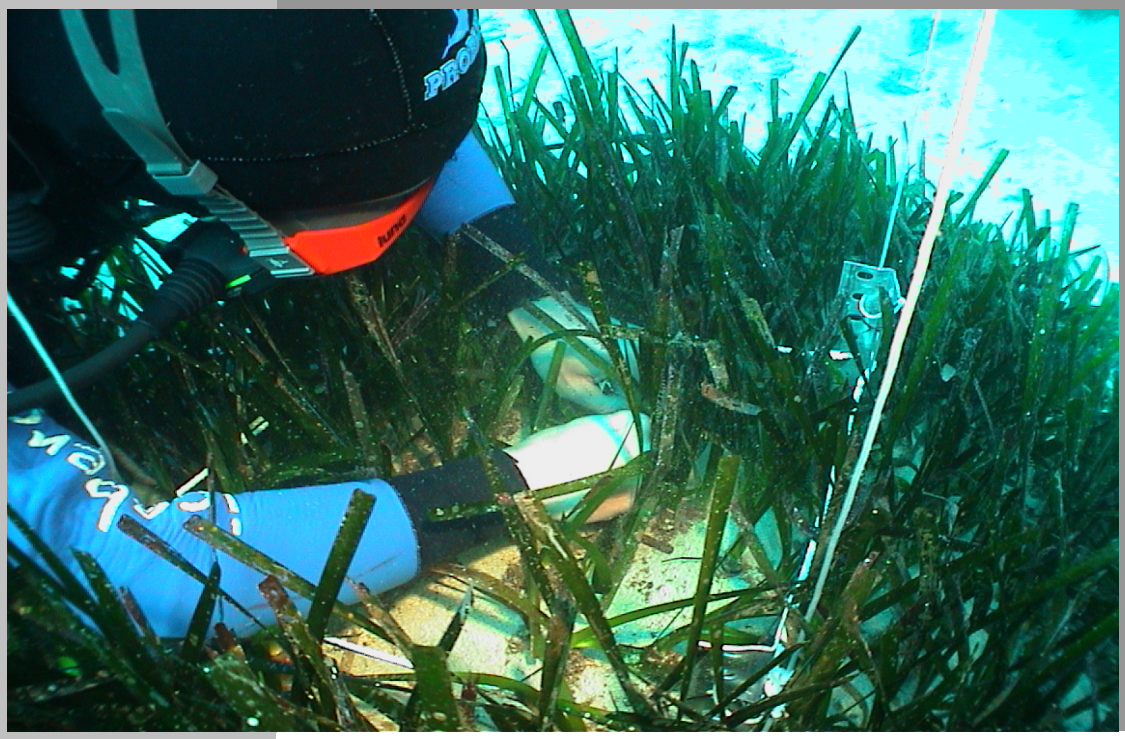

European seagrasses: an introduction to monitoring and management

A publication by the EU project Monitoring and Managing of European Seagrasses (M&MS) EVK3-CT-2000-00044

Edited by

Jens Borum, Carlos M. Duarte, Dorte Krause-Jensen and Tina M. Greve

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Preface

By Jens Borum (FBL), Carlos M. Duarte (IMEDEA), Dorte Krause-Jensen (NERI) and Tina M. Greve (FBL)

Seagrasses are rooted, flowering plants that grow in the marine environment with great success. Seagrasses form dense and highly productive beds of great importance to invertebrates, fish and many birds, and the beds provide protection against coastal erosion. The four European seagrass species grow from the intertidal and down to 5-15 meter depth in North European waters (Zostera marina, Z. noltii), but seagrasses may be found even deeper than 50 meter in clear Mediterranean waters (Cymodocea nodosa and Posidonia oceanica). In the Mediterranean Sea, P. oceanica beds cover between 25,000 and 50,000 km^2 of the coastal areas corresponding to 25% of the sea bottom at depths between 0 and 40 m. Hence, seagrasses are extremely important components of coastal European waters.

Seagrass beds are, however, also vulnerable and have declined substantially in many coastal areas due to increasing human pressure in the form of nutrient loading, siltation and mechanical disturbance. Several countries within the European Union have national legislation and regulations to protect seagrass beds from anthropogenic disturbance, but with the Water Framework Directive the Member States have established a mutual platform and obligation to ensure a “good ecological status” corresponding to conditions with minimal anthropogenic impact of all surface waters. Extended seagrass beds with a good penetration to deep waters are characteristic of coastal waters with minimal anthropogenic impact. Since seagrasses are mostly perennial organisms, they reflect the temporally integrated environmental conditions, and, therefore, seagrasses are excellent indicator organisms on which environmental monitoring and management of coastal waters can focus. Hence, there is a high need for efficient monitoring and management strategies for the European seagrass species, and we hope that the present booklet will strengthen the focus on these unique coastal organisms.

With this booklet, we aim to give environmental managers a basic introduction to monitoring and management of European seagrasses. Several countries already have established comprehensive and advanced monitoring programmes for seagrasses, but in many European countries programmes are virtually absent. We here, primarily, target the latter group by compiling and presenting basic information on what seagrasses are, what their importance is, the factors controlling their performance, the threats against them and the temporal scales for seagrass recovery. Next, we introduce the reader to basic monitoring strategies and parameters, and, finally, we present recommendations as to how seagrass beds can be protected and recovered through environmental management. If coastal managers on this basis decide to initiate monitoring and management programmes for seagrasses, they will need more detailed information than provided by this booklet. Hence, we have listed links and references to existing seagrass monitoring programmes, manuals on seagrass monitoring and a number of relevant scientific papers.

The booklet is an outcome of a joint EC-funded project (EVK3-CT-2000-00044) entitled “Monitoring and Management of European Seagrass Beds

(M&MS)" under the 5th Framework Programme. The partners involved in the project and contributing to this booklet are:

Freshwater Biological Laboratory, University of Copenhagen, Denmark (FBL)

Dept. of Marine Ecology, National Environmental Research Institute, Denmark (NERI)

School of Ocean Sciences, University of Wales, United Kingdom (SOS)

Centro de Ciências do Mar, Universidade do Algarve, Portugal (CCMAR)

Parque Natural da Ria Formosa, Instituto da Conservação da Natureza, Portugal (PNRF)

Instituto Mediterraneo de Estudios Avanzados, Consejo Superior de Investigaciones Cientificas, Spain (IMEDEA)

Direcció General de Pesca, Govern de Les Illes Balears, Spain (DGP)

The objective of the project has been to conduct research on European seagrasses with respect to their dependence on water and sediment quality, their capacity for spreading and expansion through vegetative propagation and sexual reproduction, and their response to environmental conditions as reflected by genetic diversity and isotopic composition. The results are - and will be - published in scientific journals but they are also, combined with knowledge from the literature, the foundation for this booklet. More information about the project is available on the project home page (www.seagrasses.org) from where the booklet can be downloaded in pdf format.

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We are indepted to Jan E. Vermaat, Jens S. Laursen, Sif Johansson, Nuno Grade and Elvira A. Pérez for their contributions as external and internal reviewers providing constructive critique and comments on the structure of the book and on drafts of chapters.

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1 The four European seagrass species

Seagrasses are flowering plants with terrestrial ancestors. Of about 60 species world wide, only four seagrass species are native of European waters. In spite of their terrestrial origin, seagrasses are well adapted to the marine environment and can in Europe be found from the intertidal zone at the shore to depths down to 50-60 m. The European species are easy to identify and their geographical distribution range is well known.

By Jens Borum and Tina M. Greve (FBL)

There are four European species of seagrasses:

- *Zostera marina* (eelgrass)
- *Zostera noltii* (dwarf eelgrass)
- *Cymodocea nodosa*
- *Posidonia oceanica*

Seagrasses have evolved from different groups of freshwater plants, and some seagrass species are more closely related to freshwater plants than to other seagrasses. However the four European species are closely related. Other species of water plants occur in marine areas of low to moderate salinity but only the group defined as seagrasses can be found in oceanic waters with high salinity.

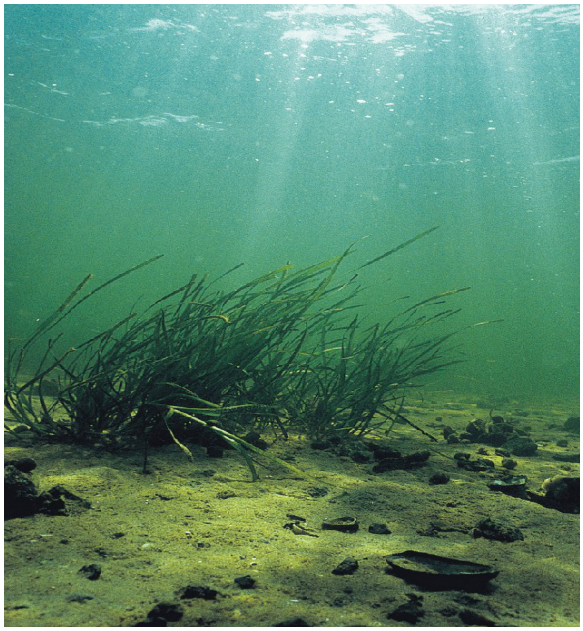
Seagrasses grow and reproduce sexually being continuously submerged under water. They do not require contact with air, and the reproductive cycle with flowering and pollination is completed under water. Seagrasses have a number of adaptations to the submerged life form. They rather efficiently take up inorganic carbon from the water, and the nutrients required for growth can be taken up by the roots, as for terrestrial plants, or from the water column through the leaves. Since the sediment comprising the sea floor is most often without oxygen, the underground parts are supplied with oxygen from the leaves through a system of air-filled channels within the tissue. The spreading of pollen to female flowers, which is typically mediated by wind or insects in terrestrial plants, occur by water currents.

Seagrasses may look quite different but the European species have several characteristics in common. The above-ground, visible part of seagrasses consists of shoots or leaf bundles with 3 to 10 linear leaves. The shoots are attached to rhizomes (vertical and/or horizontal) creeping within or on top of the sediment from which roots penetrate into deeper layers of the sea floor. The rhizomes divide and form new leaf bundles, and each branched rhizome system can hold many genetically identical shoots, which are then interconnected as one individual like in other clonal plants.

Zostera marina

Zostera marina (eelgrass; Figure 1.1) is found from arctic waters along the northern Norwegian coast, where it can survive several months of ice cover, to the Mediterranean (Figure 1.2). The species is very abundant in the Baltic Sea, the North Sea and along the Atlantic coasts down to northern Spain. Further south, *Z. marina* becomes more rare and in the Mediterranean the species is mostly found as small isolated stands, but dense eelgrass beds do occur, especially, in lagoons. *Z. marina* is predominantly subtidal and may grow down to 10-15 meters depth depending on water clarity. *Z. marina* is most often perennial but annual stands are found intertidally in the Wadden Sea.

The shoots of *Zostera marina* have 3 to 7 leaves. Leaf width varies between 2 mm for young plants and up to 10 mm for large individuals. The leaves are usually 30 to 60 cm long but may be up to 1.5 m in beds on soft sediments at intermediate depths.



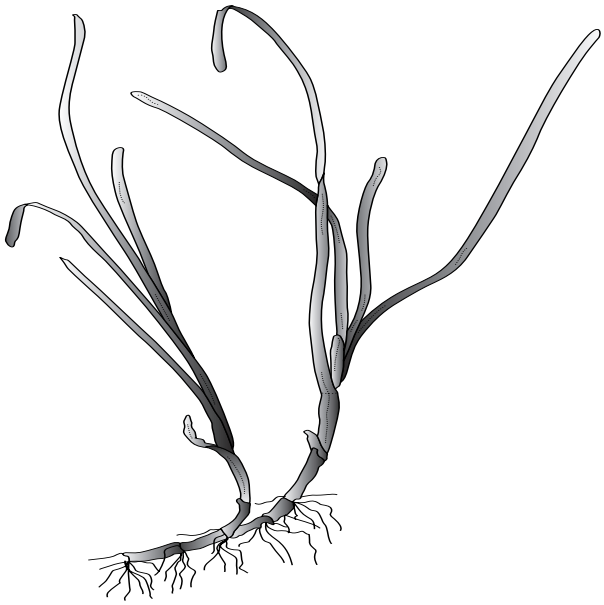


Figure 1.1. *Zostera marina* (eelgrass) forms dense stands from the intertidal zone to depths of 10-15 meters in areas with clear water. The species is easily identified by the terminal shoots on only horizontal rhizomes. Photo: P.B. Christensen; drawing: redrawn from Dawes 1981.

The leaf bundles are terminal shoots on horizontal rhizomes. The rhizome branches during the growth season forming new terminal shoots. For each new leaf produced a new rhizome segment

(internode) is formed and two bundles of roots develop from the nodes between the segments. The roots are thin (0.2-1 mm), covered by fine root hairs and may be up to 20 cm long. The

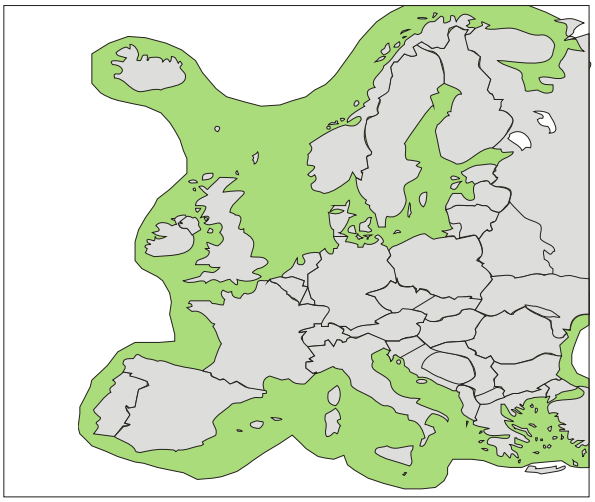


Figure 1.2. Geographical distribution of *Zostera marina* (eelgrass) in European coastal waters. Eelgrass is found from arctic waters along the northern Norwegian coasts to the Mediterranean, where it is more sparse and only forms dense and extensive beds in some lagoons. Eelgrass is the only seagrass species growing around Iceland.

rhizome segments are 2-6 mm thick, the length of each segment varies from 5 to 40 mm and the

colour changes from white-green in newly formed segments to dark brown in old segments.

The male and female flowers of *Zostera marina* are small, grenish and partly hidden in pockets within the leaf sheaths. Male and female flowers are found on the same individual. *Zostera marina* flowers frequently and may produce several thousand seeds per square meter. Flowering can be observed from early spring to fall. During flowering the shoots change morphology to produce more leaf bundles separated by long, thin stem segments. The seeds are 2-4 mm long and, when fully developed, the flowering shoots detach and float away from the bed. The seeds either drop to the sediment within the bed or are dispersed along with the floating shoots (rafting). Seeds are probably also spread by ducks and geese feeding on eelgrass stands.

Zostera noltii

Zostera noltii (dwarf eelgrass; Figure 1.3) is distributed from the southern coasts of Norway to the Mediterranean Sea, the Black Sea, the Canary Islands and has been recorded as far south as on the Mauretanean coast (Figure 1.4). *Z. noltii* forms dense beds in the muddy sand of intertidal areas, where *Zostera marina* is sparse due to its lower tolerance to dessication. The

narrow, flexible leaves of *Z. noltii* often become fully exposed to the air but are kept moist by the thin film of water on muddy sediments. *Z. noltii* also occurs subtidal but often seems to be outcompeted by other seagrasses where the water cover is permanent.

Zostera noltii has small leaf bundles with 2 to 5 narrow leaves attached to a horizontal rhizome. Each rhizome holds many shoots on short branches separated by rhizome segments. The leaves are 0.5-2 mm wide and 5 to 25 cm long. The rhizomes are 0.5 to 2 mm thick and the rhizome segments are from 5 to 35 mm long. The most recently formed internodes are light green while older segments turn yellow or brown. *Z. noltii* have 1-4 thin (< 1 mm) roots attached to each node between the rhizome segments.

The male and female flowers of *Z. noltii* are small and found on the same individual. The seeds are 1½-2 mm long. The seeds are probably spread by ducks and geese feeding on the intertidal beds.

Figure 1.3. *Zostera noltii* (dwarf eelgrass) forms dense stands within the intertidal zone, where other seagrass species are excluded. The species is best identified by the many small shoots with narrow leaves attached by short branches to the horizontal rhizome. Photo: J. Borum; drawing: redrawn from NN.

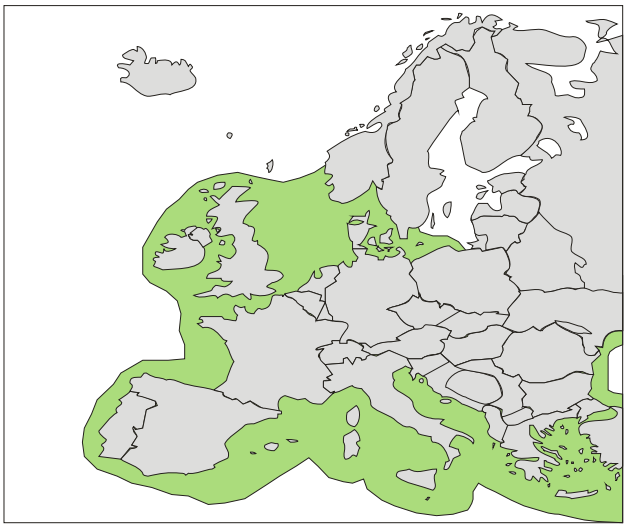


Figure 1.4. Geographical distribution of *Zostera noltii* (dwarf eelgrass) in European coastal waters. Dwarf eelgrass is found from the southern coast of Norway to the Mediterranean and even as far south as the Mauretanean coast.

Figure 1.5. *Cymodocea nodosa* (seahorse grass) occurs from the intertidal zone to large depths of even 50-60 m in areas with clear water. The species is best identified by its vertical rhizomes and the long, white or pink segments of the horizontal rhizomes. Photo: O. Pedersen; drawing: redrawn from Luque and Templado 2004.

Cymodocea nodosa

Cymodocea nodosa (Figure 1.5) has no vernacular English name, but we propose to use 'seahorse grass' as an appropriate name because beds of *Cymodocea nodosa* are characteristic habitats for seahorses. *C. nodosa* is a warm water species and is widely distributed throughout the Mediterranean, around the Canary Islands and down the North African coast (Figure 1.6). The species does not extend further north than the southern coasts of Portugal. *C. nodosa* can be found from shallow subtidal areas to very deep waters (50-60 m).

Cymodocea nodosa has leaf bundles consisting of 2 to 5 leaves. The leaves are 2 to 4 mm wide and from 10 to 45 cm long. The leaves resemble those of medium sized *Zostera marina*. However, the shoots are attached to vertical rhizomes with short rhizome segments which again are attached to a horizontal rhizome with 1-6 cm long segments. The apex forms vertical rhizomes and branches to new horizontal rhizomes. The rhizome may grow several meters per year, and *C. nodosa* is considered a pioneer species which can quickly colonize bare areas of the sea floor. *C. nodosa* can easily be identified by its vertical rhizomes and the long white to pink horizontal rhizome segments. The roots are dispersed along the vertical and horizontal rhizomes. Each rhizome segment only has one root which is often strongly branched and may be up to 3 mm thick and up to 35 cm long.

