardi and the low frequency of Elytrigia spp. found in artificial mounds led to the conclusion that unexploited salt marshes were scarce during most of the occupation period (Esselink 2000). Nowadays salt marshes in the Wadden Sea that have never been grazed by livestock are of recent origin, found only at the eastern point of the Wadden Sea islands; the oldest one, since 1930, is found on the island of Terschelling, the Netherlands.

Marshes that had accumulated enough sediment were embanked. The incentives for embankments have gradually changed during the twentieth century from land claims for agriculture to coastal protection. During the early twentieth century, large-scale accretion works with sedimentation fields have been started to create salt marshes that were to be reclaimed for agriculture, but some decades later this was no longer economically feasible due to changes in both socioeconomic conditions and agricultural policies. From about the 1970s, there has been a growing recognition that the remaining salt marshes, though largely ‘developed’ as a result of human intervention, have an important nature conservation interest (Esselink 2000). Increasing areas of seminatural salt marshes with sedimentation fields were designated as nature reserves in Denmark, Germany and the Netherlands, and were included in national parks. Erosion is not allowed because of coastal defence, and new sedimentation fields are not promoted, as they reduce the area of intertidal flats. Hence, existing salt marshes are squeezed between the seawall or artificial sand dikes and the intertidal flats. The existing marshes undergo a process of maturation, and pioneer and young marshes become lost as a result of decreased dynamics. This process is still enhanced when livestock grazing on the marshes ceases. Nowadays only about 40% of the salt marshes of the Wadden Sea support livestock grazing (Esselink et al. 2009).

Long-term (>25 years) livestock enclosures in seminatural back barrier marshes in the Wadden Sea revealed that the variation in plant communities along the elevational gradient decreased. Especially at the mid- and higher marsh, plant species diversity declined (Bos et al. 2002). Similar changes were recorded in long-term ungrazed seminatural marshes. At sites with fast colonization of Elytrigia atherica in a seminatural salt marsh, the typical zonation of entomofauna communities along an elevational gradient disappeared (Figure 19.3) and characteristic halobionic species were replaced by common inland species of tall forb communities (Andresen et al. 1990). In contrast, the invasion of E. atherica in salt marshes in western France did reveal an increase in noncoastal spider species, web-building and cursorial spiders, but did not interfere with resident species distributions, finally resulting in higher species densities and species richness (Pétillon et al. 2005). The grazing intensity of winter-staging geese was less in long-term ungrazed than in grazed salt marshes in the Wadden Sea (Figure 19.4; Bos et al. 2005). Although goose numbers declined, especially in autumn, in the 10-year-ungrazed part of the Hamburger Hallig, Germany, the numbers of some breeding birds increased (Stock & Hofeditz 2002).

North American salt marshes have a different history. Along the eastern coast, tidal marshes formed within the last 3000–4000 years as sea level rise slowed to about 1 mm yr$^{-1}$, favouring the establishment of the initial colonizer Spartina alterniflora, a 1–2 m tall grass (Niering 1997). The accumulation of organic matter controls the accumulation of inorganic matter, not the reverse. Below-ground plant material is very important in maintaining salt marshes once they are established. During the seventeenth to nineteenth centuries CE, salt marshes were mown, grazed, ditched and embanked in order to make them more suitable for agricultural exploitation. In conjunction with these early activities, some ditching and diking were done to regulate tidal flooding, and increase the profitability of cattle grazing and hay production. However, these impacts were minor compared to those that followed the Industrial Revolution (1850s) when, with increased mechanization, marshes were dredged for marinas, filled for development, ditched for mosquito control, filled with dredge spills and tidal-gated in order to prevent upland flooding. The subsequent invasion of Phragmites australis in disturbed coastal wetlands resulted in outcompeting of the native plant species and desiccation as a result of strong transpiration out of its habitat (Bertness et al. 2009). Wetland protection laws since the 1970s, and no-net-loss policies, have led
to restoration efforts. Moreover, with the past decade Open Marsh Water Management has been widely practised using biological control, which favours small fish to control mosquitoes and simultaneously promotes restoration (Niering 1997).