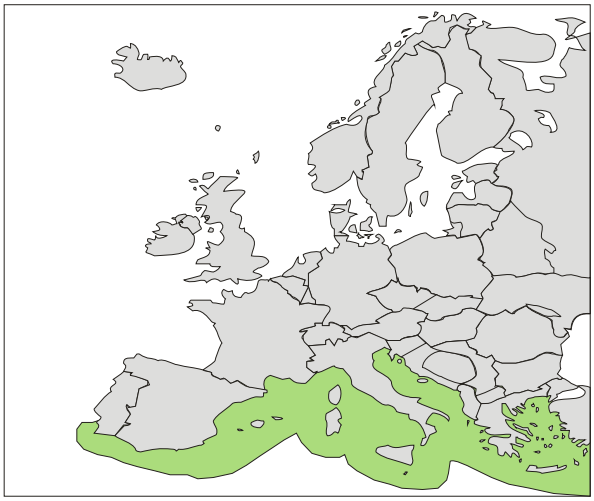


Figure 1.6. Geographical distribution of Cymodocea nodosa (seahorse grass) in European coastal waters. Seahorse grass is a South European species found in the Mediterranean and along the Atlantic coasts from southern Portugal to Mauretania.

The individuals are either male or female plants. The female flowers have two ovaries and the two lentil-shaped seeds produced from each flower are around 8 mm long and, hence, considerably larger than the seeds of the Zostera species.

Flowering is observed between May and August but is in general rare.

Posidonia oceanica

Posidonia oceanica (Figure 1.7) is restricted to the Mediterranean Sea and its distribution stops at the boarder line where Mediterranean and Atlantic waters mix in the western part of the Mediterranean Sea (Figure 1.8). *P. oceanica* grows from shallow subtidal waters to 50-60 m depth in areas with very clear waters. In contrast to the other European seagrasses, *P. oceanica* can form reefs of slowly accumulating, compacted sediments. In spite of its very slow growth, *P. oceanica* is the most wide-spread higher plant in the Mediterranean, and beach cast up of *Posidonia* leaves can be found in large amounts.

Posidonia oceanica has leaf bundles consisting of 5 to 10 leaves attached to vertical rhizomes. The leaves are broad (5 to 12 mm) and the length usually varies from 20 to 40 cm but may be up to 1 m. The vertical rhizomes are, as for *C. nodosa*, attached to horizontal rhizomes which branch and expand by terminal apices. The rhizome internodes are short (0.5 to 2 mm) reflecting the slow horizontal growth of the plant, and the thickness of the rhizomes vary between 5 and 10 mm. The roots are 3-4 mm thick, up to 40 cm long and richly branched. The rhizomes of *P. oceanica* can easily be distinguished from those of the other three species, among other things by the dense, hairy remains of old, degrading leaf sheaths around the rhizomes. The hairy remains are also found as conspicuous balls of fibers washed ashore on the beaches.

Posidonia oceanica rarely flowers. Usually, less than 1 flower is produced per 10 square meters per year, but flowering may be more frequent during warm years. The flowers are large and produce a large fruit (10 mm). Young individuals originating from seeds (seedlings) are rarely found and spreading of *P. oceanica* primarily occurs vegetatively by branching of the rhizomes.

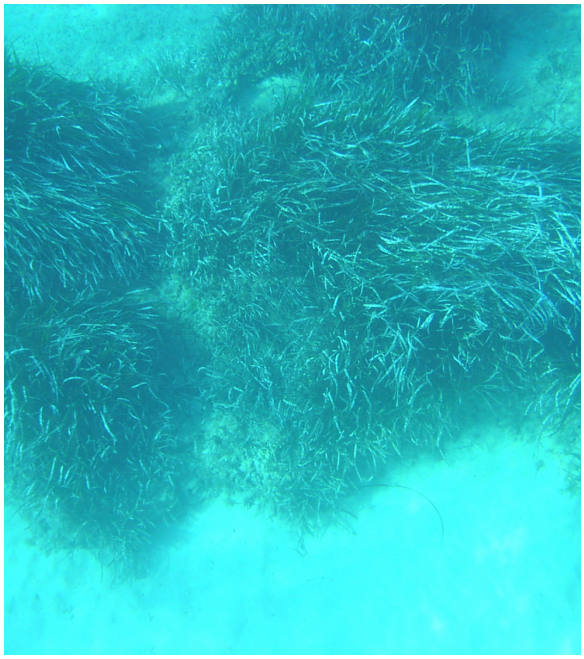


Figure 1.7. *Posidonia oceanica* forms very dense stands from the subtidal to depths down to 50-60 m in areas with clear water. The species is easily identified by the dense, broad leaves and the hairy remains around the rhizomes and lower parts of the shoots. Photo: P.B. Christensen; drawing: redrawn from Luque and Templado 2004.

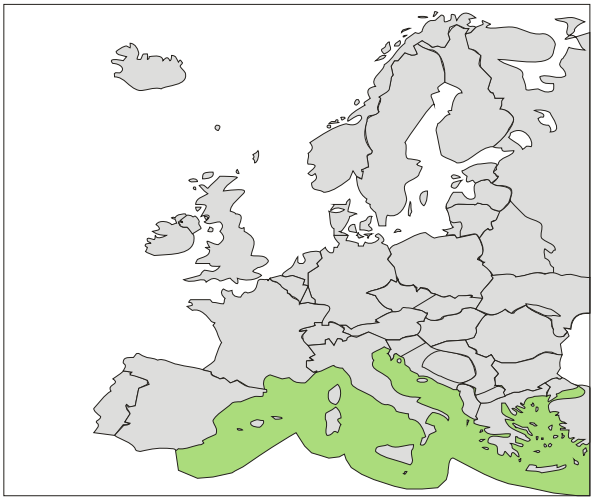


Figure 1.8. Geographical distribution of *Posidonia oceanica* in European coastal waters. *Posidonia oceanica* is strictly Mediterranean and its distribution stops in the western part of the Mediterranean where the warm Mediterranean water mixes with cold Atlantic water.

Confusion of species

The four seagrass species are easily distinguished but confusion may occur under some circumstances. Young specimens of *Zostera marina* may be confused with *Z. noltii* in the transition zone between the intertidal and the subtidal area. However, the leaves of *Z. noltii* are more narrow than those of *Z. marina*. Where *Z. marina* and *Cymodocea nodosa* occur in mixed stands, their leaves may be difficult to distinguish, but *C. nodosa* is easily identified by the long white to pink horizontal rhizome segments and the vertical rhizomes with compressed segments.

Zostera noltii may grow in the same intertidal habitats as species of the genus *Ruppia*. Although *Ruppia* species also have narrow leaves, they can be identified by their long white rhizome segments. Each node forms a leaf bundle and one long root. Finally, *Z. marina* may be confused with young specimens of the freshwater plant, *Potamogeton pectinatus* that grows in brackish water. However, the rhizome of *P. pectinatus* has two segments between each shoot and root bundle.

Invading seagrasses

Other species of seagrasses may successfully invade European waters if seeds or fragments are accidentally introduced. Currently, the warm water seagrass species *Halophila stipulacea* is invading the eastern Mediterranean and has reached the southern coasts of Italy. The species was introduced through the Suez Channel. *Halophila stipulacea* is a small species with 3-6 cm oblong leaves looking very different from the linear leaves of the native European seagrass species.

Box 1.1 There are four European species of seagrasses:

- *Zostera marina* (eelgrass) grows from the Arctic to the Mediterranean Sea and is the only seagrass species found along the coast of Iceland.
- *Zostera noltii* (dwarf eelgrass) is widely distributed along the Mediterranean and atlantic coasts but does not extend further north than to the southern coasts of Norway.
- *Cymodocea nodosa* (seahorse grass) is found in the Mediterranean Sea and in the warmer regions of the Atlantic Sea from southern Portugal to the northwestern African coast.
- *Posidonia oceanica* is a strictly Mediterranean species.
- Sometimes *Zostera marina* is separated in two species, *Z. marina* and *Z. angustifolia*, but the species characteristics are not clear.
- In older literature *Zostera noltii* is refered to as *Zostera nana*.
- Some consider the genus *Ruppia* to belong to the group of seagrasses, but according to den Hartog (1970) the species within this genus are not true seagrasses because they do not occur in oceanic water with consistently high salinity.

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2. Why are seagrasses important? - Goods and services provided by seagrass meadows

Seagrass meadows produce a variety of goods (finfish and shellfish) and provide ecological services (maintenance of marine biodiversity, regulation of the quality of coastal waters, protection of the coast line) which are directly used or beneficial to humans and condition the economic development of European coastal zones. In addition seagrasses are indicators of the status of the coastal zone which can be used in coastal management strategies aiming at preserving or improving the environmental quality of the coastal zone.

By Jorge Terrados (IMEDEA) and Jens Borum (FBL)

Although seagrasses might not be widely known by the public nowadays, they were known and used for different purposes by coastal communities in the past. The leaves of *Posidonia oceanica* were traditionally used as packing material to transport fragile items (i.e., glassware, pottery) in Mediterranean countries. They were also used to ship fresh fish from the coast to cities. As parasites thrived less in *P. oceanica* leaves than in straw, they were used as cattle bedding in stables and, later, as filling material for mattresses and cushions (Pope Julius III popularized this practice throughout Italy in the 16th century). Respiratory infections seemed to be prevented when sleeping in this type of bedding; other medicinal uses included the alleviation of skin diseases (i.e. acne) and pain in legs caused by varicose veins. When straw was scarce dry *P. oceanica* leaves were used to make adobes, and as roof insulation (i.e., in SE Spain and the Balearic islands), and leaves of *Zostera marina* (eelgrass) have been used as roof covering (Fig. 1). In the Netherlands eelgrass leaves have been used as constituents of dikes ("wierdyken"), were the preferable stuffing of baby mattresses until the 1950's, and they are still used in chair seats. Seagrass leaves have been used as soil amendment and to feed pigs, rabbits and hens in several areas.

The large knowledge about the biology and ecology of seagrasses gained during the last third of the 20th century has driven increased awareness of the economic value of seagrasses to humans. The biological resources and ecological services provided by seagrasses are based on the physical structure of the plants themselves and the underwater meadows they form, their biological activity, and that of the associated fauna and flora. The first appraisal of the value of the services provided by seagrass ecosystems produced a minimum estimate of 15837 € ha^{-1} y^{-1}, which is two orders of magnitude higher than the estimate obtained for croplands. Even if these estimates have limitations and caveats, they highlight the importance of seagrass ecosystems.

Seagrasses as promoters of biological productivity and biodiversity

Seagrasses provide habitat for a large set of organisms which can not live in unvegetated bottoms. The leaf canopy and the network of rhizomes and roots provide substratum for attachment, which is scarce in unconsolidated bottoms, stabilize the sediment, and reduce irradiance producing an array of microhabitats not present in unvegetated bottoms. In addition, the three-dimensional structure of seagrasses creates hiding places to avoid predation. As a result, the abundance and diversity of the fauna and flora living in seagrass meadows are consistently higher than those of adjacent unvegetated areas. Seagrasses, therefore, increase habitat diversity and the biodiversity of the coastal zone.

Seagrasses feature high rates of primary production. As any other photosynthetic organism, seagrasses fix carbon dioxide using the energy provided by light and transform it into organic

carbon to sustain seagrass growth and biomass production. High rates of biomass production imply high rates of oxygen production, a by-product of photosynthesis, which is released to the surrounding waters. The biomass of some seagrasses decomposes slowly and certain species (i.e. *Posidonia oceanica*) store a significant amount of carbon in the sediment over long periods. Seagrass primary production is only 1% of total primary production in the oceans but seagrasses are responsible for 12% of the total amount of carbon stored in ocean sediments. This uncoupling between carbon dioxide fixation by photosynthesis and release by respiration determines that seagrasses play a significant role in the regulation of the global carbon cycle.

Figure 2.1. Seagrass leaves have for centuries been used as soil amendment, cattle feed and as filling and building material. Here leaves of *Zostera marina* are used for roof covering on a farm house. Photo: J. Borum

The primary production of periphytic algae growing on seagrasses and of benthic algae living in seagrass meadows is comparable to that of the seagrasses themselves. Together with the secondary production of associated fauna, those contribute to make seagrass ecosystems as productive as many agricultural crops and forests on land. The coastal zone is a dynamic environment and currents and waves detach part of seagrass biomass and transport it to adjacent marine and terrestrial ecosystems. These inputs of organic matter may locally be quite high (i.e., to the point that they sustained direct exploitation by humans as described previously), and contribute significantly to the function of biological communities of adjacent habitats (such as beach fauna).

Seagrass meadows are key habitats in the life cycle of many organisms. The populations of crustaceans (e.g. shrimps) and fishes living in seagrass meadows are typically composed of a high proportion of larvae and juvenile individuals suggesting that seagrass meadows are preferred nursery habitats. Increased food availability and/or refuge from predation explains the importance of seagrass meadows as nursery and feeding habitats for these organisms, some of them target of highly important commercial fisheries. In addition, migrating birds use shallow and intertidal seagrass meadows as resting and feeding areas during their travels. Brent geese, wigeons and pintails feed preferentially on seagrasses, other birds feed on associated fauna.

Seagrasses as filters improving water quality

Seagrass leaf canopies dampen water movement and favor the retention of suspended particles, both living and dead, becoming a sort of a filter for coastal waters.

The particle trapping capacity of seagrasses is enhanced by the organisms living on the leaves either through filter feeding and active capture, or through the direct attachment of the suspended particles to the mucus-covered seagrass surfaces which result from their activity. As a result seagrasses can to a certain extent control the transparency of the water column. Increased light availability at the bottom facilitates the life of seagrasses themselves and that of other benthic plants which will further increase the control of water transparency.

Seagrasses and associated algae are able to absorb inorganic nutrients through both roots and leaves. The acquisition of nutrients from the water column allows seagrasses tocompete with phytoplankton for the inorganic nutrients that support the primary production of coastal ecosystems. Lower phytoplankton abundance means higher irradiance at the bottom because the phytoplankton cells absorb the light. Seagrass meadows, therefore, can be considered as filters with capacity to control some of the elements (suspended dead particles, phytoplankton cells, nutrients) that determine the quality of coastal waters.

Seagrasses as coastal protection elements

The leaf canopy and the network of rhizomes and roots fix and stabilize the sediment over which seagrasses grow, and reduce the resuspension of the sediment by currents and waves. This role is

driven by reduced water motion due to canopy friction and by the structural frame that rhizomes and roots provide to the sediments. Sediments vegetated by seagrasses are less likely to be mobilized by waves and currents, so that seagrasses can reduce the erosion of the coastline.

Detached seagrass leaves, which are lost either at the end of their life or earlier due to waves and storms, and their accumulation in the beaches represent another way by which seagrasses have a role in the protection of the shoreline. Large accumulations of leaves, such as those of *Posidonia oceanica* in the Mediterranean and eelgrass in northern Europe, dissipate wave energy and directly protect beach sediments from the impact of waves.

Seagrasses are important elements of coastal protection not only because they protect the sediments from being eroded but also because they actually may produce sediment. In the Mediterranean, for example, the particles that constitute the sediment have in many cases a biological origin being fragments of the skeletons, shells or spines of marine animals or being the calcareous remains of benthic algae. As seagrasses harbor a large diversity of marine organisms, the meadows can be considered a net source of new sediment. Biogenic particles can be the main component of sediment in coastlines with no rivers or with low fluxes of particulate matter from land to the sea. In such areas sediment produced by seagrass meadows may contribute significantly to feed the beaches, further contributing to curb coastal erosion.

Conclusions

Seagrass meadows produce a variety of goods (finfish and shellfish, sediment) and provide ecological services (maintenance of biodiversity, water-quality control, shore-line protection) that are directly used or beneficial to humans.The presence and abundance of seagrasses, can be considered, therefore, as indicators of the overall environmental quality of the coastal zone. Hence their long-term maintenance could be a surrogate target of coastal management strategies aiming at preserving or improving the environmental quality of the coastal zone.

In addition to fisheries, urban development, tourism and other recreational activities are significant parts of the economy of European countries with access to the sea, and these activities are highly dependent on the quality of the coastal water and the stability of the coastline, conditions which are strongly supported by healthy, well-developed seagrass meadows. Seagrasses are, therefore, a valuable resource amenable to economic quantification and their conservation should be given high priority in coastal management.

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3 How do seagrasses grow and spread?

Seagrass colonisation and meadow maintenance proceeds by patch stablishement, from seed germination and plant fragments, and clonal growth. Knowledge on seagrass growth rates and success of reproductive effort is crucial to manage seagrass ecosystems, particularly to derive expectations on the recolonisation times required to recover seagrass meadows.

By Núria Marbà (IMEDEA), Carlos M. Duarte (IMEDEA), Ana Alexandre (CCMAR) and Susana Cabaço (CCMAR)

Seagrasses are clonal plants sharing a similar architecture and presenting a highly organised growth. Seagrass growth relies on the reiteration of ramets, which are composed of modules (i.e. leaves, piece of rhizome, roots, flowers or inflorescences). The understanding of the design of seagrasses provides insight on their growth patterns. Despite the similar architecture of seagrasses, plant size and growth rate vary some orders of magnitude across species. To a large extent, variability in rates between seagrass species reflects differences in plant size, with smaller species growing faster than larger ones. In addition, seagrass growth is able to adapt to environmental change, and it exhibits substantial plasticity. Knowledge on seagrass growth rates allows assessment of meadow productivity and seagrass health, as well as forecasting their capacity to survive disturbances. In addition, seagrass rhizome growth responses to disturbances remain imprinted on the plant allowing reconstruction of past disturbance dynamics.

Vegetative proliferation is the main mechanism of seagrasses to occupy habitat space, and thus it is a critical process for seagrass meadows to spread and persist. Most ramets in seagrass populations are produced as rhizomes elongate. Rhizome growth is the process that regulates the rate of formation and the spatial distribution of ramets (and, thus, modules) within seagrass meadows, and, thus, it constrains the development of their populations. The spread, and maintenance, of seagrass meadows also depend on sexual reproduction since it is the main mechanism regulating patch formation. Hence, information about the effort and success of seagrass reproduction and rhizome growth patterns are essential to predict the time scales of seagrass colonisation and, thus, recovery.

In this chapter we aim to provide an understanding of seagrass growth processes and their rates, as well as mechanisms of seagrasses to spread. We do so by describing seagrass architectural features and the wide repertoire of module addition and growth rates, and discussing the mechanisms and rates of seagrass colonisation. Four sections of this chapter are dedicated to the growth pattern and spreading mechanisms of those seagrass species present along European coasts. At the end we discuss the implications of European seagrass growth and spread for management.

Seagrass achitecture

Seagrasses share a common architecture, all species being clonal, rhizomatous plants. Rhizomes are stems extending horizontally below the sediment surface or vertically, raising the leaves towards, or above, the sediment surface. Seagrasses are modular plants composed of units repeated during clonal growth. Each unit is composed of a set of modules: a piece of rhizome, which can be either horizontal or vertical; a bundle of leaves attached to the rhizome; and a root system (see chapter 1). In addition, the units may hold flowers or fruits, depending on the timing of observation. The morphology of seagrasses does not present any peculiar deviations relative to those of other terrestrial monocotyledons.

The rhizome is responsible for the extension of the clone in space, as well as for connecting neighbouring ramets, thereby maintaining

integration within the clone (see below). The rhizomes of most seagrass species are flexible, whereas those of *Posidonia oceanica* are hihgly lignified and explain the persistence of dead tissues in the sediments, which extends for millenia in *Posidonia oceanica*. Seagrass rhizomes are composed of internodes, the rhizome fragments in between two nodes, which are the insertion points of leaves. The distinct lines identifying the nodes after leaf abscission are also referred to as "leaf scars". Rhizome internodes range widely in size among seagrass species (Table 3.1).

Cymodocea nodosa and *Posidonia oceanica* have both horizontal and vertical rhizomes, whereas *Zostera* species have only horizontal rhizomes. Horizontal rhizomes can revert into vertical rhizomes, which leads to the cessation of their horizontal growth. In turn, vertical rhizomes can branch to produce horizontal rhizomes when the apical meristem of the original horizontal rhizome dies, thereby resuming the capacity for horizontal growth. *Zostera* species bear a leaf on each horizontal rhizome node, and *Posidonia oceanica* and *Cymodocea nodosa* bear leaves on the nodes of both horizontal and vertical rhizomes.

European seagrasses have long and relatively narrow strap-like leaves, ranging in size, from the small leaf areas of *Zostera noltii* to the large leaf areas of *Posidonia oceanica* (Table 3.1). The leaves are often present in bundles on shoots, with up to 8 leaves per shoot in *Posidonia oceanica* and 2 to 5 leaves in the other species. Seagrass roots provide the necessary anchoring and nutrient acquisition, and vary greatly in size across the European species (Table 3.1). Seagrass flowers are often inconspicuous and very simple, for they do not rely on animals for pollination. Seagrass flowers, seeds and fruits range greatly in size from the minute flowers of *Zostera noltii*, which contain multiple ovaries and seeds to the large infloresences of *Posidonia*, which yield large fruits known as sea olives. *Cymodocea nodosa* has separate male and female clones, unlike the other European seagrasses, which are hermafrodites. *Cymodocea nodosa* shoots produce two seeds, attached to the base of the shoot, *Posidonia oceanica* produces half a dozen seeds per shoot, and flowering shoots of *Zostera noltii* and *Z. marina* produce hundreds of seeds.

Formation of leaves, rhizomes and roots: clonal growth

The rate of formation of seagrass leaves, rhizomes and roots, and, therefore, the spread of the clone, depends on the activity of meristems, which are the areas where active cell division, and therefore, growth takes place. The division of the meristems is a rather continuous process, responsible for the maintenance and expansion of seagrass clones. Small seagrass species, such as *Zostera noltii*, produce new leaves much faster (13.71 days) than species with large leaves, such as *Posidonia oceanica* (50.68 days). Roots are typically formed in the internodes of rhizomes, both horizontal and vertical. The death of the meristems results in the discontinuity of the production of new modules (leaves, internodes, etc.). The production of new rhizome material, which leads to the development of new shoots and roots, as well as new branches, is the basis of the growth of seagrass clones. Clonal growth is, therefore, a fundamental component of the production and space occupation of seagrasses, particularly during the colonization of new habitats or their recovery from disturbance.

Meristematic death is also associated to sexual reproduction in *Zostera spp*, which have terminal inflorescences. Meristematic death is followed by the loss of functionality of the modules, which may be subsequently shed, thereby avoiding respiratory losses by non-functional organs. The life span of seagrass shoots, leaves and roots, which reflects the life span of their associated meristems, differs greatly among species, and is scaled to their size, with small species having short leaf life spans and larger species having longer leaf life spans. Leaf life span ranges from a few days in *Zostera noltii* to almost a year in *Posidonia oceanica* (Table 3.1). Available information suggests that roots are longer-lived than leaves, remaining attached to the plants longer than leaves do. The life span of the shoots of European seagrasses ranges from weeks (*Zostera noltii*) to decades for *Posidonia oceanica* (Table 3.1). The life span of the meristems of horizontal rhizomes is presently unknown, except for species, such as *Zostera spp*, for which sexual reproduction is a terminal event.

Seagrass growth rate

Leaf growth rates

New leaves are produced centrally in the leaf bundles by the meristems. Once the leaf has

Table 3.1 Average architectural features and growth rates of European seagrass species. nd: no data. Range of values are indicated within brackets when available. Data from Duarte 1991, Marbà et al 1996, Duarte et al 1998, Marbà and Duarte 1998, and Hemminga et al 1999.

	Cymodocea nodosa	*Posidonia oceanica*	*Zostera marina*	*Zostera noltii*
Leaf surface (cm^2)	9	82.8	34.6	1.15
Shoot mass (mgDW)	82.8	731	272.5	6.5
Fruit size (mm^3)	48	523.6	18	2.8
Horizontal internodal length (mm)	25 (6-53)	3 (1-4)	11 (9-12)	12 (3-20)
Vertical internodal length (mm)	1.4 (0.1-2.5)	1 (0.4-2)	nd	nd
Root length (cm)	21.3	43.1	nd	3.2
Shoot elongation rate (cm shoot $^{-1}$ d^{-1})	1.3	0.8	3.2	0.7
Horizontal rhizome elongation rate (cm $apex^{-1}$ yr^{-1})	40 (7-204)	2 (1-6)	26 (22-31)	68 (10-127)
Vertical rhizome elongation rate (cm $apex^{-1}$ yr^{-1})	1.4 (0.1-16)	1 (0.1-4)	nd	nd
Leaf life span (days)	79 (50-155)	295 (256-345)	88 (33-164)	86 (46-125)
Shoot life span (days)	876	4373	554.8	nd

been produced, it elongates from its basal part, where the leaf meristem is located, until it attains the length characteristic of the species and enforced by the habitat conditions. Seagrass shoots produce new leaves while the standing ones are still growing. When summed over the leaves present on any given shoot, the total daily leaf elongation rate per shoot tends to be on the order of one or a few centimetres

Horizontal and vertical growth rates

The growth rates of seagrass rhizomes vary greatly, from a few centimetres per year in *Posidonia oceanica* to more than 2 m yr^{-1} in *Cymodocea nodosa* (Table 3.1). The vertical rhizomes of *P. oceanica* and *C. nodosa* also extend, but at slow rates of a few centimetres per year. Seagrass horizontal rhizomes branch, accelerating the occupation of space. Horizontal branching is more profuse in small species (e.g. *Zostera noltii* may branch at every node), than in large seagrass species (e.g. *P. oceanica* rhizomes produce, on average, a branch every 30 years). In addition, the horizontal branching angle varies across species, small seagrasses (e.g. *Zostera noltii*) branching with angles close to 90º and large ones (e.g. *P. oceanica*) branching with angles about 40 º. Small seagrass species elongate and branch their rhizomes at much faster rates than large ones. Hence, small seagrass spread in two dimensions at a greater rate than large seagrass species do, so that small seagrass species play a pioneer role and are better able to recover from disturbance.

Environmental and internal controls on growth rates

Seagrasss growth rates, to large extend, are species-specific and scaled to plant size. The negative relationship between seagrass growth and size derives from the increasing construction costs of seagrass modules the larger they are. In addition, seagrasses modulate their growth in response to environmental (e.g. climate, nutrients, sediment quality) and population (shoot density) conditions. Leaf and rhizome growth of European seagrasses, except horizontal rhizome growth of *P. oceanica*, exhibits wide seasonal fluctuations in response to changes in temperature and/or irradiance. Growth of European seagrasses is maximal during summer and minimal during winter, when in most species (*C. nodosa*, *Z. marina, Z. noltii*) it almost ceases. Seagrass response to seasonality remains imprinted on the length of rhizome internodes, allowing retrospective quantification of rhizome growth over the time scale equal to rhizome longevity (i.e. from few months in *Z. noltii* to decades in *P. oceanica*). Seagrass growth requires light conditions being at least 11 % of surface irradiance. During periods of fast seagrass growth, ambient nutrient availability may constrain seagrass growth. Seagrasses with leaf nitrogen and phosphorous concentrations below 1.8 % plant dry weight and 0.2 % plant dry weight, respectively, are susceptible to encounter growth limitations, and they often respond to nutrient sediment additions by increasing leaf growth (e.g. *C. nodosa* and *P. oceanica*). Seagrass

rhizospheres tend to be denser, with more branched root networks when they grow in more nutrient-poorsurroundings. On the contrary, excess of ambient ammonia has been demonstrated to be detrimental for seagrass (*Zostera marina*) survival. Deterioration of sediment quality, reflected by strongly reducing sediment conditions and high concentrations of toxic compounds (e.g. sulphide), has been shown to suppress seagrass growth. Sediment dynamics also alters seagrass growth. Sand erosion and deposition on seagrass beds decrease shoot survival. However, seagrass species with vertical rhizomes are able to cope with sand deposition because the growth of vertical rhizomes, and leaves, of surviving shoots is enhanced with moderate sand burial, increasing at proportional rates as the height of sand accreted. Similarly, they reduce vertical rhizome and leaf growth to minimum rates when sand is eroded. The response of seagrasses to sand burial is triggered by darkness around vertical rhizome meristems. The seagrass response to sediment deposition remains imprinted on the vertical rhizomes as long internodes, allowing retrospective identification of burial/erosional events. Seagrass clonal growth is also dependent on the density of neighbouring shoots. Horizontal rhizome elongation and branching rate of at least *C. nodosa* and *Z. marina* decrease with increasing shoot density around the growing rhizome apex. Because rhizome growth regulates shoot proliferation in seagrass beds, regulation of clonal growth rate in response to neighbouring shoot density is an important mechanism to avoid shoot density-mortality under crowding conditions, and, thus, intra-specific competition.

The response of seagrass growth to environmental and population changes is species-specific as a consequence of differences in sensitivity to environmental forcing and capacity to uncouple plant growth from ambient conditions among species. For instance, seasonality in *C. nodosa* growth is highly dependent on temperature, whereas that in *Z. marina* growth is mainly forced by light conditions. The high sensitivity of *C. nodosa* to temperature conditions has been attributed to the tropical origin of this seagrass genus. Seagrasses store resources (carbon and nutrients) in their belowground organs that they use during periods when ambient resource availability does not suffice to fulfil resource demand. Resource storage capacity largely depends on plant size and plant longevity, both features being strongly species dependent. In addition, the architecture of seagrasses allows the ramets of a clone to remain physiologically integrated over time scales ranging from, at least, less than one month in small species (*Z. noltii*) and several years in large ones (*P. oceanica*). The greater capacity of large seagrasses to store and mobilize resources in their clones than small ones allows them to uncouple their growth more from ambient resource heterogeneity, and, thereby, buffer their growth responses.

Seagrass spreading

Vegetative vs. sexual spreading

The reproductive biology of seagrass species has interested naturalists for about two centuries. Flowering of seagrasses is often controlled by temperature and often occurs simultaneously across large spatial scales. European seagrass species flower in late spring , , and some of them (*Zostera spp*) throughout the summer as well, when irradiance improves and water temperature increases, except for the Mediterranean species *Posidonia oceanica*, which flowers in the fall (October). Flowering is a rare event for most seagrass species, where typically < 10 % of the shoots flower each year. Yet, flowering is profuse in annual *Z. marina* populations developing at the intertidal zone. The reproductive effort of seagrasses can be highly variable between years and among populations, and episodic mass flowering can occur in connection to climatic extremes, such as the massive flowering of *Posidonia oceanica* in connection to extreme summer temperatures in 2003. Disturbances, such as burial derived from the migration of sand waves may also enhance seagrass flowering. Because of the low probability of flowering, sexual reproduction is a negligible component of the carbon allocation of seagrasses, involving < 10 % of the annual production for most species. All European seagrass species have hydrophilous pollination, in which pollen grains are released in the water column to fertilise the female flower. Seed production can reach thousands of seed m^{-2} for *Zostera* species, whereas it is in the order of, at most, tens of seeds m^{-2} for *Cymodocea nodosa* and *Posidonia oceanica*. A significant percent of the seeds seagrass produce are lost before being released due to predation by invertebrates and fish.

Seagrasses can disperse through sexual propagules as well as through detached or drifting rhizome fragments. Dispersal by fragments was considered to be rare, but new evidence suggests that the importance of this mechanism may have been underestimated. Dispersal can also occur

through a combination of both processes, as flowering shoots may detach and disperse, subsequently releasing the seeds. The mature seeds of *Cymodocea nodosa* are produced at the base of the shoots, and are often positioned at, or just below, the sediment surface. These seeds are, therefore, not likely to disperse far. *Zostera* seeds disperse with currents, and have been shown to have a relatively short dispersal range restricted to tens of meters. In contrast, the seeds of *Posidonia oceanica* remain buoyant for hours and can be potentially dispersed across distances of tens or even hundreds of kilometres, although there are no direct observations to confirm whether such potential is realised.

Patch formation

Once in the sediments, the seeds of some seagrass species can remain dormant for some time before germinating, with a documented dormancy period of about half a year for *Zostera marina* and 7-9 months for *Cymodocea nodosa*, thereby building a rather ephemerous seed bank. Seedling density is comparatively low (one or two orders of magnitude) relative to seed production, due to multiple aggregated losses. These losses are due to many factors, including lack of viability, physical damage, export to unsuitable areas, burial, and predation. Initiation of clonal growth (i.e. rhizome extension) will lead to the formation of patches by the seedlings. Yet, most seedlings die without ever initiating clonal growth, because these require the accumulation of important amounts of resources, such as nutrients.

Patch growth

The basic components of the clonal spreading of seagrasses, which leads to the formation of patches or the maintenance of closed meadows, are the rate of horizontal extension, and the probability and angle of branching of the rhizomes. The clonal growth of seagrasses can, therefore, be simulated from knowledge of this basic set of rules, an approach that has proved most useful in the examination of space occupation by clonal plants. The simulation of the space occupation by seagrass clones confirms the prediction that small seagrass species have a less efficient, but more compact occupation of space, which has been referred to as the “phalanx” strategy, whereas large, slow growing species have a more efficient, but looser occupation of the space, the “guerrilla strategy” Indeed, if large species did have broader branching angles, the time required to occupy the space would be so long that they would not be able to develop meadows despite the long life span of their modules. A simulation analysis of seagrass clonal growth also showed that the branching process continuously accelerates the occupation of space, such that the space occupied by a seagrass clone increases as the third power of time for all seagrass species simulated. Thus, branching rates and branching angles are even more important determinants of the rate of space occupation than the linear extension rate of the rhizomes, the parameter that has received most attention to date.

The growth rates of patches of European seagrass species are constrained by the growth rate of their rhizomes, with *Posidonia oceanica* patches showing the slowest growth (2 cm year^{-1}) and *Cymodocea nodosa* showing the fastest growth (200 cm year^{-1}).

Posidonia oceanica

P. oceanica is a long-living and very slow-growing seagrass species. *P. oceanica* leaves may live for a bit less than 1 year, vertical rhizomes for several decades, and clones probably for centuries. The slow horizontal rhizome elongation and branching rate (Table 3.1) of this species explains the extremely slow spread of its clones. Simulation models based on rhizome growth and branching patters indicate that *P. oceanica* should spend 350 yr to develop a 15 m diameter clone. *P. oceanica* vertical rhizomes elongate at rates that may be of similar order of magnitude as horizontal rhizomes do, which is unusual for seagrass species with differentiated rhizomes. Because of the relatively fast vertical growth of *P. oceanica* as compared to horizontal growth, the long life span of the meadows and the slow decomposition of its rhizomes, *P. oceanica* is able to develop reefs up to 3 m high and meadows with complex topography, particularly when *P. oceanica* colonises shallow coastal areas. *P. oceanica* growth is particularly sensitive to deterioration of sediment quality and, at meadow depth limit, water quality.

P. oceanica flowers between August and November. The number of flowering shoots in *P. oceanica* meadows is usually very low, generally lower than 3 % per year. However, flowering intensity widely fluctuates between years. Massive flowering events (when more than 10% shoots flower) have been observed associated to extremely warm summers. Flowering intensity also varies with water depth, decreasing the number of flowering shoots with increasing water depth, and it depends on local conditions. Many

P. oceanica female flowers do not succeed to develop viable fruits as a consequence of fruit abortion and, to less extent, predation. Actual seed production is less than 1% of potential seed production provided the amount of ovaries produced during flowering. Very little information is available on seedling survival and clone initiation rate, but it should be extremely slow.

The little investment and low success of sexual reproduction, combined with the extremely slow clonal spread of *P. oceanica* explains the extremely slow colonisation rate of this species. Numerical models simulating the occupation of space by a *P. oceanica* meadow indicate that it would need 600 years to cover 66 % of the space available. Similar colonisation time scales have been retrospectively calculated based on patch size and patch growth rate in patchy *P. oceanica* meadows. The very long time scales for colonisation of this species indicate that recovery of disturbed *P. oceanica* meadows, where important plant losses have occurred, would involve several centuries.

Zostera marina

Zostera marina has intermediate rates of growth and spread compared to other European seagrasses. Besides the potential for spread derived from the horizontal growth rates of the rhizomes, *Zostera marina* is able to release large numbers of seeds. At the time of reproduction, eelgrass shoots produce inflorescences which can each develop large numbers of seeds. Reproductive shoots die off following seed set, so that flowering represents a terminal event for eelgrass shoots. Seed production rates in *Zostera marina* beds reach several thousand per square meter. However, they do not travel far – a few meters at best - from the mother plant after being released, as they are negatively buoyant and sink to the bottom. However, flowering shoots may detach, because of disturbance, and float away, releasing the seeds at considerable distances from the stand where they were produced, which is the mechanism for long-term dispersal available to this species. In addition, swans and geese may ingest seeds and transport them, although this potential mechanism has not yet been investigated. A significant fraction of the seeds released are lost due to the activity of grazers, such as crabs, which have been shown to significantly reduce the seed pool produced by eelgrass.

The germination of eelgrass seeds leads to the initiation of patches, which are subject to intense dynamics. Most of the new patches formed disappear within one year and only a few grow to effectively increase the cover of elelgrass in coastal zones. The time scales required for the recolonisation of eelgrass meadows has been estimated, provided favourable environmental conditions, to be in the order of a decade. *Z. marina* clones, however, may persist over centuries in areas where sexual reproduction is scarce (e.g. Baltic Sea).

Zostera noltii

The fast rate of rhizome elongation (68 cm year^{-1}) and profuse great branching rate lead to a compact space occupation. The species has a high leaf turnover rate, i.e. as new leaves are formed, the older are shed in a rapid process during the shoot lifetime. Besides, the leaf growth rate is also high, as well as the shoot production, which represent much of the production of the species. As a small species, the modules of *Z. noltii* have a short life span, with high mortality and recruitment rates, which is typical of colonizing seagrass species. The high rates of growth and production of *Z. nolti* allow this species to sustain even under considerable disturbance.