South American (Costa et al. 2009) and Australasian (Thomson et al. 2009) coastal salt marshes have a natural geomorphology. They are not intersected with an intensive ditching system. The main human impact is heavy exploitation. The effect of intensive livestock grazing, resulting in short swards and possible erosion, in combination with drought is regarded as a disturbance.

19.4 RESTORATION OF SEMINATURAL SALT MARSHES

19.4.1 Targets for restoration of salt marshes

A list of structural attributes that best indicate the status of restored salt marshes was given by Zedler and Lindig-Cisneros (2000): (1) vegetation structure: species composition, stem density, percent cover and biomass or total stem length are probably the most
widespread descriptors of salt marsh structure, and a tall canopy is regarded as important for birds and invertebrates; (2) soil attributes: texture, nutrients and organic matter are regarded important as far as they contribute to a tall canopy of *Spartina* spp.; (3) invertebrate and fish assemblages: the presence of species alone is not very useful, and abundance and some measure of functioning in the foodweb are preferred; and (4) complex topography: tidal creek networks are important to the distribution of plant species. Creek density and the relative distribution of creeks of different order (first, second and third) can be measured. It is hoped that relatively easily measurable structural attributes can be related to functional attributes.

Apart from abiotic and diversity targets, other ecosystem goods and services may play a role in setting targets for salt marsh restoration. Some of the goods and services that salt marshes provide are considered to be ‘high’ importance by an expert panel (Jones et al. 2011; see Table 19.2). Salt marshes were estimated to

**Table 19.2** Goods and benefits provided by ecosystem services from (semi)natural salt marshes (after UK National Ecosystem Assessment: http://uknea.unept-wcmc.org) (see Jones et al. 2011) and interpretation for restored salt marshes by an expert team (see Jones et al. 2011) and the author of this chapter. Class (high, some, none) indicates importance of each good and benefit.

<table>
<thead>
<tr>
<th>Ecosystem service</th>
<th>Goods and benefits</th>
<th>Salt marsh (semi) natural</th>
<th>Salt marsh restored</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>P: provisioning</strong></td>
<td>Meat: sheep/cattle/ fish</td>
<td>High</td>
<td>High</td>
</tr>
<tr>
<td></td>
<td>Wild food: Salicornia/other plants/ fish/wildfowl</td>
<td>Some</td>
<td>Some</td>
</tr>
<tr>
<td></td>
<td>Wool (sheep)</td>
<td>Some</td>
<td>Some</td>
</tr>
<tr>
<td></td>
<td>Genetic resources of rare breeds, crops</td>
<td>Some</td>
<td>Some</td>
</tr>
<tr>
<td></td>
<td>Turf/peat cutting</td>
<td>Some</td>
<td>None</td>
</tr>
<tr>
<td></td>
<td>Military use</td>
<td>Some</td>
<td>None</td>
</tr>
<tr>
<td></td>
<td>Industrial use: pipeline landfall</td>
<td>Some</td>
<td>None</td>
</tr>
<tr>
<td><strong>R: regulating</strong></td>
<td>Carbon sequestration</td>
<td>High</td>
<td>Some</td>
</tr>
<tr>
<td></td>
<td>Sea defence</td>
<td>High</td>
<td>High</td>
</tr>
<tr>
<td></td>
<td>Preventing soil erosion</td>
<td>High</td>
<td>Some</td>
</tr>
<tr>
<td></td>
<td>Immobilization of pollutants</td>
<td>High</td>
<td>Some</td>
</tr>
<tr>
<td><strong>C: cultural</strong></td>
<td>(P) High diversity, or rare/unique plants, animals and birds, insects</td>
<td>High</td>
<td>Some</td>
</tr>
<tr>
<td></td>
<td>(P) Ecosystem-specific protected areas</td>
<td>High</td>
<td>High</td>
</tr>
<tr>
<td></td>
<td>(R) Nursery grounds for fish</td>
<td>High</td>
<td>High</td>
</tr>
<tr>
<td></td>
<td>(R) Breeding, over-wintering, feeding grounds for birds</td>
<td>High</td>
<td>High</td>
</tr>
<tr>
<td></td>
<td>Water filtration: groundwater, surface flow, seawater</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Sites of religious/cultural significance; World Heritage Sites; folklore; TV and radio programmes and films</td>
<td>Some</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>Paintings, sculpture, books</td>
<td>High</td>
<td>Some</td>
</tr>
<tr>
<td></td>
<td>Many opportunities for recreation: incl. sunbathing, walking, camping, boating, fishing, birdwatching etc.</td>
<td>High</td>
<td>Some</td>
</tr>
<tr>
<td></td>
<td>Opportunities for exercise, local meaningful space, wilderness, personal space</td>
<td>Some</td>
<td>Some</td>
</tr>
<tr>
<td></td>
<td>Resource for teaching, public information, scientific study</td>
<td>High</td>
<td>High</td>
</tr>
</tbody>
</table>
provide more economic value per unit area than most other ecosystems if tourism, carbon storage and coastal defence were combined. An attempt is also made to indicate to which extent these services can be regained along with salt marsh restoration (Jones et al. 2011).

The salt marsh harbouring all these conditions features (1) tidal inundation with salt water and drainage of fresh water from the hinterland, (2) zonation of communities from pioneer zone towards high salt marsh, (3) sufficient width from low intertidal flats towards high elevation to include creeks, creek bank levees and depressions and (4) geomorphological dynamics (i.e. continuous building of young marsh, or periodic erosion followed by building of young marsh) (Esselink et al. 2009). Such salt marshes show the highest richness of characteristic saline and brackish plant and animal species, and the habitat types and characteristic plant communities to be protected or restored in the Natura 2000 system (EUNIS 2006). This is a supranational instrument providing a common framework for the conservation of plant and animal species and their natural habitats within the 27 member states of the European Union. This EUNIS classification is the basis for the creation of an European network of Special Areas of Conservation, a major constituent of Natura 2000 (see also Chapter 14). Whereas the EUNIS classification deals with the ‘ideal’ vegetation, the ‘ideal’ situation or reference of a natural salt marsh, from a geomorphological point of view, is shown in Figure 19.5. Although this represents a scheme of an island, it can hold for a mainland marsh when points 1–4 (discussed in this paragraph) are taken into account.

The ‘ideal’ system may not be achievable in many restoration projects. The best option for restoration is to create variation in abiotic conditions with respect to inundation and drainage, and allow the plant and animal species to find their way. However, it is possible that despite proper tidal inundation, plants may not establish as a result of too-low sediment redox potential (Mossmann et al. 2011). Other functions of salt marshes, such as carbon storage and wave attenuation (Table 19.2), will be fulfilled both in a homogeneous and a heterogeneous marsh. There seems to be a contradiction in the fixed classification of EUNIS and the dynamics of the geomorphology. Also here, priority should be given to the dynamic abiotic conditions. Most plant and animal species will establish, as there

Figure 19.5 Geomorphological model of a barrier island with its characteristic main units and traits. Values in parentheses refer to expected longevity of main units (years). (1) Island head with temporal green beaches, (2) old dune and salt marsh systems disconnected from the beach, (3) wash-over complexes and salt marshes with a wide gradient from high elevation with fresh water from the dunes to intertidal flats with saline water, (4) island tail with young dunes and salt marshes connected to the beach and intertidal flats and (5) beach and foreshore with temporal green beaches. The model represents the full dynamics and zonation of salt marshes, and is a reference model for restoration. (Modified from Esselink et al. 2009.)
Restoring salt marshes requires less intensive management to get somewhere near the targets and is essentially about reinstating dynamic physical processes. Because of the additional functions that salt marshes provide, however—such as coastal defence and carbon storage—above more general biodiversity provisioning, the importance of successful restoration is high.

Although there are many similarities in approaches for restoration applied in various parts of the world, some striking different accents can be recognized. In North America, examples are given of removal of fill, amendment of the soil, sowing target species and use of herbicides to remove dominant *Phragmites australis* or *Typha angustifolia*. Sometimes, this happens at a large scale. However, a plea is made for restoration through an adaptive approach by subdividing the site into modules to be restored in phases. The most urgent question should be matched to the first module. An early question might be, which species need to be planted and in what assemblages? A second question could be, how should the soil be amended or the topography be manipulated to achieve the project goals (Callaway & Zedler 2009)? Mitigation projects, including raising a site and planting *Spartina* spp., should not be considered restoration, as there is no balance between tides, elevation, drainage pattern, substrate type and vegetation, hence it is not a gradually self-organizing system. In Europe, de-embankment and increase of tidal amplitude are mainly practised without further assistance for the development of a self-organizing marsh. The same holds for changes in livestock grazing. The latter is the option to be used in South America and Australasia.