Beside vegetative development, these plants can reproduce sexually by producing flowering shoots (Potographs 3-4) and seeds (Photograph 5). Seed production and other events related to this process (flowering, seed release, dispersal, and germination) are valuable to maintain genetic diversity and may be the only significant mechanism for seagrass colonization of bare sediment areas. Coupling both vegetative and reproductive patterns may therefore constitute an excellent survival strategy in adverse and disturbed environments or in the establishment of new areas.

Flowering of *Z. noltii* can extend from March to November but the flowering season may vary from place to place, since factors such as water temperature, day length, tidal amplitude and fluctuating salinity regimes control the flowering event. The flowering shoots flourish from the rhizome as the vegetative ones and consist of several inflorescences, each containing the male and female flowers. The female flowers are pollinated by males from different inflorescences, to avoid self-pollination. The females that were fertilized develop a fruit inside, which originates a seed. Seeds are not likely to disperse far since they are negatively buoyant. However, detached flowering shoots containing seeds may be

transported by water currents over long distances. Observation of *Z. noltii* seedlings in the field is a rare event, for less than 5% of the plants originate from seeds. In spite of the investment in sexual reproduction, flowering represents less than 10% of the shoots, which suggests that this is not the main way of reproduction.

Cymodocea nodosa

C. nodosa growth ranks amongst the fastest ones across European seagrasses. The fast clonal growth of this species allows the clones to spread across 300 m^2 after 7 years. The life span of *C. nodosa* modules and ramets is intermediate, average shoot population life-span varying between 4-22 months, and average leaf life-span ranging from 2 to 5 months. *C. nodosa* clones, however, may live for at least 1 decade. *C. nodosa* growth almost exclusively occurs during spring and summer. *C. nodosa* growth exhibits substantial plasticity, which allows this species to survive disturbances. For instance, vertical and horizontal rhizome growth of *C. nodosa* is plastic enough for this species to colonise areas with intense sediment dynamics, such as bedforms with subaqueous dunes, with an average amplitude of 20 cm (range 7-65 cm) and wave length of 21 m (7-29 m), that migrate at average velocities of 13 m yr^{-1}. The close coupling between *C. nodosa* vertical rhizome growth and sediment accretion has been used to quantify shallow coastal sediment dynamics impossible to be measured with conventional sedimentary techniques. *C. nodosa* also exhibits substantial plasticity in response to ambient nutrient availability.

Only *C. nodosa* shoots older than 1 year flower, and they do so between March and June. Fruit development takes 2-3 months, although maximum density of shoots bearing fruits is observed in July-August. Afterwards, fruits detach from the mother shoot and, because they have negative-buoyancy, they are rapidly buried into the sediment nearby the mother plant. During events of intense sediment dynamics (e.g. strong storms), however, seeds may be transported across long distances, since there are meadows separated from the closest one by more than 300 km, and seeds of *C. nodosa* can be observed, although not very often, washed on the beaches. From April til June of the following year seeds germinate. *C. nodosa* clone formation rate has been estimated to be about 0.009 clones m^{-2} yr^{-1} in an area with intense sexual reproduction. However, clone mortality rate is about 50-70 % during the first year of life, hence, decreasing substantially the success of sexual reproduction. Reproductive effort and success in *C. nodosa* exhibits temporal and spatial heterogeneity. Flowering intensity, for instance, has been observed to increase in response to sand burial, like in other seagrasses. In addition, seed production in *C. nodosa* should be constrained by the spatial distribution and abundance of male and female clones. The consequences of clone sex composition on reproductive success are evident when examining *C. nodosa* meadow genetic diversity. For instance, there is almost no genetic diversity in a *C. nodosa* meadow at the Algarve (S Portugal), where no female flowers have been observed.

The fast growth of *C. nodosa* clones and the relatively high patch formation rate of this species, when compared with the other European seagrasses, indicate that *C. nodosa* should be able to develop a meadow within a decade, if the colonisation process were initiated, on bare sediments. The time scales for meadow recovery if not all *C. nodosa* vegetation were lost should be even shorter. The rapid occupation of space by *C. nodosa* resulting from fast clonal growth, and the relatively high patch formation rate of this species explains the pioneering role that *C. nodosa* play during succession process in the Mediterranean.

Conclusion

European seagrass flora encompasses species with slow-growing rhizomes (*Posidonia oceanica*), and intermediate rhizomes expansion rates (*Cymodocea nodosa*, *Zostera marina*, *Z. noltii*), when compared with the range of clonal growth rates displayed by seagrasses. The slow horizontal rhizome elongation and branching rates observed in *P. oceanica* forecast slow (from centuries to millenia) recovery time scales for this species. Conversely, the rates of horizontal rhizome extension and branching frequency quantified for the other European seagrasses should allow recovery time scales of decades. In addition to the differences in clonal growth rules observed between European seagrass species, their growth is able to adapt to environmental change. Seagrass plasticity, however, differs among species, *C. nodosa* being amongst the most plastic species, and *P. oceanica* growth being the least plastic.

Knowledge on seagrass growth rates and success of reproductive effort is crucial to manage seagrass ecosystems, particularly to derive expectations on the recolonisation times required to recover seagrass meadows. Because of the

low survival rate of seedlings and young patches, they should be particularly protected. The time scale in which patches are most vulnerable ranges from a few months for *Zostera noltii* to half a century for *Posidonia oceanica*.

The time required for the patches to develop meadows ranges, across the European seagrass flora, from months to a year for *Zostera noltii*, to less than a decade for *Zostera marina* and *Cymodocea nodosa*, and several centuries for *Posidonia oceanica*. The acceleration of the colonisation process along these, sometimes too extended, time scales can be promoted through the maintenance of adequate habitat conditions, including, improved light penetration and reduced organic and nutrient inputs to the waters.

The recovery of *Zostera noltii* is relatively fast, whereas that of the other species is slow, and *Posidonia oceanica*, in particular, does not have the capacity to recover in opperational time scales. Hence, the management approach to the slow-recovering species should emphasize the conservation and protection of the area they cover as to avoid losses.

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4 Which factors regulate seagrass growth and distribution?

The growth and distribution of seagrasses are controlled by the physical, chemical and biological properties of the environment they live in. Sufficient light, nutrients and inorganic carbon are basic needs for photosynthesis, but also a suitable substratum, moderate exposure, temperature and various biological factors affect the distribution of seagrasses. A complicated interaction between the factors makes it difficult to separate the effects of single factors as well as predict presence or distribution at a given time and place. Although complicated the most important factors determining seagrass growth and distribution can be identified.

By Tina M. Greve and Thomas Binzer (FBL)

The basic physical requirements of seagrasses are sufficient light, a suitable substratum and moderate levels of wave exposure, but the presence and distribution of seagrasses around the world are also regulated by a number of other factors. The basic requirements for seagrass growth are similar to that of terrestrial plants. However, life in the aquatic environment differs considerably in many respects from the terrestrial environment, and some resources may be limited in accessibility or quantity. Besides basic physical and chemical requirements for growth, biological competition from other species may also influence the growth and distribution of the seagrasses. In this chapter the regulating factors of seagrass growth and distribution will be described along with a short description of the possible differences between the four European seagrass species that might exist.

Abiotic factors

Light

Light is one of the most important factors in the regulation of seagrass maximum depth distribution and seagrasses are therefore only growing in shallow coastal waters at depths receiving enough light for seagrass growth. Light is required to drive photosynthesis and hence growth, but in contrast to the terrestrial environment light is a limited resource in the aquatic environment. When light is passing through the water column it is absorbed or reflected by particles, such as phytoplankton, suspended material and dissolved substances. Light therefore attenuates exponentially with increasing depth. In addition to varying between areas, light attenuation may also vary considerably within an area since water turbidity can be a result of local physical and biotic factors. Seagrasses have a minimum requirement for light: a rule of thumb of approximately 10 % of the surface irradiance (Fig 4.1). This value is however, an average and therefore not applicable for all seagrass species at all times, since the light compensation point of the plants differs among species and depends on other environmental factors such as temperature and sediment chemistry.

Within the range of irradiances that seagrasses can exist, special morphological acclimatisation's in growth strategy are required. Decreasing light level causes the plants to prolong the leaves and thin the density of shoots. By this acclimatisation more light can be captured and converted into photosynthetic production. The growth acclimatisation to reduced irradiance is most pronounced in *Z. marina*, where leaf length varies from 15-20 cm in shallow water to more than 120 cm in deep water, but also *C. nodosa* and *P. oceanica* acclimatise to reduced irradiance, primarily by reducing shoot density in deep waters. *Zostera noltii* rarely shows this acclimatisation since it primarily grows in shallow intertidal waters with sufficient light. However, for most species light defines the lower limit of their depth distribution.

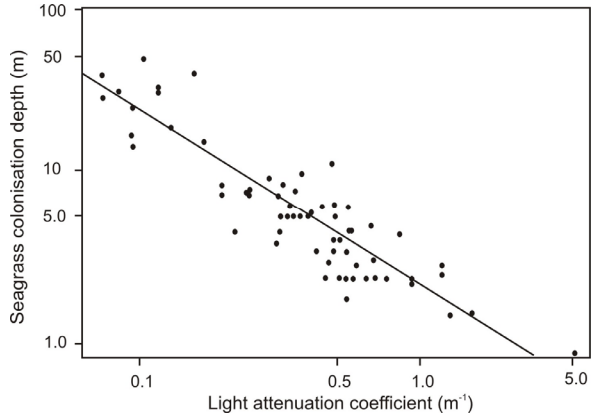

Figure 4.1. The relationship between seagrass colonisation depth and light availability expressed as the light attenuation coefficient (m^{-1}). The line is expressed by: log Zc (m)=0.26-1.07 log *K* (m^{-1}). Redrawn from Duarte 1991.

Physical exposure

Currents, wave action and tide are in contrast to light, the most important factors controlling the upper depth limit for seagrass distribution. In general it is estimated that seagrasses do not exist at flow velocities above 1.5 m per second or at very exposed shores. Currents and wave action prevent seagrass growth and distribution by causing resuspension and transport of the sediment. Besides affecting the general light climate of the water column, erosion can expose roots and rhizomes causing the seagrasses to detach from the sediment. Additionally, very strong currents or wave action may tear up entire plants or prevent new shoots from being established. As the sediment resettles other plants may be buried by sediment. High rates of resuspension or siltation can be crucial for seagrass populations although *Posidonia* or *Cymodocea* species with vertical shoots, might survive events of high sediment deposition by elongating vertical shoots.

Substratum

Another very important factor in the regulation of seagrass distribution is the presence of a suitable substratum. While macro algae are attached to stones and rocks on the seafloor, seagrasses mainly require a soft substrate of gravel, sand or mud, were rhizomes can elongate and roots can fasten. *Zostera marina, Z. noltii* and *C. nodosa* can be found on gravel as well as in mud rich in organic matter. In contrast, *P. oceanica* is usually found in more coarse sediments. There are exceptions and some seagrass beds can actually be found on rocky substrates. Some *C. nodosa* populations near Albufeira on the Algarve coast of Portugal are growing on rocks with the roots and rhizomes inserted in cracks and crevices in the rock.

Carbon

In addition to light, seagrasses need inorganic carbon for photosynthesis. In water, inorganic carbon exists in three forms: CO_2, HCO_3^- and CO_3^{2-} depending on the pH of the water, and both CO_2 and HCO_3^- are assimilated by seagrasses in the photosynthetic process. However, the leaves of seagrasses have a low capacity for extracting inorganic carbon and the photosynthesis seems, even under normal pH and salinity conditions, to be limited by the availability of inorganic carbon under high light conditions (Beer and Koch 1996). At high pH due to high rates of photosynthesis in shallow waters, carbon limitation is even more likely. Hence, seagrasses may profit from the ongoing increase in global atmospheric carbon dioxide caused by the profound use of fossil fuels (chapter 5). In estuaries supplied with freshwater of low inorganic carbon contents (e.g. the Baltic Sea), carbon limitation is more pronounced, and the increase in atmospheric CO_2 could have even greater impacts. It is, however, not known to what extent inorganic carbon can be supplied from the sediment via the roots and plant lacunae to seagrass leaves, and, therefore, the actual importance of carbon limitation to seagrass photosynthesis and growth needs to be examined further.

Nutrients

Seagrasses also require different kinds of inorganic nutrients, where nitrogen and phosphorous are the most quantitatively important. Nutrient requirements for seagrasses are lower than for other aquatic organisms such as macro algae and phytoplankton. It is estimated that seagrasses requires about 4 times less nitrogen and phosphorous per weight than phytoplankton cells. This gives the seagrasses an advantage for growth in nutrient-poor environments compared with other primary producers. In general, nutrient levels in the water column of seagrass beds are typically low, especially in warmer areas such as the Mediterranean, but in addition to uptake of nutrients from the water column seagrasses can take up nutrients from the sediment. Most sediments are rich in nutrients due to the

mineralization of organic matter. Exceptions are carbonate sediments, which bind phosphorus and thereby induce phosphorous limitation to the these plants. *Posidonia oceanica* and *C. nodosa* often grow in carbonate sediment with low nutrient content, and it has been documented that very low nutrient concentrations can reduce the growth of plants. *Zostera marina* and *Z. noltii* usually grow in organic sediments and are seldom limited by nutrients.

Temperature

Temperature affects all biological processes primarily by increasing reaction rates of the biochemical pathways. The most important processes, photosynthesis and respiration, are slow at very low temperatures but increase with increasing temperature. Respiration exceeds however, photosynthesis at high temperatures resulting in a negative energy balance within the plant. The temperature therefore defines the geographical limits for growth, although some adaptation to the local temperature regime is possible. The temperature tolerance differs between species. *Zostera marina* is in general adapted to relatively cold habitats with temperatures ranging between -1ºC in winter and approximately 25ºC in summer. *Zostera noltii* also thrives in cold habitats in the north but endures higher temperatures than *Z. marina*. The absence of *Z. noltii* in the northern/arctic part of Europe might be due to a higher temperature required for flowering than in *Z. marina* but this aspect needs further investigation. *Cymodocea nodosa* and *P. oceanica* grow in warmer environments with temperatures ranging from approximately 10ºC up to about 30ºC. The temperature is therefore considered the overall parameter controlling the geographical distribution of the species in Europe.

Salinity

Seagrasses grow at salinities ranging between 5 ‰ and 45 ‰. Salinity affects the osmotic pressure in the cells, but many seagrasses are well adapted to sudden changes in salinity. For example, seagrasses often grow in river outlets or estuarine habitats where salinity changes rapidly and varies considerably over time. Some species have, however, a higher tolerance to changes in salinity than others. *Zostera marina* and *Z. noltii* are frequently observed in estuaries and *Z. noltii* also occurs on intertidal flats where changes in salinity can change from only a few ‰ to more than 30 ‰ within a few hours. *C. nodosa* usually inhabits more saline areas with fluctuations from 26 - 44‰, whereas *P. oceanica* only inhabits marine waters with high salinities. Furthermore *P. oceanica* does not endure large variation in salinity like the three other species. In rare cases, salinity increases significantly above the mean oceanic values of 33-36 ‰ because of desalination plants or profound water evaporation from shallow areas. Very high values of salinity (up to 60 ‰) are suspected to cause severe diebacks in shallow areas.

Oxygen

Seagrasses need oxygen to supply their metabolism of both above and below ground tissue. But, while leaves are usually situated in the oxygenated water column, roots and rhizomes are buried in anoxic sediments. Under normal circumstances, photosynthetically generated oxygen or water column oxygen is transported to roots and rhizomes by simple diffusion from the leaves to the roots in a well developed system of air tubes (lacunae) running through the plant. The below ground tissue may experience lack of oxygen if the water column becomes hypoxic or anoxic during periods of high degradation of organic matter in the sediment coupled with a stratified water column. Anoxic conditions influence the metabolism of the plants resulting in poor energy availability and production of toxic metabolites, both of which may negatively affect growth and survival of the plants. Anoxic conditions in the belowground tissue may also cause invasion of sulphide from the sediment.

Sulphide

High concentrations of sulphide in the sediment can harm seagrasses since sulphide is a plant toxin inhibiting respiration. Sulphide is present in sediment rich in organic matter and poor in iron. To be toxic, sulphide has to enter the plants, which is not possible under normal conditions when oxygen is present in the belowground tissue and in micro zones around the roots. Sulphide is oxidized in the root zone to the harmless compound sulphate before reaching the root surface. During oxygen deficiency in the water column, the supply of oxygen will be inadequate resulting in root anoxia and sulphide invasion. When the gaseous sulphide penetrates to the lacunae it will readily spread and can reach the meristem where it might be fatal to the plant. In the presence of iron in the sediment, iron reacts with the sediment sulphides, precipitating them as iron-sulphur minerals and thereby “buffer” the effects of toxic sulphides on the seagrasses.

Biotic factors

Growth and distribution of seagrasses are affected by other organisms primarily through competition or herbivory. High nutrient concentrations in the water column cause epiphytes and filamentous algae to develop in high densities, affecting the light climate in the water column and hence seagrass depth distribution. Furthermore, epiphytes expand the boundary layers around leaves limiting uptake of oxygen, inorganic carbon and nutrients. Filamentous algae can also form dense mats at the seafloor that will reduce water flow around the leaves and reduce the oxygen content in the water when they are degraded.

Competition

Competition between different species of seagrass will also set limits to growth and distribution. For example, *Z. noltii* often colonises the intertidal zone or the shallow waters where other species cannot establish populations. In deeper waters where *Z. marina* or *C. nodosa* can establish, they apparently have a competitive advantage and *Z. noltii* beds will disappear. Mussels (*Mytilus edulis*) may also compete with seagrass for the occupation of space. Mussels occasionally settle on the leaves of e.g. *Z. marina* in very high numbers. As they grow and become larger they eventually cover the bottom and suppress plant growth

Grazing

Grazing by waterfowls or invertebrates can also be important for the distribution of seagrasses in certain areas although it is not considered a major controlling factor. Birds usually graze on fresh leaves on very shallow water causing the leaves to disappear. Occasionally even rhizomes are consumed causing the seagrass population to disappear for a period. The effect of grazing by birds is highly variable in time and space since the population density of birds in a given area may vary considerably. Examples of grazing birds in European coastal water are mute swan (*Cygnus olor*), brent goose (*Branta bernicla*), pintail (*Anas acuta*), wigeon (*Anas penelope*) and mallard (*Anas platyrhynchos*), all of which graze on the *Zostera* species. Grazing from fish is probably not as profound as from birds and only a few fish species in the world feed exclusively on seagrasses. As an example of fish grazing on European seagrasses are the sparid fish (*Sarpa salpa* L.) which graze on *Posidonia oceanica* in the Mediterranean. The crustacean *Idotea chelipes* and the purple sea urchin *Paracentrotus lividus* have also been found to graze on *Z. marina* and *P. oceanica*, respectively. The importance of grazing for the distribution of seagrasses is in general considered relatively low in European waters, but investigations suggest that grazing on fresh leaves is more important in the fast growing species *Z. marina*, *Z. noltii* and *C. nodosa* than in *P. oceanica* probably because the content of indigestible lignin-cellulose compounds is considerably larger in *P. oceanica*.