### 19.4.2 De-embankments and other measures to repair geomorphological conditions

Embarkments interrupt not only salinity gradients but also sediment deposition. It is obvious that continuous rise in net surface elevation occurs on the unembanked marsh in front of the newly created polder after embankment. Differences in soil level in front of and behind the seawall or summer dike will be greater when the polder is intensively drained for agricultural purposes. This will also hold for coastal systems with accumulation of peat where great shrinkage can take place (Roman et al. 1995). For the sake of coastal protection and the costs of seawall maintenance, it is assumed that a well-inundated tidal marsh with a good rate of sedimentation in front of the seawall or summer dike is better than a low-lying polder without sedimentation. Coastal defence and nature conservation might be combined by de-embankment of polders and subsequent restoration of these former tidal marshes described as ‘managed retreat’ or ‘managed realignment’ (Boorman 1999). In such cases, a new seawall is necessary inland of the present coastal defence that will be knocked down or breached.

After de-embankment of a summer polder, renewed contact with the sea results in rapid re-establishment of abiotic conditions (Erchinger et al. 1994). A restoration is also expected to be quickly successful for birds during high tides, as they have few dispersal problems. It can take a long time before a site is appropriate for foraging on the proper type of food, or breeding in the proper vegetation structure. However, there might be dispersal constraints for plants. Are tidal plants still available in the community species pool as persistent seeds in the soil seed bank as a historic record of the former marsh vegetation? A study in natural salt marshes indicated that most salt marsh species have a transient or short-term persistent seed bank (Wolters & Bakker 2002). This suggests that restoration cannot rely on a persistent seed bank of salt marsh species. Percentages of target species, as related to the regional species pool, established in 70 de-embanked sites in north-western Europe, may amount to 70%, but most sites show lower figures (Figure 19.6: Wolters et al. 2005). The reason is often the constricted elevational zone for all possible salt marsh communities. Nearly 100% of the newly established species originated from the adjacent salt marsh in the de-embanked sites. Apparently, dispersal of diaspores to de-embanked sites may not pose a problem.

On barrier islands in the Wadden Sea, the restoration of dynamics by (partly) removing artificial sand dikes (see Plate 19.1) is in the phase of planning. Along the mainland coast of the Wadden Sea, it is hoped that the intensive ditching pattern in seminatural marshes will transform into natural drainage patterns with current rates of sedimentation. However, an experiment with excavation of 1.5 m of clay for seawall
were revealed to be most successful with respect to the re-establishment of halophytic plant species after de-embankment were all grazed (Wolters et al. 2005). This can be attributed to breaking up monospecific swards and reducing competition. In fact, these restored marshes ‘behave’ as aforementioned existing salt marshes.

**19.4.3 Increase of tidal amplitude**

In Connecticut, United States, tidal flows were reinstituted by placement of a 1.5 m diameter culvert in an impoundment. Restoration targets were expressed in terms of inundation, salinity, productivity and community structure. Vegetation integrates a number of factors; however, food chain support as manifested by macro-invertebrates, fish and birds was also taken into account. In 1988, 10 years after the start of the restoration, the existing freshwater species *Typha angustifolia* had declined from 75% to 15%, whereas *Spartina alterniflora* had increased from <1% to 45%. In addition, high-marsh species had re-established and covered 20% of the site, but *Phragmites australis* had also spread. After 1988 *Phragmites* declined and salt marsh vegetation had increased to cover 85%. Although the restoration exhibits a striking result, the restored marsh only moderately resembles the pre-impoundment marsh (Fell et al. 2000). Monitoring between 5 and 20 years of nine sites along the coast of Connecticut revealed different results for various parameters. Rapid recovery of vegetation was related to sites with low elevation, greater inundation and higher groundwater tables as a result of greater colonization potential. Recovery of other organisms was not always related to that of the vegetation. A viable population of the native, high-marsh snail *Melanopsis bidentatus* took 20 years to recover in a site where the vegetation recovered much faster. Characteristic fish species assemblages were found in creeks and ditches within 5 years. Populations of breeding birds typical of salt marshes had established after 15 years in the restored sites (Warren et al. 2002). Hence, within 20 years, successful restoration by the increase of tidal amplitude is possible in this site with these conditions. However, possibilities and time scales for other salt marsh functions, such as carbon storage and wave attenuation, are not known.