Conclusion

Seagrasses can grow in the shallow coastal zone of marine habitats, but several factors regulate growth and distribution of the plants in these areas. Light is the most important factor regulating the lower depth distribution, whereas exposure is the most important factor regulating the upper depth distribution. Seagrasses might be nutrient limited even though they assimilate nutrients from the sediment. In local areas biological factors such as grazing may also affect the distribution of seagrasses, but the overall effect of grazing in European waters are considered small.

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5 What may cause loss of seagrasses?

Seagrass decline is a worldwide phenomenon requiring managerial interventions to be reverted. However, any actions to stop or reverse seagrass decline require a proper understanding of the causes behind the decline. This chapter described the multiple man made and natural causes for seagrass decline.

By Carlos M. Duarte, Nuria Marbà (IMEDEA), Rui Santos (CCMAR)

While seagrasses are recognized as priority subjects for conservation efforts in international (e.g. Rio Convention, EU's Habitats Directive) and national frameworks, there is evidence that they are experiencing significant widespread decline. Seagrasses exist at the land-sea margin and are highly vulnerable to pressures from the world's human populations, which live disproportionately along the coasts. Human population growth, with concomitant increased pollution, hardening and alteration of coastlines, and watershed clearing, threatens seagrass ecosystems and has resulted in substantial and accelerating seagrass loss over the last 20 years. Globally, the estimated loss of seagrass from direct and indirect human impacts amounts to 33,000 km^2, or 18 % of the documented seagrass area, over the last two decades, based on an extrapolation of known losses. Reported losses probably represent a small fraction of those that have occurred and many losses may remain unreported, and indeed may never be known because most seagrasses leave no long-term record of their existence. Causes range from changes in light attenuation due to sedimentation and/or nutrient pollution, to direct damage and climate change. Losses are often experienced first at either the lower or upper depth limit of the seagrass meadows. Seagrasses require an underwater irradiance generally in excess of 11 % of that incident in the water surface for growth, a requirement that typically sets their depth limit. The upslope limit of seagrasses is imposed by their requirement for sufficient immersion in seawater or tolerable disturbance by waves and, in northern latitudes, ice scour.

There is, therefore, concern that the functions seagrasses have performed in the marine ecosystem will be reduced or, at places, lost altogether. Seagrass loss leads to a loss of the associated functions and services in the coastal zone. The consequences of seagrass loss are well documented, through observations of the changes in the ecosystem upon large-scale seagrass losses. Seagrass loss involves a shift in the dominance of different primary producers in the coastal ecosystem, which can only partially compensate for the loss of primary production. For instance, the increased planktonic primary production with increasing nutrient inputs does not compensate for the lost seagrass production, so that there is no clear relationship between increased nutrient loading and ecosystem primary production. The loss of the sediment protection offered by the seagrass canopy enhances sediment resuspension, leading to a further deterioration of light conditions for the remaining seagrass plants. The extent of resuspension can be so severe following large-scale losses, such as that experienced during the *Zostera marina* wasting disease, that the shoreline may be altered. The loss of seagrasses will also involve the loss of the oxygenation of sediment by seagrass roots, promoting anoxic conditions in the sediments. Seagrass loss has been shown to result in significant loss of coastal biodiversity, leading to a modification of food webs and loss of harvestable resources. In summary, seagrass loss represents a major loss of ecological as well as economic value to the coastal ecosystems, and is therefore, a major source of concern for coastal managers.

Natural and anthropogenic causes

The likely primary cause of seagrass loss is reduction in water clarity, both from increased nutrient loading and increased turbidity. Run-off of nutrients and sediments from human activities on land has major impacts in the coastal regions where seagrasses thrive. The relatively high light requirements of seagrasses make them vulnerable to decreases in light penetration of coastal waters. Along temperate, more industrialized coasts, the losses of water clarity come from rapidly increasing inputs of nitrogen and phosphorus from waste discharge, atmospheric deposition, and watershed run-off. In contrast, in tropical areas, the major impacts on water clarity are extensive sediment discharge into coastal waters caused by watershed deforestation and clearing of the mangrove fringe.

Worldwide, anthropogenic nutrient over-enrichment of coastal waters is the factor responsible for much of the reported seagrass decline. The primary cause of nutrient enrichment in estuarine and coastal waters is anthropogenic loading from coastal watersheds. In general, pristine estuaries and coastal seas are nitrogen-limited and nitrogen inputs from point and non-point sources causes eutrophication. Increased nutrient loading is widely acknowledged to alter the structure and function of coastal ecosystems. In addition to nutrient inputs from land, increased nutrient inputs are also occurring in coastal areas adjacent to industrialized regions of the world through direct atmospheric deposition of nitrogen, providing additional nutrients that can only be reduced at the source.

Poor land use practices also result in increased soil erosion and the delivery of vast quantities of sediment into coastal waters. Removal of terrestrial vegetation leads to erosion and transport of sediments through rivers and streams to estuaries and coastal waters where the suspended particles create turbidity that reduces water clarity and and increase sedimentation above levels tolerable to seagrass.

Direct human impacts to seagrasses, in addition to the major indirect impacts discussed above, threaten the habitat particularly in densely populated areas. Direct impacts from human activity include: i) fishing and aquaculture, ii) introduced exotic species, iii) boating and anchoring, and iv) habitat alteration (dredging, reclamation and coastal construction). Fishing methods such as dredging and trawling may significantly affect seagrasses by direct removal. Damage to *Zostera marina* by scallop dredging reduces shoot density and plant biomass, and digging for clams can also exert extensive damage. Many of these impacts remain unquantified as yet, and their long term effects are poorly known. In the Mediterranean, the exploitation of marine resources, and the use of certain types of fishing gear like bottom trawls, has detrimental effects on seagrass beds. Mussel harvest in the Dutch Wadden Sea is believed to be a major factor in the loss of *Z. marina* and *Z. noltii* there.

Aquaculture is growing fast in both the Mediterranean and Atlantic European coast. Aquaculture has been shown to produce major environmental impacts, particularly due to shading, eutrophication and sediment deterioration through excess organic inputs. The effects of fish farms and other aquaculture developments are of concern as areas of productive seagrass habitats are often targeted for such developments, such as in the Mediterranean coast. Mussel culture adversely affects *Z. marina* and *Z. noltii* beds in France. Seagrass beds as far as 100 m from fish cages can be impacted by the delivery of feed to the fish.

The introduction of exotic marine organisms, from accidental release, vessel ballast water, hull fouling and aquaculture, remains an area of concern, particularly where the introduced species are competitors for soft bottom substratum such as the alga *Caulerpa* and the fan worm *Sabella spallanzanii*. Large scale engineering projects have also resulted in species invasion, such as that by the Lesepian migrant *Caulerpa racemosa*, introduced through the Suez Canal. The Suez Canal also allowed the introduction of the seagrass *Halophila stipulacea* into the Mediterranean.

Direct boat propeller damage to seagrass communities has been recorded, particularly in the Florida Keys, and is prevalent in shallow areas with heavy boat traffic. Boat anchoring leaves scars in *Posidonia oceanica* landscapes, as do boat moorings. Return of large temperate meadow-forming seagrasses to mooring scars may take decades. Docks and piers shade shoreline seagrass, an effect that may fragment the habitat. Boating may also be associated with organic inputs in areas where boats do not have holding tanks.

The development of the coastline, particularly related to increased population pressure, leads to alteration and fragmentation of habitats available for seagrasses in coastal waters. Coastal development, construction of ports, marinas and groynes, is usually localized to centers of

population. Housing developments impact coastal water quality, and the number of houses in a watershed has been directly correlated to rate of seagrass loss. Construction of the causeway at the southern end of Cockburn Sound, Western Australia, in combination with industrial pollution, destroyed existing seagrass. Construction of roads through shallow waters, without proper consideration to maintaining water flow, may also affect circulation and lead to seagrass loss, such as observed in Cuban coastal waters rendered hypersaline by the effects of road construction over shallow lagoon areas on water exchange.

Dredging and reclamation of marine environments, either for extraction of sediments or as part of coastal engineering or construction, can remove seagrasses. Filling of shallow coastal areas, known as reclamation, can directly eliminate seagrass habitat and results in hardening of the shoreline, which further eliminates productive seagrass habitat, as seen throughout Tokyo Bay. Groynes alter sediment transport in the nearshore zone. Dredging removes seagrass habitat as well as the underlying sediment, leaving bare sand at greater depth, resulting in changes to the biological, chemical and physical habitat values that seagrasses support. Beach replenishment may impact adjacent seagrasses by delivering sediment that may shade or bury the seagrasses. Beach nourishment can also impact seagrasses growing in areas where sediments are collected, often at depths < 30m.

Because of the requirements of seagrass for adequate light and sediment conditions, they are particularly affected by disturbances that modify the water and sediment quality. The seagrass shallow coastal environment is also particularly prone to physical disturbance, whether by waves or turbulence associated with strong storms. Because disturbance is a sporadic phenomenon, seagrass meadows are highly dynamic ecosystems. These dynamics include widespread loss, such as the wasting disease that led to catastrophic die-back of eelgrass (*Zostera marina*) meadows on both sides of the Atlantic in the 1930s, as well as more recent massive losses such as that in Florida Bay (USA), which is one of the largest areas of seagrass ecosystem world-wide. The causes and the possible role of human-derived impacts in such losses are still uncertain. Strong disturbances, such as damage by hurricanes, can also lead to major seagrass losses. Smaller-scale, more recurrent disturbances, such as that caused by the motion of sand waves in and out of seagrass patches and that caused by large predators, such as dugongs or geese, represent the main factor structuring some seagrass landscapes, which are characteristically patchy. In contrast, some seagrass species have been able to form long-lasting meadows, with meadows of the long-lived *Posidonia oceanica* dated to > 4000 years, and single clones of *Zostera marina* dated, using molecular techniques at 3000 years.

In addition to natural variability of the seagrass habitat, human intervention is becoming a major source of change to seagrass ecosystems, whether by direct physical modification of the habitat by the growing human activity in the coastal zone (e.g. boating, fishing, construction), or through their impact on the quality of waters and sediments to support seagrass growth, as well as changes in the marine food webs linked to the seagrasses.

Human impacts

Humans impact seagrass ecosystems both through direct proximal impacts, affecting seagrass meadows locally, and indirect impacts that may affect seagrass meadows far away from the sources of the disturbance. Proximal impacts include mechanical damage and damage created by the construction and maintenance of infrastructures in the coastal zone, as well as effects of eutrophication, siltation, coastal engineering and aquaculture. Indirect impacts include those from global anthropogenic changes, such as global warming, sea level rise, CO_2 and UV increase, and anthropogenic impacts on marine biodiversity, such as the large scale modification of the marine food web through fisheries. Indirect impacts are already becoming evident at present.

The most unambigious source of human impact to seagrass ecosystems is physical disturbance. This susceptibility derives from multiple causes, all linked to increasing human usage of the coastal zone for transportation, recreation and food production. The coastal zone is becoming an important focus for services to society, since about 40 % of the human population presently inhabit the coastal zone. Direct habitat destruction by land reclamation and port construction is a major source of disturbance to seagrass meadows, due to dredging and landfilling activities as well as the reduction in water transparency associated with both these activities. The construction of new ports is associated with changes in sediment transport patterns, involving both increased erosion and sediment accumulation along the adjacent coast. These changes can exert significant damage on

seagrass ecosystems kilometres away, which can be impacted by both erosion and burial associated with the changing sedimentary dynamics. The operation of the ports also entails substantial stress to the neighboring seagrass meadows, due to reduced transparency and nutrient and contaminant inputs associated with ship traffic and servicing, as well as dredging activities associated with port and navigation-channel maintenance. Rapid increase in sea-based transport, as well as recreational boating activities have led to a major increase in the number and size of ports worldwide, with a parallel increase in the combined disturbance to seagrass meadows. Ship activity also causes disturbance to seagrass through anchoring damage, which can be rather extensive at popular mooring sites, as well as fisheries operation, particularly shallow trawling and smaller-scale activities linked to fisheries, such as clam digging and use of push nets over intertidal and shallow areas and, in extreme cases, dynamite fishing. The exponential growth of aquaculture, the fastest growing food production industry has also led to impacts on seagrasses through shading and physical damage to the seagrass beds, as well as deterioration of water and sediment quality leading to seagrass loss.

The coastal zone also supports increasing infrastructure, such as pipes and cables for transport of gas, water, energy, and communications, deployment and maintenance of which also entail disturbance to adjacent seagrass meadows. The development of coastal tourism, the fastest growing industry in the world, has also led to a major transformation of the coastal zone in areas with pleasant climates. For instance, about 2/3 of the Mediterranean coastline is presently urbanized, with this fraction exceeding 75 % in the regions with the most developed tourism industry, with harbors and ports occupying 1250 km of the European Mediterranean coastline. Urbanization of the coastline often involves destruction of dunes and sand deposits, promoting beach erosion, a major problem for beach tourism. Beach erosion, however, does not only affect the emerged beach, and is usually propagated to the submarine sand colonized by seagrass, eventually causing seagrass loss. Wave break constructed to prevent beach erosion often create extensive problems, by altering long-shore sediment transport patterns, further impacting the seagrass ecosystem. Extraction of marine sand for beach replenishment is only economically feasible at the shallow depths inhabited by seagrasses, which are often impacted by these extraction activities. The threats coastal tourism poses to seagrasses are in cases direct, as in some cases of purposeful removal of seagrass from beach areas to “improve” beach conditions. Fortunate, there are symptoms that coastal tourism is moving, at least in some areas, to embrace sustainable principles, including the maintenance of ecosystem services, such as those provided by seagrasses, and could well play a role as an agent pressing for seagrass conservation in the future.

Increased nutrient inputs, causing eutrophication is a major component to seagrass loss (see below). Increased siltation of coastal waters is also a major human impact on seagrass ecosystems, which derives from changes in land use leading to increased erosion rates and silt export from watersheds. Siltation is a particularly acute problem in other regions of the world, such as SE Asian coastal waters, which receive the highest sediment delivery in the world as a result of high soil erosion rates derived from extensive deforestation and other changes in land use, and may be important in European waters adjacent to deforested watersheds. Siltation severely impacts seagrass meadows through increased light attenuation and burial, leading to seagrass loss and, where less intense siltation occurs, a decline in seagrass diversity, biomass and production.

Large-scale coastal engineering often alters circulation and salinity distributions, leading to seagrass loss. Hence, seagrass meadows, previously abundant in Dutch coastal areas, are now much reduced in surface, partially related to shifts of coastal waters from marine to brackish or freshwater regimes.

Pollution, other than that of nutrients and organic inputs, may be an additional source of human impacts on seagrass ecosystems, although, seagrass appears to be rather resistant to pollution by organic and heavy metal contaminants. These substances may possibly harm some components of the seagrass ecosystem, although such responses have not been examined to a significant extent.

Global trade has increased the mobility of marine species, whether purposely, such as aquarium specimens, or inadvertant, such as organisms carried in ballast waters. The increased human-mediated transport of species between geographically distant locations has increased the incidence of invasive species. A case affecting Mediterranean, and probably soon Eastern Pacific seagrasses, is the invasion of the Mediterranean by the tropical algal species *Caulerpa taxifolia*, which first invaded the French Mediterranean in the early 1980s, apparently released from an

aquarium, and has been reported to have expanded since along the French coast to reach the Italian and Spanish (Majorca Island) coasts. *C. taxifolia* grows rapidly and appears largely to colonize areas devoid of seagrasses, but has been reported to compete for space and resources with *Posidonia oceanica* off Monaco, being able to damage the *Posidonia oceanica* meadows. These meadows appear to be particularly vulnerable to invasion by exotic species when already under stress. The species has recently been reported in the Californian coast, raising concerns that it can also cause problems to the seagrass beds there. The Mediterranean Sea has also been invaded by *Halophila stipulacea* across the Suez Canal, but no damage to the local seagrass meadows has been reported.

Most of the impacts discussed above result from direct or indirect human intervention at the local scale. However, human activity conducive to large-scale changes at the regional or global scale also exert an important impact on seagrass ecosystems. These effects are remarkably difficult to separate from responses to background natural changes of the highly dynamic coastal ecosystem. These impacts involve the effects of the realized and predicted climate change, and result from changes in sea level, water temperature, UV irradiance, and CO_2 concentration (see below).

Natural vs. anthropogenic influences

Some cause-effect relationships between local seagrass loss and direct human activities, such as increased nutrient and organic loading, constructions on the coastline or boating activity, can be readily demonstrated. For instance, the loss of *Zostera marina* in a bay on the Atlantic coast of the USA has been shown to be closely correlated to housing development. However, the link between seagrass losses and indirect human influences is more elusive, since the coastal zone is a highly dynamic ecosystem, where many conditions vary simultaneously. Disturbances such as strong storms, hurricanes and typhoons, severely impact seagrass beds, to the point that they may be essential components of the dynamics of seagrass meadows. The difficulties of discriminating between sources of seagrass loss are best illustrated by example. The wasting disease decimated *Zostera marina* meadows in the 1930s on both sides of the Atlantic. The proximal cause for the loss seems to have been an infection by a fungus, *Labyrinthula zosterae*, although it may have affected *Zostera marina* meadows that were already stressed. Hypotheses to account for this widespread loss also point to natural changes, such as unusual seawater warming, as possible triggers for the decline. Whether indirect human impacts on global processes may have played a role remains untested.

Hence, despite clear signals of anthropogenic effects on climate components, the responses of the seagrass ecosystem are still unclear, probably due to the still modest size of the changes experienced but also, and perhaps to a greater extent, to the lack of adequate long-term monitoring systems allowing the detection of responses in the seagrass ecosystem.

Pathogens

Very little is known about seagrass pathogens and related diseases. However, some marine slime moulds of the genus *Labyrinthula* have been recognised as seagrass pathogens causing the "wasting disease". The symptoms of seagrass infections by these fungi are the presence of small dark brown or black lesions on the leaves, spreading longitudinally covering the totality of the leaf after few weeks. Usually the infections occur on mature leaves, but during severe infections young leaves may also be affected. In early 1930s *Labyrinthula zosterae* was responsible of dramatic declines of *Zostera marina* meadows at both north Atlantic coasts. Since then, large scale seagrass losses due to *Labyrinthula* infections have not been observed although scattered outbreaks of this disease continue effecting *Z. marina* meadows at local scale (Hemminga and Duarte 2000). *Labyrinthula zosterae* can also infect *Z. noltii* producing similar symptoms as in *Z. marina*. However, infected *Z. noltii* beds do not exhibit major losses. In fact, species of the genus *Labyrinthula* occur often on different seagrass beds around the world, without causing seagrass losses. It has been suggested that the interaction between seagrasses and species of slime moulds may turn detrimental for seagrasses under specific environmental conditions that make seagrasses already more vulnerable (Hemminga and Duarte 2000). This could be the case of the wasting disease in the 1930s, which occurred after a long period of high water turbidity.

The scarce reports of seagrass diseases, however, may not reflect a high resistance of seagrasses to infections but the difficulty to detect them. Information about seagrass diseases could increase in the near future with the development of new techniques able to detect microbial plant infections, as is occurring in the field of vascular terrestrial plants.