The polder Beltrigharder Koog, Germany, was embanked in 1987 in order to have a reservoir to

---

**Figure 19.6** Frequency distribution of scores of percentage of characteristic salt-marsh plant species related to the regional species pool of over 70 de-embanked sites in north-west Europe. Note the absence of monitoring in about 50% of the sites. (Modified from Wolters et al. 2005.)
control drainage of the hinterland. It consisted of intensively grazed salt marsh and intertidal flats. The tidal range was reduced from 3.4 m to zero immediately after embankment. Former salt marshes were colonized by tall fresh forb communities and former intertidal flats by salt marsh species. In order to compensate for the losses, a lagoon system was established allowing reduced tidal influence through two sluices. From 1990 onwards a 0.3 m tidal amplitude was established with stormflood simulations of 0.8 m tidal amplitude twice a month in the non-breeding period of birds. The restoration was successful in the lower inundated parts where a typical salt marsh zonation established. However, the highest parts remained fresh grassland, and started to get overgrown by shrub species, in the absence of livestock grazing (Wolfram et al. 2000).

19.4.4 Changes in the grazing regime

Cessation of livestock grazing can result in a decrease of the number of plant and animal species, as discussed above. In an experiment of resuming livestock grazing in a formerly grazed back barrier salt marsh in the Wadden Sea, the changes in the vegetation after abandonment turned out to be reversible within 10 years at a relatively high stocking rate of cattle (Bakker 1989). At the larger scale of the entire elevational gradient, the number of plant communities increased, whereas at the small scale of each elevational level, the number of plant species per unit area increased, mainly because of the removal of the tall grass *Phragmites australis* in saline marshes and of *Elytrigia atherica* in brackish and fresh marshes (Bakker 1989; Bakker et al. 2003). However, at the low marsh the number of species decreased. The soft, frequently inundated substrate is trampled by cattle and only annual pioneer species can be maintained. It is estimated that the numbers of brent geese in the entire Wadden Sea in May can be a factor of 4–8 higher if all salt marshes were to be grazed by livestock, than in the absence of grazing (Bos 2002).

19.5 PERSPECTIVES

Targets for the successful restoration of salt marshes include changes in the geomorphology and succession of plant communities. Squeezing of salt marshes between seawalls or artificial sand dikes in the hinterland and intertidal flats can be prevented by realigning landward. This should not be carried out by the extension of new sedimentation fields into the intertidal flats, because this would imply a reduction of the area of intertidal flats with their characteristic conservation issues such as waders. Along the mainland coast of Europe, seawalls and summer dikes have successfully been removed and new salt marshes have developed in former polder areas. On barrier islands in the Netherlands, removal of artificial sand dikes is planned, but experiments are needed. Increase of the tidal amplitude in embanked polders through sluices is not always successful, and especially the higher parts of the elevational gradient may be lost for salt marsh development when only part of the tidal amplitude can be reinstated. Intensive ditching patterns in salt marshes with sedimentation fields or mosquito control do not transform into meandering creeks. Consequently, reduced abiotic variation may establish, and hence habitats for plant and animal communities.