Climate extremes and climate change

Global climate changes derive, at least partly, from anthropogenic combustion of fossil fuels and changes in land use with increasing concentrations of carbon dioxide and emission of other greenhouse gasses and will most likely have substantial long term impacts on seagrass ecosystems. Climate changes of potential importance for seagrass growth and distribution include global warming, rising sea level, the increase of carbon dioxide in the atmosphere and ocean, and the increasing frequency and strength of storms. While the increase in carbon dioxide can be predicted with relatively high precision, global warming and especially its meteorological implications are more difficult to foresee. Making prediction even more complex, climate changes interact with other human-caused changes in the marine environment.

The expected increase in global temperature may have numerous effects on seagrass performance. Temperature affects almost every aspect of seagrass metabolism, growth and reproduction and also has important implications for geographic patterns of seagrass species abundance and distribution. Progressively increasing temperature may be a major threat to local populations of seagrasses, especially if living close to their low latitude borders of distribution, such as is the case for *Posidonia oceanica*, *Zostera marina* and *Cymodocea nodosa*, which encounter their southern (*Posidonia oceanica*, *Zostera marina*) or northern (*Cymodocea nodosa*) distributional limits in European waters. Seagrass distribution shift could be even greater if oceanic circulation changes in response to global warming, leading to abrupt changes in water temperature beyond those directly resulting from warming, as water masses shift at the edge of present biogeographical boundaries between seagrass floras.

While rising temperature may have major negative effects on local seagrass beds, there seems to be less reason for concern for seagrasses on the global scale. When reviewing the literature for effects of temperature on seagrasses there seems to be some bias towards the detrimental effects of high temperature and less focus on the negative impact of low temperatures. Seagrasses probably evolved in warm waters, suggested by the high species diversity of seagrass beds in the Indo-Pacific tropical region, and although a few genera, such as *Zostera*, have had great success colonizing cold temperate waters, most species grow in warm waters. It is reasonable to expect that, although there may be no apparent physiological limitations setting the latitudinal limits of seagrass distribution, increasing global temperature will increase species diversity in subtropical regions and allow cold water species to expand their geographic distribution further towards higher latitudes, thereby increasing the importance of seagrass ecosystems on the global scale.

The rise in temperature in the next 25 years will result in a projected increase in sea level of between 10 and 15 cm, mostly because of thermal expansion of the ocean and, to a lesser extent, because of melting glaciers and ice sheets. The rise in sea level may have numerous implications for circulation, tidal amplitude, current and salinity regimes, coastal erosion and water turbidity, each of which could have major negative impacts on local seagrass performance.

The projected increase in atmospheric CO_2 is expected to affect photosynthesis and growth of seagrasses, particularly increasing seagrass photosynthesis. The increasing levels of dissolved CO_2 in seawater may increase the competitive advantage of seagrasses over algae because seagrasses are currently more CO_2-limited than algae. The photosynthetic rates of light-saturated seagrass leaves are often limited by the availability of dissolved inorganic carbon, and, since the concentration of CO_2 in well-mixed, shallow coastal waters is in equilibrium with the atmosphere, the increase in atmospheric CO_2 concentration by 25 %, from 290 ppm to 360 ppm, over the 20th century, may have led to an increase in light-saturated seagrass photosynthesis by as much as 20 %. There is, however, little evidence that such physiological responses have led to observable changes in seagrass ecosystems at present.

Global warming will produce an increase in frequency and strength of storm events, resulting in increased coastal erosion and sediment resuspension with more turbid waters and poorer light conditions for benthic plant communities. Although many species of seagrasses are adapted to, and can survive, periods of low light and partial burial, storm events often reduce growth and survival and require new colonization by seeds to re-establish seagrass beds. Conversely, physical disturbance represents an energetic subsidy and may be of advantage to species diversity and improve growth conditions for climax plant species. Overall, the net effect of increasing frequency and strength of storm events on seagrasses is not clear.

The mean global sea level has risen about 10 – 25 cm over the 20th century, which should have

generated an average recession of the global coastline by 10 to 25 m , and, therefore, a large scale erosion of shallow marine sediments. There is little doubt that such changes must have affected seagrasses, which are very sensitive to sediment erosion, although there is little or no direct observational evidence for these changes. Effects of the seawater warming of 0.3 – 0.6 ^{o}C over the 20th Century on seagrasses are generally less evident than those of sea level. Temperature affects many processes that determine seagrass growth and reproduction, including photosynthesis, respiration, nutrient uptake, flowering and seed germination. Although observations of increased flowering frequency of *Posidonia oceanica* in the Mediterranean have been tentatively linked to the seawater temperature increase, there is little evidence at present to suggest any impact of increased temperature as a result of global warming at present. The temperature increase may further impact seagrass ecosystems through effects on other components, such as an increased respiratory rate of the associated microbial communities. Stimulation of microbial respiration would further enhance the problems derived from high organic inputs to seagrass sediments. Summer UV irradiance has greatly increased at high latitudes, and an increase in the north-temperate zone is also becoming evident. Increased UV levels are expected to negatively impact shallow, particularly intertidal, seagrasses.

Eutrophication

Widespread eutrophication of coastal waters derived from the excessive nutrient input to the sea, is leading to a global-scale deterioration of the quality of coastal waters, which is identified as a major loss factor for seagrass meadows worldwide. Human activity presently dominates the global nitrogen cycle, with anthropogenic fixation of atmospheric nitrogen now exceeding natural sources and anthropogenic nitrogen now dominating the reactive nitrogen pools in the atmosphere, and therefore rainwater, of industrialized and agricultural areas. Hence, anthropogenic nitrogen dominates the nitrogen inputs to watersheds, with the human domination of nitrogen fluxes being reflected in a close relationship between nitrate export rate and human population in the world's watersheds. Tertiary water-treatment plants only achieve a partial reduction in nitrogen inputs to the sea, for nitrogen inputs to the coastal zone are already dominated by direct atmospheric inputs in heavily industrialized or agriculture areas.

Although seagrass meadows are often nutrient limited, increased nutrient inputs can only be expected to enhance seagrass primary production at very moderate levels at best. Whereas seagrasses, through their low nutrient requirements for growth and their high capacity for internal nutrient recycling are well fitted to cope with low nutrient availability, other primary producers, both micro- and macroalgal, are more efficient, because of greater affinity and higher uptake rates, in using excess nutrient inputs. Coastal eutrophication promotes phytoplankton biomass, which deteriorates the underwater light climate, and the stimulation of the growth of epiphytes and opportunistic macroalgae, which further shade and suffocate seagrasses. The alleviation of nutrient limitation, together with the proliferation of phytoplankton and epiphyte biomass as a result of increased nutrient inputs imply that coastal eutrophication leads to a shift from nutrient limitation to light limitation of ecosystem production, enhanced through competitive interactions between different types of primary producers for light. These effects result in seagrass loss, particularly in the deeper portions of the meadows. The effects of overgrowth by phytoplankton, epiphytes and macroalgae may be attenuated by heavy grazing, which can buffer the negative effects of eutrophication. Eutrophication may, furthermore, have negative effects directly derived from the high resulting nutrient concentration, for high nitrate and ammonium concentrations may be toxic to seagrasses.

Whereas research on the effects of eutrophication on seagrass meadows has focussed on the effects of reduced light quality, the deterioration of the sediment conditions may also play a critical role in enhancing the loss of seagrasses. Seagrass sediments are typically rich in organic materials, due to the enhanced particle deposition and trapping under seagrass canopies compared to adjacent bare sediments. Microbial processes are, therefore, stimulated in the seagrass rhizosphere, which, if sufficiently intense lead to the depletion of oxygen and the development of bacterial communities with anaerobic metabolism, which release by-products, such as sulphide and methane, that may be toxic to seagrasses. In order to avoid such toxicity effects, seagrasses pump a significant fraction of the photosynthetic oxygen produced to the roots, which release oxygen to maintain an oxidized microlayer at the root surface. However, eutrophication reduces seagrass primary production both through shading and seagrass loss, thereby reducing the oxygen seagrass roots may release. This allows anaerobic processes and the resulting metabolites

to accumulate closer to the root surface, increasing the chances of toxic effects to seagrass. At the same time, the increased pelagic primary production leads to a greater input of organic matter to the sediments, enhancing microbial activity and the sediment oxygen deficit, which may increase the production of metabolites from anaerobic microbial metabolism. Both these processes result in the deterioration of the sediment environment to support seagrass growth, leading, through its interaction with the consequences of reduced light availability, to accelerated seagrass loss. Eutrophication effects on sediments may be more acute where sewage is the dominant source of nutrients, for this is discharged along with a high organic load, stimulating microbial activity. Aquaculture activities are becoming increasingly prominent in the shallow, sheltered coastal waters where seagrass meadows abound. Shading and high inputs of organic matter from fish cages have been shown to lead to seagrass decline below and around fish cages, through processes comparable to those of the eutrophication outlined above.

Mechanical loss

Mechanical damage is an important anthropogenic cause of seagrass decline. The removal of the plants and the damage of the shoots and rhizomes result in drastic reductions of seagrass cover. As important as the direct effects are the indirect impacts related to the alteration of the water circulation and sediment dynamics that may increase the erosion of the seagrass prairies, to the sediment re-suspension that increase water turbidity decreasing plant photosynthesis, to the mobilization of contaminants stored in the sediment, to the modification of sediment chemistry and nutrient availability or to the promotion of the competitive ability of seaweed species that may locally out compete the seagrasses.

Because seagrasses develop in sandy bottoms they are very susceptible to trawling. Drastic seagrass losses in European coastal zones, from the Northern Sea to the Mediterranean, were caused by trawling. Fishing boats harvesting fish and clams associated to the seagrass habitat, in many cases illegally have reduced severely the seagrass cover and prevent the seagrass re-establishment. Seagrasses are not physically robust and rhizomes are likely to be damaged, and seeds buried too deep to germinate, by activities such as trampling, anchoring, digging, power boat and jet-ski wash.

Figure 5.1. Propeller scars in intertidal seagrass meadows. Photo: R. Santos.

In shallow coastal zones where recreational boats are very numerous, the continuous damages related to anchoring and to propellers (Fig. 5.1) result in important vegetation losses.

The dredging of navigation channels in coastal systems such as estuaries and coastal lagoons

Figure 5.2. Channel dredging in coastal systems. Photo: R. Santos.

(Fig. 5.2) may have direct mechanical effects by removing the seagrasses, but often their indirect effects are more important. The re-suspension of the sediments or the remobilization of toxic contaminants stored in the bottom may affect seagrasses. Seagrass populations are likely to survive increased turbidity for short time periods.

However, prolonged increase in light attenuation will result in loss or damage of meadows. Toxic contaminants can decrease photosynthesis and nitrogen fixation, reducing seagrass growth. Moreover, the sliding of sediments from the channel edges to the bottom compensating the sediment removal may cause important erosion of seagrass beds that develop in the shallow banks.

Suction dredging for scallop in northern Europe removed the seagrass *Zostera* in affected areas while in un-dredged areas the seagrass remained abundant. Dredging increases fragmentation and destabilization of the seagrass meadows, which lead to reduced sedimentation and increased erosion, resulting in a decline over larger areas.

Figure 5.3. Preparing a clam culture bed over an intertidal seagrass meadow. Photo: R. Santos.

Sediment disturbance, siltation, erosion and turbidity resulting from coastal engineering have also been implicated in the decline of seagrass beds world wide. Coastal engineering, including the construction of marinas and piers or the nourishment of the beaches alter the littoral dynamics of the water circulation and sediment transport or simply bury the seagrasses. Particularly the construction of solid piers perpendicular to the shore has huge impacts on the sediment transport. Sedimentation will increase near the pier, but important coastal erosion will result downstream. In general, seagrass beds are intolerant of any activity that changes the sediment regime when the change is greater than the natural variation. A striking case of direct mechanical loss of intertidal seagrass meadows occurs in Ria Formosa coastal lagoon, southern Portugal, where 90% of the clams consumed in the country are produced. The beds for clam culture are prepared by destroying the intertidal *Zostera noltii* meadows with mechanical ploughs and covering the natural sediment with coarser terrestrial sediment (Fig. 5.3).

Conclusions

The current rate of seagrass loss illustrates the imperilled status of these ecosystems and the need for increased public awareness, expanded protective policies and active management. In order to achieve such goals it is important to focus resources to monitor seagrass habitat trends and conserve existing seagrass resources, act to attenuate the causes of seagrass loss, and develop knowledge and technologies to revert on-going seagrass decline.

The widespread loss of seagrasses is largely a consequence of the rapid growth in human activities and transformation of the coastal zone, with the resulting direct and indirect impacts on seagrasses. Global population growth is concentrated in the coastal zone, which also harbors a disproportionate fraction of the world's wealth. Indeed, some industries linked to the marine environment, such as tourism, maritime transport and aquaculture are rapidly growing. Consequently, human activity in the coastal zone is likely to continue to increase, with a potential for even greater impacts on seagrasses.

6 Have seagrass distribution and abundance changed?

Seagrasses are beneficial to coastal ecosystems in many ways and their distribution and abundance reflect coastal environmental quality. As a consequence, changes in these parameters are widely studied. This chapter provides examples of changes in the distribution and abundance of European seagrasses and demonstrates that these parameters have indeed changed markedly at both local and global scales, an overall negative trend having revealed itself during the last century. The causes are many, ranging from storms and diseases to eutrophication and coastal reclamation and construction works, but at present reduced water quality is considered the most important factor.

By Dorte Krause-Jensen (NERI), Elena Diaz Almela (IMEDEA), Alexandra H. Cunha (CCMAR) and Tina M. Greve (FBL)

Seagrass meadows may appear stable to the observer but can be highly dynamic on both shorter and longer time scales. Changes may occur locally, affecting individual shoots or patches or at the landscape scale encompassing entire estuaries and regions, and may even extend to the global level. Natural disturbances like storms or ice winters and human-induced disturbances like eutrophication, construction works or shallow fish-trawling may cause declines, the extent of which is determined by the frequency and intensity of the disturbance in question. There are indications that seagrass regression may initially occur quite slowly and that the process may then accelerate due to negative effects of reduced seagrass cover. For example, loss of seagrass cover may lead to resuspension of sediments, which reduces water transparency and thereby further reduces seagrass cover. Negative cascading effects such as these may also delay re-establishment of the seagrass meadows.

Studies on temporal changes in seagrasses may focus on different seagrass parameters depending on the scale of the study. Seasonal studies often involve a small spatial scale and focus on parameters such as shoot density or biomass while long-term studies generally include parameters like presence/absence, area cover or distribution limits.

Seagrass parameters, like shoot density and biomass, may change markedly over the year because of changes in light and temperature. Seasonal changes in seagrass parameters are smallest close to the equator and gain significance towards higher latitudes where differences between winter and summer are more pronounced. The magnitude of seasonal fluctuations in seagrass parameters also varies among species depending on their capacity for storing resources and allocating them between shoots – abilities that generally increase with plant size. The large and slow-growing *P. oceanica* can grow more independently of seasonal variations in environmental conditions and therefore has a more constant biomass over the year than the smaller and faster-growing species *Z. marina* and *C. nodosa*. When comparing seagrass data from different years it is therefore important that samplings represent the same time of year, especially in the case of small, fast-growing species.

Inter-annual changes can be caused by episodic disturbances like storms, trawling impact, ice scouring or local anoxic events. If the disturbed area is limited and the affected seagrasses are efficient in recolonisation (see chapter 3 for more details) seagrass beds may be restored within a few years. Such changes are likely to be very common but are easily overlooked if the seagrasses are not monitored regularly. Changes involving large areas or long time periods are, of course, more conspicuous and more readily detectable. The most marked examples of large-scale changes have been caused by widespread, persistent disturbances like eutrophication and coastal construction and by widespread but more transitory disturbances like a contagious disease.

The aim of this chapter is to give examples of small-scale and large-scale changes in

distribution of the 4 European seagrasses and how changes were detected. Examples of reductions as well as increases in seagrass distribution are presented, but as reductions are far most common they dominate our array of examples.

Present seagrass area distribution and recent global losses

Many of the world's shallow coastal areas are not monitored and therefore no precise estimate of the global seagrass area exists. The recently published World Atlas of Seagrasses compiles the existing information on the distribution area of seagrasses in various regions of the world and arrives at an estimated global area of 177,000 km^2, which can be considered a minimum estimate (Green and Short 2003). For Europe the minimum estimate is 6,340 km^2, but a recent minimum estimate of the distribution area of *Posidonia oceanica* meadows in the Mediterranean would increase this European estimate by 25,000 km^2. The World Atlas also provides a rough maximum estimate of the global seagrass area of 500,000 km^2, calculated as 10% of the global shallow-water area down to a depth of 40 m.

The global losses of seagrasses are difficult to quantify, as many areas are not monitored regularly. The World Atlas of Seagrasses estimates that the global loss of seagrasses from the mid-1980s to the mid-1990s is close to 12,000 km^2. A more extended analysis of the worldwide loss of seagrasses is based on an extrapolation of known losses over the last 2 decades and this analysis concludes that the global seagrass loss due to human impact amounts to 33,000 km^2 (Duarte et al. in press).

Changes in Zostera marina

The wasting disease that struck *Zostera marina* worldwide in the 1930s is the most notable natural event to have caused large-scale decline in seagrass communities. Many populations, especially along the Atlantic coasts of Europe, USA and Canada were completely eradicated (Short and Wyllie-Escheverria 1996). The disease was most likely caused by the slime mould *Labyrinthula zosterae*, and the same pathogen has also more recently caused local diseases, for example along the northeastern coasts of the USA. In fact, *Labyrinthula sp*. is a widespread, apparently benign parasite of different seagrass species, and it is likely that the slime mould only becomes pathogenic when unfavourable environmental conditions render the plants susceptible to infections.

Recolonisation after the wasting disease has not led to complete reestablishment of the former distribution and abundance of eelgrass. In many areas various kinds of anthropogenic disturbances hinder full recolonisation and cause further decline. Reduced water clarity and quality are the main reasons for loss of deep eelgrass populations and are now the most serious causes of global seagrass decline (Short and Wyllie-Escheverria 1996). Examples of long-term changes in eelgrass meadows therefore typically involve decline caused by the wasting disease, incomplete recolonisation and further decline due to eutrophication.

In the Dutch Wadden Sea the gradual disappearance of intertidal eelgrass after the mid-1960s and the failure of subtidal eelgrass beds to recover from the wasting disease have both been interpreted as responses to increased turbidity caused by eutrophication and physical activity like shell fisheries. The seagrass-covered area was estimated at 150 km^2 in 1919 but at only 5 km^2 in 1971 and 2 km^2 in 1994, when it mainly consisted of *Z. noltii* (Giesen et al. 1990). Eelgrass in the German and Danish Wadden Sea followed the same trend but still covers considerable areas.

In the Kattegat and Belt Sea, Denmark, eelgrass was also markedly affected by the wasting disease, except in the most brackish areas where the disease did not occur (Rasmussen 1977). Aerial photography of shallow Danish eelgrass populations documents that populations affected by the wasting disease exhibited a time lag of more than 10 years before substantial recolonisation began, probably reflecting long distances to seed-producing populations and extreme climatic events like storms and ice-winters during that period. After the initial time lag, eelgrass covered areas increased rapidly, and substantial recolonisation had taken place in the 1960s. Since then, the area distribution of the shallow eelgrass meadows has fluctuated markedly without displaying any trend (Frederiksen et al. 2004). The deep eelgrass meadows of Danish coastal waters have never recovered to their previous extent. While depth limits along open coasts averaged 7-8 m around 1900, they presently average 4-5 m. As a consequence of the loss of the deep meadows and reduced cover of shallow meadows, the present distribution area of eelgrass is estimated at 25% of that found in 1900 (Boström et al. 2003).