Increased sea level rise caused by global warming might have a drowning effect on salt marshes. However, the salt marsh elevation may follow the level of Mean High Tide without negative consequences for increased sea level rise. Is current vertical accretion enough for salt marshes to keep their relative position in the tidal frame? The current sea level rise in western Europe is about 3 mm yr⁻¹. Recent modelling studies suggest the overwhelming importance of tidal amplitude and concentrations of suspended sediment (Kirwan et al. 2010). At future sea level rise of about 10 mm yr⁻¹ salt marshes in regions with a tidal range greater than 1 m and suspended sediment concentration greater than 20 mg L⁻¹ will not be permanently submerged. When future rates of sea level rise beyond 20 mm yr⁻¹ occur, only salt marshes in regions with tidal ranges over 3 m and sediment concentrations above 30 mg L⁻¹ are expected to keep pace with future sea level rise. In the latter case, many marshes will permanently submerge despite their tendency to accrete more quickly (Kirwan et al. 2010). Knowledge on vertical accretion rates at the scale of catchment areas (creeks with their drainage area) in salt marshes is poorly developed. Back barrier marshes with a low vertical accretion might suffer earlier than mainland marshes with a higher vertical accretion.

Mangrove forests occur lower in the zonation than salt marshes. Sea level rise will increase the inundation frequency, reduce soil salinity and increase transport...
There are more general trade-offs between functions of salt marshes, and hence targets for restoration. Long-term monitoring on different spatial scales in restoration sites (Long-Term Ecological Research sites) of changes in geomorphology, hydrology, vertical accretion, plants, fish, birds, mammals and invertebrates, in a foodweb context, will be helpful as decision tools for future projects. An important issue is to learn at what time scale the different elements of the ecosystem can be restored.

In parts of the world with intensive grazing such as South America and Australasia, experiments with removal of cattle are needed to assess their impact on salt marshes. In north-western Europe, abandonment of livestock grazing and mowing of salt marshes has already taken place with negative results for diversity of characteristic plant and animal species. Recovery of mono-species stands (resulting from abandonment from agricultural exploitation) is successful after reintroducing livestock grazing. Experiments are needed with respect to stocking density and rotational grazing for the benefit of various groups of organisms in their food web context such as plants, invertebrates and breeding and migrating birds. In general, the scientific basis for management based on experiments is largely missing.

It is not clear how to deal with the trade-offs between vertical accretion and biodiversity issues. Highest vertical accretion can be accomplished on salt marshes with tall and dense canopy, hence without livestock grazing. However, the diversity of plant and animal communities including characteristic halobiontic organisms is better preserved or restored in grazed marshes with lower vertical accretion than in abandoned salt marshes. Highest diversity will be achieved with stocking densities that create patterns of short grazing lawns alternating with patches of tall vegetation. Particularly long-term ungrazed marshes have low diversity of plant and animal communities. Short-term abandonment may result in flowering plants attracting pollinating insects, breeding birds foraging on them, and plants setting seed and attracting granivorous birds.

There are more general trade-offs between functions of salt marshes, and hence targets for restoration. Long-term monitoring on different spatial scales in restoration sites (Long-Term Ecological Research sites) of changes in geomorphology, hydrology, vertical accretion, plants, fish, birds, mammals and invertebrates, in a foodweb context, will be helpful as decision tools for future projects. An important issue is to learn at what time scale the different elements of the ecosystem can be restored.

In parts of the world with intensive grazing such as South America and Australasia, experiments with removal of cattle are needed to assess their impact on salt marshes. In north-western Europe, abandonment of livestock grazing and mowing of salt marshes has already taken place with negative results for diversity of characteristic plant and animal species. Recovery of mono-species stands (resulting from abandonment from agricultural exploitation) is successful after reintroducing livestock grazing. Experiments are needed with respect to stocking density and rotational grazing for the benefit of various groups of organisms in their food web context such as plants, invertebrates and breeding and migrating birds. In general, the scientific basis for management based on experiments is largely missing.

It is not clear how to deal with the trade-offs between vertical accretion and biodiversity issues. Highest vertical accretion can be accomplished on salt marshes with tall and dense canopy, hence without livestock grazing. However, the diversity of plant and animal communities including characteristic halobiontic organisms is better preserved or restored in grazed marshes with lower vertical accretion than in abandoned salt marshes. Highest diversity will be achieved with stocking densities that create patterns of short grazing lawns alternating with patches of tall vegetation. Particularly long-term ungrazed marshes have low diversity of plant and animal communities. Short-term abandonment may result in flowering plants attracting pollinating insects, breeding birds foraging on them, and plants setting seed and attracting granivorous birds.