Even at the Glenan Archipelago, France, located 9 miles off the coast where direct effects of eutrophication are small, aerial photography documents that recolonisation after the wasting disease has not generated the former distribution area of eelgrass. In 1930 eelgrass covered 10 km^2 of the area but in 2000 only 4 km^2. Fishing and anchoring activity most likely contribute to limit the present distribution area (Glemaréc et al. 1997).

The Puck lagoon, Poland, may not have been affected by the wasting disease because of low salinity, and the area had abundant eelgrass meadows down to a depth of 10 m in the 1950s (Figure 6.1). These deep eelgrass meadows were almost totally displaced by filamentous brown algae and *Zanichellia palustris* during 1957-88 and marked eutrophication and pollution of the area most likely caused this change (Kruk-Dowgiallo 1991).

While there are many examples of decline in eelgrass meadows along with increased eutrophication, there are still only few examples of recolonisation upon reduction of eutrophication. For example, a general reduction in nutrient loading and a stabilisation in nutrient concentrations in Danish coastal waters over the last decade have not yet led to increased depth penetration of eelgrass. On the contrary, depth limits have continued to decrease over this period. This may be due to the still-frequent occurrences of bottom-water anoxia, the alteration in sediment conditions during the period without seagrasses rendering some areas unsuitable for seagrass growth, and the fact that recolonisation can be a lengthy process.

A more positive development has occurred in the Black Sea. Here, too, the wasting disease seriously impaired eelgrass communities in the 1930s, but the present communities are extensive, abundant and healthy. Seagrasses are most abundant in the northwestern Black Sea where eelgrass communities occupy the depth range 0.2-17 m, have an average biomass of 1-3 kg ww m^{-2} and a maximum biomass of 5 kg ww m^{-2}. The biomass and density of eelgrass in this area have increased 1.5-3 times from the early 1980s to the late 1990s, most likely due to a decrease in industrial pollution coupled with increased recreational use of the bays (Milchakova and Philllips 2003).

There are also many examples of small-scale changes in *Zostera marina*. The causes can be diverse and are not always identified. Storms are one of the major causes of small-scale changes in shallow-water eelgrass meadows and the changes differ depending on meadow density. Patchy and sparse eelgrass meadows with cover levels below 60% are generally more vulnerable and suffer greater losses during storms than do dense, uniform meadows. This is probably because dense patches possess self-protective properties which render them more stable.

Anoxic events can also induce small-scale changes. Extremely rapid changes in eelgrass area were observed in the Archipelago of Southern Funen, Denmark. The distribution area of eelgrass declined by 80% following an anoxic event during a warm summer period. But eelgrass recovered completely within 3 years through a combination of vegetative growth of surviving shoots and germination of seeds. A parallel case was observed in the Thau Lagoon in the French Mediterranean Sea, where eelgrass also died following an anoxic event, but recolonised the area within one year (Plus et al. 2003). Rapid recolonisation seems possible if the disturbance causing the seagrass decline is limited in time and space and if seedlings originating from the sediment bank or from neighbouring populations experience suitable growth conditions the following year. If the seedlings die and recolonisation must rely on spreading from neighbouring populations, the process can be very long (see chapter 3 and 12 for more details).

Changes in Zostera noltii

There are also examples of long-term changes in

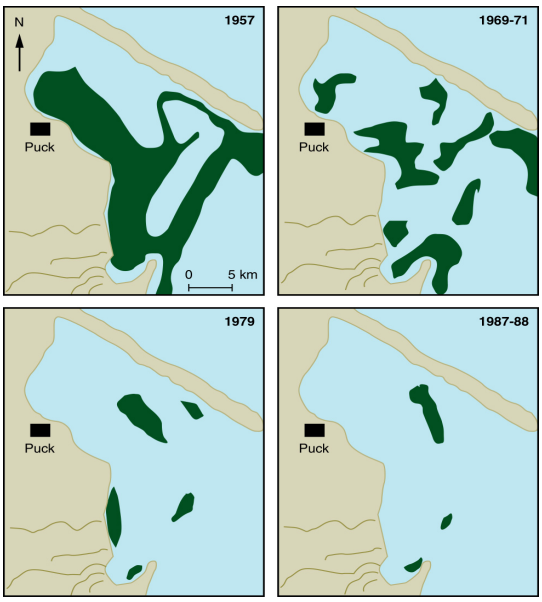

Figure 6.1. Eelgrass area distribution in Puck Lagoon, Poland: 1957, 1969-71, 1979, 1987-88 . Redrawn from Kruk-Dowgiallo 1991.

the distribution and abundance of *Z. noltii*. In Ria

Formosa in the south of Portugal, long-term changes in *Zostera noltii* meadows were assessed using aerial photography from 1940, 1980, 1989, 1996 and 1998, in a study area of 465 ha (Figure 6.2). The seagrass in the area covered by this study grows in intertidal mudflats of the lagoon and is protected by a barrier island. The island and the associated inlet have a cyclic period of migration, infilling, and inlet reopening of 40-50 years. In 1940 and 1980 the inlet was approximately in the same region of the barrier island after having completed a migrating cycle, and the area covered by *Zostera noltii* was approximately the same (6-7 ha) in the two years. Most probably the seagrass cover had changed substantially within this period but returned to its original level after completing the 40-year cycle. In 1989, after 9 additional years of growth, the seagrass-covered area had expanded to almost 8 ha as a more stable environment had evolved behind the barrier island. In 1996 the area covered by the seagrasses was about 6 times larger (~50 ha), showing that the seagrass was able to recuperate and colonise the entire area behind the barrier island in less than a decade. By this time, the inlet was at the end of the migration cycle and almost infilled, a situation which created water-quality problems interfering with recreational purposes and the clam aquaculture. Therefore, the environmental authorities decided to reopen the inlet to the west of its location in 1980. The opening caused a 50% decrease in the seagrass area, because of the direct effect of dredging and/or deposition of sand over the seagrass beds. The study is to be continued to follow the new inlet migration cycle. This case study revealed the importance of the landscape approach and the historical perspective when studying seagrass populations and it shows the importance of taking long-term changes in seagrass landscapes into consideration to be able to distinguish between man-induced effects and natural cyclic events.

As in the case of *Z. marina* there are also examples of eutrophication causing changes in *Z. noltii* meadows. In the Dutch Wadden Sea, *Z. marina* and *Z. noltii* both declined following the increase of eutrophication in the early 1960s (Giesen et al 1990). This trend was later reversed as the total area of *Z. noltii* in the area almost doubled between the early 1970s and the late 1980s after a change of management (Philippart and Dijkema 1995).

In the Mondego estuary, Portugal, where eutrophication became a dominant phenomenon in the 1980s, the main symptom has been a drastic reduction of the *Zostera noltii* meadows, parallelled by the occurrence of seasonal blooms of green macroalgae, especially in the southern, most nutrient-rich part of the estuary (Cardoso et al. 2004). Arcachon Bay, France, still has extensive meadows of the seagrass *Zostera noltii*, but as a consequence of eutrophication, massive blooms of green macroalgae have occurred since the late 1980s and constitute a potential threat to the seagrasses.

Though *Labyrinthulae sp*. can also infect *Zostera noltii* and cause disease as in *Z. marina*, no large-scale deterioration in *Z. noltii* meadows has been recorded. But in a survey of Western European populations conducted in 1989 and 1990 at least a few specimens with wasting disease-like damage patterns were found in all investigated populations. Another and totally different biological effect causing small-scale decline in *Z. noltii* was observed in the Dutch Wadden Sea where bioturbation caused by an increased density of lugworms (*Arenicola marina*) covered the small shoots with layers of sediment.

Changes in Cymodocea nodosa

Only little information exists on dynamics of *C. nodosa* meadows on small and large scales. Studies of the response of *C. nodosa* to periodical erosion and burial by migrating dunes demonstrate marked interannual changes and a high capacity of this species for recolonising bare habitats following loss of patches. For example in Alfacs Bay, Spain, subaqueous dunes maintain patches of *C. nodosa* in a continuous colonisation process and generate characteristic patchy seagrass landscapes . The time interval between passage of dunes ranged from 2 to 6 years and this time period allowed *C. nodosa* to recolonise bare substrata, produce seeds and thereby enable subsequent recolonisation (Marba and Duarte 1995).

To our knowledge, no large-scale changes in *Cymodocea nodosa* meadows have been reported in the international literature. In the Urbinu lagoon, Corsica, where *C. nodosa* co-occurs with other seagrasses, the areal distribution of the seagrasses was recently assessed based on field data, aerial photography and image processing, and the results are aimed at serving as a reference baseline for comparison with future studies in the area.

Changes in Posidonia oceanica

Posidonia oceanica, the endemic and dominant seagrass in the Mediterranean, forms extensive green belts occupying between 25,000 and 45,000 km^2 and covering about 25% of the sea bottom between 0 and 45 m in the Mediterranean basin (Pasqualini *et a*l. 1998). It is difficult to accurately assess variation in the distribution of *P. oceanica* meadows during the last decades, because information is sparse, and because production of systematic carthographies of Mediterranean coastal sea bottoms has only recently begun. Moreover, most information available on the distribution of seagrass meadows is focused on the European Mediterranean coasts, while information along the Mediterranean African coasts is sparse. However, in a bibliographic review of 46 local or regional studies on 135 surveyed meadows, 20 meadows showed extreme regression (more than 50% of the area lost), 62 showed some degree of decline, 30 appeared stable and 23 experienced some degree of progression. In e.g. the Istrian Gulf (Adriatic coasts of Yugoslavia), most *P. oceanica* meadows present in 1938 had disappeared in 1998 (Zavodnik and Jaklin 1990). Most of the reported declines were associated with direct and local human influences.

The long-term evolution of meadows around Marseille (France) has been followed, thanks to the existence of sea bottom cartographies dating back to the end of the XIX century: The *P. oceanica* beds between the "Plateau des Chevres" and the Island of Riou in the vicinity of Marseille covered around 473 ha at this time, and 449 ha in 1975, corresponding to a 5% reduction. During the period of only twelve years between 1975 and 1987 the meadows underwent marked regression, during which 44% of the meadow area was lost and the lower depth limit moved from 20 to 10 m, coincident with a strong increase in outlet of urban sewage in the area. Between 1988 and 1994, after the opening of a new sewage treatment plant, the distribution area of the meadows appears to have stabilised, and even increased locally. However, shoot density within the remaining meadow still decreased by 33%

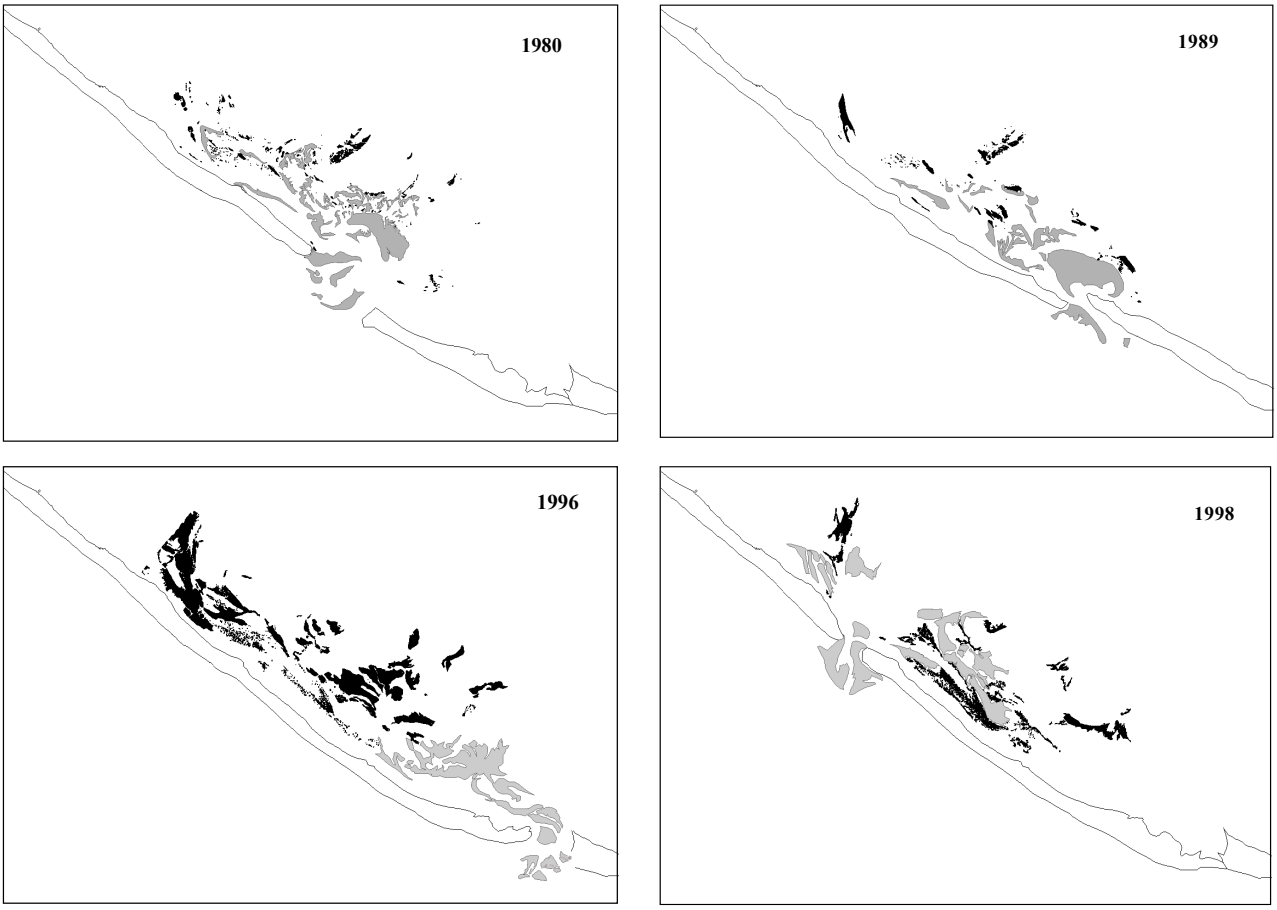

Figure 6.2. Changes in area distribution of *Z. noltii* in the Ria Formosa, Portugal as assessed by aerial photography. Black features: seagrass meadows; grey features: tidal delta. From Cunha et al (submitted).

during this period, indicating that the meadow is still subject to a declining trend. The release of nutrients from the sewage is still very high and sediment conditions may therefore have continued to deteriorate and thereby contributed to the decline in shoot density (Pergent-Martini and Pergent 1996).

In the gulf of Trieste (Italy), between Capo Circeo and Terracina, the deeper half of a large meadow occupying more than 3,000 ha has been reduced to less than 50 shoots per m^2, and the lower depth limit has regressed more than 1 km inward to the coast from 1961 to 1982. This regression of the deep meadows was caused by intensive illegal trawl fishing in the area (Ardizzone and Pelusi 1983). Such activity also markedly affected the distribution of *P. oceanica* along the coasts of Alicante (Spain). For example, a meadow of 250 ha extending along 7 km of coastline was destroyed by trawling. However, 3 years after deployment of protective artificial reefs, shoot densities had increased from 10 to 60 shoots per m^2 in some places (Sanchez-Lizaso et al. 1990).

Coastal constructions along Mediterranean coasts also produce widespread reductions of upper and lower depth limits of many meadows. Apart from the areas that are actually buried under the new constructions, meadow losses are often considerable, because these new structures may permanently alter local currents and waves, thereby increasing water turbidity, erosion and/or siltation processes. We find an illustration in the Rade de Vignettes (Toulon, France). Here, the construction of artificial beaches on top of 16 ha of reef-forming *P. oceanica* meadows in 1970-72 was followed by destruction of 199 ha of meadows, and an additional 37 ha had deteriorated due to siltation by 1978 (Nodot et al. 1978).

Natural recovery of *P. oceanica* meadows is an extremely slow process, even following small-scale disturbances. For example, during the Second World War, in 1943, a bomb dropped and exploded within a dense meadow in the Rade de Villefranche (France): a circular area 80 m in diameter was completely destroyed, while the area altered attained a diameter of 170 m. Forty years later the crater was still perfectly distinguishable although surrounded by dense and apparently healthy meadows. Many small patches have colonised the zone at an average rate of 3 new patches per ha per year, and the surrounding meadow has migrated slightly from the borders into the centre of the crater. The estimated average linear growth was only 3.4 cm per year, however, which is half the potential horizontal growth of this species. The time necessary to completely recover this small area is estimated at 120 to 150 years (Meinesz and Lefèvre 1984).

In the National Park of Cabrera Island (Spain), in a shallow area (0.8 ha) of active patch colonisation, patch settlement and space colonisation appear to have accelerated in the last 15 years. However, the process is still so slow that coalescence of patches and adjacent beds into a homogeneous meadow has been estimated to take more than 6 centuries.

At a larger spatial scale in coastal areas of Mallorca (Spain) comparison of aerial photos from 1956 and 2001 shows that the 569 ha of meadow initially present had produced 28 ha of new meadow in the 45 years, representing a gain of 5%. However, seagrass losses in the area were higher than the gains, as 81 ha were lost in the same period (Sanchez 2003).

The slow recovery rates of *P. oceanica* necessitate detection of trends in meadow dynamics before declines in seagrass cover and density become evident. Along the Spanish coasts, this was achieved by studying shoot demography, i.e. variation in shoot recruitment and mortality in meadows. The studies showed declining trends (mortality>recruitment) in 21 of 37 meadows and increasing trends (recruitment>mortality) in 7 meadows, while the remaining 9 meadows were in steady state. Among the meadows showing a declining trend, several are located far away from direct human influences. This finding suggests the existence of a background level of generalised *P. oceanica* declines possibly caused by global environmental factors, such as general deterioration of water transparency, or seawater temperature increase (both changes documented for the Mediterranean waters).

Accurate mapping of *P. oceanica* meadows as well as systematic monitoring of sensitive parameters such as shoot demography and meristematic activity throughout the Mediterranean are badly needed if conservation strategies are to be implemented. Some accurate regional meadow cartographies already exist, most of them from the North Western Mediterranean, e.g. the Gulf of Lion (Paillard et al. 1993), including the Ligurian (Bianchi and Peirano 1995) and the Catalan coasts (García et al. 2001). Good cartographies of seabottom assemblages are also available from the South-East Iberian Peninsula in the region of Murcia, and new material is being developed in other areas like the Balearic Islands. This work constitutes a useful

tool for seagrass managers, although in the case of this slow-growing species, other early warning indicators will be needed to detect seagrass decline in an early stage.

Conclusion

The examples given above illustrate that changes in seagrass meadows have indeed occurred at both smaller and larger scales. Most large-scale changes have been documented as changes in area or depth distribution and, recently, the balance between shoot recruitment and mortality has been use to predict large-scale changes in *P. oceanica*. Overall, the distribution and abundance of seagrasses have declined during the last century, and the main cause is reduced water quality.

The examples also underline that while reductions may take place rapidly, recolonisation may require long time periods, especially in the case of slow-growing species with low reproductive capacities. Once seagrass habitats have deteriorated and seagrasses are lost, restoration is therefore likely to be a process requiring a considerable amount of resources, and at worst may prove to be impossible. Efficient management must therefore focus upon maintaining existing populations through protection of habitats, and monitoring programmes should be designed to detect large-scale changes in time for protective measures to be taken.

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7 Which monitoring strategy should be choosen?

Monitoring programs have multiple roles, and there are multiple models possible. The choice of a monitoring strategy constraint, to a large extent, the resources required maintaining it but also the insights that can be derived from the information collected. This chapter discusses the role of monitoring programmes and provides an account of the development and the nature of seagrass monitoring programs worldwide.

By Carlos M. Duarte (IMEDEA), Elvira Alvarez, Antoni Grau (DGP) and Dorte Krause-Jensen (NERI)

Human coastal communities, which comprise almost half of the world's population, are intimately related to the marine ecosystem, which they are dependent upon for transportation, food and recreation, among many other services impinging on the quality of life of these communities. In many cases, coastal communities have been witnesses of the changes of their marine environments and how these changes affect the living resources and, in turn, their economies.

Volunteer monitoring programs increase awareness of the threats to the sustainability of coastal ecosystems and help citizens understand environmental problems and issues, and become involved in solving them. An important goal of any ecological monitoring program is, therefore, to educate and reach out to society by involving society in the observation of changes in the ecosystems. Volunteer-based seagrass monitoring programmes are, therefore, instrumental vehicles to increase awareness of the important role of seagrasses in the ecosystem, the threats these ecosystems are exposed to, and the importance to preserve the seagrass meadows to maintain the biological balance and the biodiversity of the coastal ecosystem. Seagrass monitoring programmes create a culture for the community's support of the protection and wise management of coastal habitats. Those involved develop a deep sense of the need for protection of their local marine environments that expands throughout the wider community. This social role is at least as important as the information these programs delivered on the status and trends of the ecosystems observed.

Overview of existing monitoring programs

Awareness of the need to monitor the health status of seagrass beds has rapidly grown over the past two decades (Fig. 7.1). This growing interest has developed in with a greater knowledge of the role seagrass meadows play in coastal ecosystems. Rapid development, involving multiple changes in land use and use practices, urban development, shoreline development, and intensified fisheries and aquculture has radically altered the inputs or materials and pressures on coastal ecosystems. Seagrass meadows, along with other sensitive coastal ecosystems, have experienced widespread decline at a rate of about 2 % year^{-1}, worldwide. Seagrass monitoring programmes were initiated to address these losses and provide information on the trends and status of seagrass meadows.

The first seagrass monitoring programmes started at the beginning of the 1980's in Australia, USA and France. In the 1990's seagrass monitoring programmes have experienced an important increase (Fig. 7.1).

At the moment more than 40 countries have developed seagrass monitoring programmes in more than 2000 meadows around the world. These monitoring programmes target 31 seagrass species around the world, most of them in

Australia where all the seagrass genera are present. Many of these programs are transnational in nature, such as:

- Cooperative Monitoring in the Baltic Marine Environment (COMBINE). Started in 1992 and includes 9 European countries.
- The Caribbean Coastal Marine Productivity Program (CARICOMP). Initiated in 1990 and including 29 institutions in 22 Caribbean countries.
- Seagrass-Watch. Started in 1999, including groups in Australia and 8 Western-Pacific countries.
- SeagrassNet. Started in 2001, including groups in Australia, USA and 11 Western-Pacific countries.

At the national level, the more extensive network is that provided by the National Estuary Program (NEP, USA). It was started in 1984 and includes all USA coastal states, 9 of which have seagrass monitoring programs.

Information on existing monitoring programs, including protocols, results, and contacts are available through multiple web pages (Table 7.1). Many of them have an open membership, such that new programs can be linked with them (e.g. SeagrassNet), thereby contributing to a wider network.

The structure and goals of monitoring programs are diverse, as some of them are conducted exclusively by scientists or technical personnel, and others, such as Seagrass-Watch, rely on volunteers. In addition, the number and nature of properties monitored also differ among them. Those based on scientific and/or technical personnel typically include a broader array of properties, some of them involving complex techniques and equipment, while other monitoring programmes measure less parameters, but across much larger areas. Only a few monitoring programs are exhaustive in detail and, at the same time, encompass broad areas, as these require vast resources. Volunteer-based networks offer added values, such as community outreach and awareness, but they require a clear leadership to be viable and sustainable over time. Experience shows that a leader with appropriate technical or scientific credentials, as well as social skills, is needed to drive and maintain the momentum of volunteer-based programs. In addition, volunteers must be motivated through the prompt delivery of results and diagnostics on the seagrass meadows monitored, as well as through activities that encourage communication and sharing of experience among the volunteer network. Web pages are useful tools to address both these actions and should be, therefore, central components of monitoting programs. Indeed, most of them do have web pages to serve these purposes (Table 7.1).

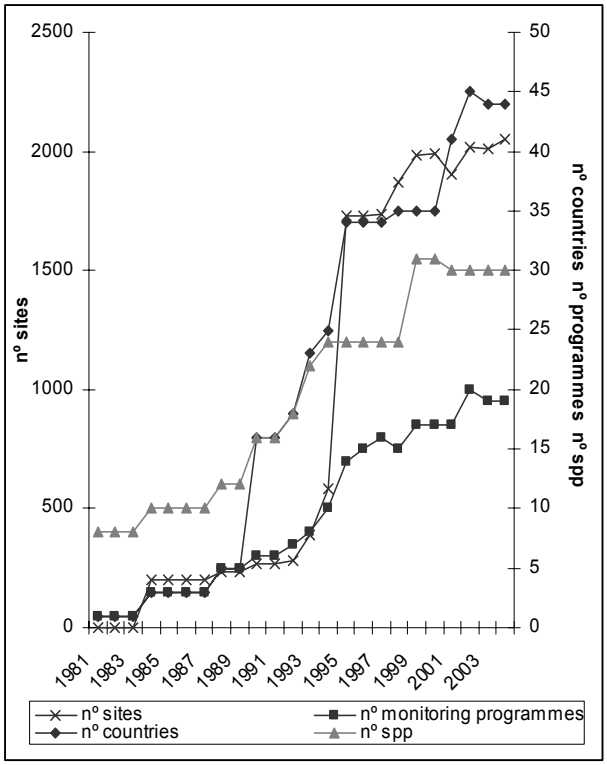

Figure 7.1. Evolution of seagrass monitoring programmes in the last two decades.

The most widely used parameters in seagrass monitoring programmes are the cover and density of seagrass meadows, as to ascertain their abundance and detect changes. The capacity to detect decline depends on the inherent uncertainty of these estimates, which can be substantial, such that only relatively abrupt changes (> 20%) can be unambiguously resolved. Direct observations of the distribution of the meadows along transects are used in many of the monitoring programmes. This method is effective in detecting declines, however, is a rather time consume procedure certainly if large areas are to be covered. For this reason, direct observations along the distributional range of large seagrass meadows become impractical. In such instances, remote sensing (optical, such as satellite or airborne remote photography; or acoustic, such as side scan sonar) is an alternative used in many of the monitoring programmes. Programs that assess changes across the entire meadows are far more effective in detecting trends than

Table 10.1 WEB-sites of existing monitoring programs

Name	WEB-site
Posidonia oceanica Monitoring Network in Balearic Islands (SPAIN)	http://lifeposidonia.caib.es
Posidonia oceanica Monitoring Network in Cataluña (SPAIN)	http://www.gencat.net/darp/faneroga.htm
Posidonia oceanica Monitoring Network in Comunidad Valenciana	http://www.ecologialitoral.com/volunt.htm
Posidonia Monitoring Network in the NW Mediterranean sea. GIS-Posidonie (FRANCE)	http://www.com.univ-mrs.fr/gisposi/
Danish National Monitoring and Assesment programme (DENMARK)	http://www.dmu.dk/forside_en.asp
Estonian Environmental Monitoring Programme-COMBINE (ESTONIA)	http://www.seiremonitor.ee/tekstid/rkp_ing/?leht=general
Cooperative Monitoring in the Baltic Marine Environment (COMBINE)	http://www.helcom.fi/Monas/CombineManual2/PartA/Introduction.htm
The Caribbean Coastal Marine Productivity Program (CARICOMP)	http://www.ccdc.org.jm/caricomp.html
Ecosistem Health Monitoring Program (Moreton Bay-AUSTRALIA)	http://www.coastal.crc.org.au/ehmp/index.html
Environment Bay of Plenty (NEW ZEALAND)	http://www.boprc.govt.nz
Environment Waikato (NEW ZEALAND)	http://www.ew.govt.nz
Seagrass Watch	http://www.reef.crc.org.au/aboutreef/coastal/seagrasswatch.html http://www.users.bigpond.com/wildlifebb/projects/seagrass/default.html http://www.dpi.qld.gov.au/home/default.html http://www.bayconnect.com.au/seagrass/default.htm http://www.ouchvolunteers.org/
SeagrassNet	http://www.seagrassnet.org/ http://www.worldseagrass.org/
North River Monitoring Program (N.Carolina-USA)	http://www.marine.unc.edu/Paerllab/research/seagrass/index.html
Florida Key National Marine Sanctuary Monitoring Program (Florida-USA)	http://www.serc.fiu.edu/seagrass/
Barnegat Bay Estuary Program (New Jersey-USA)	http://www.bbep.org/
Buzzarrds Bay Estuary Program (Massachusetts-USA)	http://www.buzzardsbay.org/
Charlotte Harbor National Estuary Program (Florida-USA)	http://www.dep.state.fl.us/coastal/activities/research/seagrass.htm http://www.charlotteharbornep.com/
Indian River Lagoon Monitoring Program (Florida-USA)	http://www.irl.sjrwmd.com http://www.sjr.state.fl.us
Long Island Sound Study (New York/Connecticut-USA)	http://www.longislandsoundstudy.net
Chesapeake Bay Monitoring Program (Maryland-USA)	http://www.chesapeakebay.net/index.cfm http://www.mdcoastalbays.org
Narraganset Bay Estuary Program (Massachusetts-USA)	http://www.nbep.org/
New Hampsire Estuaries Project (New Hampsire-USA)	http://www.state.nh.us/nhep
New York/New Jersey Harbor Estuary Program (New York/New Jersey-USA)	http://www.harborestuary.org
Peconic Estuary Program (New York-USA)	http://www.savethepeconicbays.org http://www.co.suffolk.ny.us/webtemp3.cfm?dept=6&id=980
Puget Sound Estuary Program (Washington-USA)	http://www.psat.wa.gov
Tampa Bay Estuary Program (Florida-USA)	http://www.tbep.org/
Tillamook Bay National Estuary Project (Oregon-USA)	http://www.co.tillamook.or.us/gov/estuary/tbnep/nephome.html

quadrat-based programs, which can only provide inferences on very local scales.

The health of seagrass meadows is intemately linked to the health of the wider marine environment. For this reason many programmes,

especially in USA, combine seagrass monitoring with the monitoring of water, and sometimes, sediment quality. Indeed, information on the environmental quality can help ascertain the causes of trends detected on seagrass meadows, thereby facilitating action, and, in turn, trends in seagrass health can act as alarm indicators of trends in the environment. Among the environmental properties monitored, water transparency, measured with the Secchi disc, provides the most integrative, robust and yet simple indication of water quality.

Seagrass meadows are presently legally protected in many countries, including also obligations to restore lost or deteriorated meadows. In fact legal frameworks in place in some countries establish a zero-loss policy legislation, requiring that the lost area be replaced by habitat with equivalent functional values.

The selection between different monitoring options is dependent on the structure and resources available: e.g. cumbersome methods are not practical for volunteer-based monitoring networks. The choice between protocols depends on the species monitored. For instance, shoot counts are not practical to monitor in *Zostera noltii* meadows, where small shoots occur at great densities (several thousand per m^2), and cover and density estimates are highly seasonal for most seagrass species, except for *Posidonia oceanica*, which shows very little seasonal change in shoot density, and where abrupt declines in shoot densities are, therefore, clear indication of decline. The monitoring of the upper and deep depth limits of the meadows deliver robust indications of overall status for all species, as these are easily detectable and occur where stresses are most likely: water quality affecting the deep limit and erosion or burial affecting the shallow limit. However, the monitoring of the depth limit of *Posidonia oceanica*, which reaches deepest (45 m in the clearest Mediterranean waters) among the European seagrasses, is cumbersome and requires professional diving.

Hence, the design of a new monitoring program must consider the resources available, and also the adequacy of different methods for the various species, which requires knowledge on their growth rates and basic ecology, as provided in the first half of this handbook.

8 How are seagrass distribution and abundance monitored?

Good indicators for monitoring status and changes in seagrass beds are important locally as well as globally in order to overview the extent of declines reported from many parts of the world and to identify future trends. In this chapter, a number of potential indicators of seagrass distribution and abundance is presented and evaluated to guide seagrass managers in selecting the most appropriate indicator for monitoring seagrass status and change in their area.

By Dorte Krause-Jensen (NERI), Ana Luisa Quaresma (PNRF), Alexandra H. Cunha (CCMAR) and Tina M. Greve (FBL)

Monitoring of the aquatic environment may be defined as 'the gathering of data and information on the status of water'. The purpose of monitoring varies from assessing status, detecting changes and providing early warning to detecting reasons for changes or evaluating effects of e.g. an environmental policy. Monitoring may be conducted at different scales ranging from local over regional to global scales and may involve a variety of indicators. Depending on the purpose and scale of monitoring, different monitoring strategies and indicators can be recommended (see Phillips and McRoy 1990, Bortone 2000 (part II), Short and Coles 2001).

This chapter treats three indicators of seagrass distribution: presence/absence, area distribution and distribution limits, and three indicators of seagrass abundance: cover, biomass and shoot density. It describes the way each indicator responds to human pressure and presents an overview and evaluation of the methods used to measure each indicator, without serving as a detailed method protocol, however. Finally, it evaluates the strengths and weaknesses and the potential of each indicator to forecast future seagrass status. The present evaluation of indicators is based on the requirement that a good indicator must have the potential to assess status, detect changes and help forecast future status at reasonable costs. Our evaluation therefore favours indicators that have the following characteristics:

- Predictability in its response to human pressure
- Sensitivity – i.e. low measurement error relative to the size of changes to be monitored
- Measurability through appropriate, repeatable and non-destructive methods
- Cost-efficiency
- Wide applicability – i.e. useful for many species in many types of habitat

The selected indicators are relevant for all 4 European seagrass species: *Zostera marina*, *Zostera noltii*, *Cymodocea nodosa* and *Posidonia oceanica*, and may also be applicable to other seagrass species.

Indicators of seagrass distribution

Presence/absence and area distribution

Presence/absence and area distribution of seagrasses are commonly used indicators of status and change in seagrasses at the landscape scale. Presence/absence is the simplest of all seagrass indicators. It can be measured on a coarse scale with just one observation of presence/absence in an area, or on a finer scale using a subdivision of the area into smaller units and observations of presence/absence within each sampling unit. Area distribution is closely connected with the assessment of presence/absence and may be derived using similar methods. It only requires that the areas of the seagrass-covered quadrates are measured.

Response pattern: Presence and area distribution of seagrasses may be reduced by human impacts

Table 8.1. Choosing mapping method depending on the size and the water depth of the area to be mapped (Modified from Short and Coles 2001). In-situ methods are mentioned in the table only when they can stand alone. It is implicite that remote sensing methods always require ground trouth observations. Meaning of symbols: *Video': real-time towed video camera; **'Scanner': digital airborne scanner; ': The depth intervals are only indicative as the ability of remote sensing methods to distinguish seagrasses depends on water clarity rather than absolute water depth; ": Digital aerial photos have higher sensitivity than ordinary film and is recommended when water clarity is low. Table 8.2. for more details on remote sensing methods.

Area size	Water depth	In situ methods			Remote sensing methods		
		Diver	Grab	Video*	Aerial photo	Scanner**	Satellite
Fine/micro-scale:	Intertidal	X	X		X		
<1 ha (1:100)	Shallow subtidal (<10 m)'	X	X	X	X"		
	Deep water (>10 m)	X	X	X			
Meso-scale:	Intertidal	X	X		X	X	
1 ha-1 km^2 (1:10,000)	Shallow subtidal (<10 m)'	X	X	X	X"	X	
	Deep water (>10 m)	X	X	X			
Macro-scale:	Intertidal				X	X	X
1-100 km^2 (1:250,000)	Shallow subtidal (<10 m)'			X	X"	X	X
	Deep water (>10 m)			X			
Broad-scale:	Intertidal				X	X	X
>100 km^2 (1:1000,000)	Shallow subtidal (<10 m)'			(X)	X"	X	X
	Deep water (>10 m)			(X)			

of various kinds: eutrophication, changes in land use, coastal development (including harbours), increased water-oriented activities, dredging, mariculture, etc. These impacts affect seagrasses in different ways. Eutrophication primarily increases shading because of phytoplankton blooms and increased growth of epiphytes and thereby reduces depth limits, abundance and area distribution of the seagrasses. Construction of harbours and dredging have more direct and drastic effects at least in the directly impacted areas, where all the biomass and seed pools in the sediment are lost.

Method description: Presence/absence and area distribution of seagrasses can be assessed using various methods of seagrass mapping, ranging from diver observations to remotely sensed data from satellites or airborne sensors. Table 8.1 gives a summary of available and appropriate techniques for mapping seagrasses in areas of different size and water depth.

Mapping of seagrasses by remote sensing relies on the fact that information regarding bottom features shows up as variations in the radiance directed towards the sensor. The signal received by the sensor is, however, made up of contributions from the atmosphere, the water column and the sea bottom, so the bottom signal is not always distinct. Figure 8.1 illustrates this complexity. In clear shallow waters with seagrasses occurring on a light, sandy bottom, the contours of the meadows can easily be distinguished in remotely sensed images such as aerial photos (Figure 8.2). In general, however, a wide dynamic range of colours in the image is necessary to distinguish the entire spectrum of objects from the lightest (shallow sand bottom, sun glint) to the darkest (e.g. seagrasses, mussel beds and silty bottom in deeper water).

Ground surveys, e.g. of species composition, are essential in order to interpret seagrass signatures from remotely sensed imagery and to verify the interpretation of the images, e.g. make sure that other underwater features such as macroalgae, reefs or mussel banks are not mistakenly identified as seagrass meadows. Ground surveys alone, however, are often too costly and inconvenient for mapping large coastal areas.

Large open-water areas without visible cultural features that can be used to orient the images can be problematic to map with large-scale aerial photography. In such areas, markers must be placed at known positions in the field before the photos are taken or, alternatively, satellite imagery can be used to spatially orient the large-scale photograph. It is also technically possible to log high-resolution information regarding position and orientation along with the acquisition of the images, but this technology is not generally available.

For mapping purposes alone, one mapping session is sufficient. For a monitoring programme,

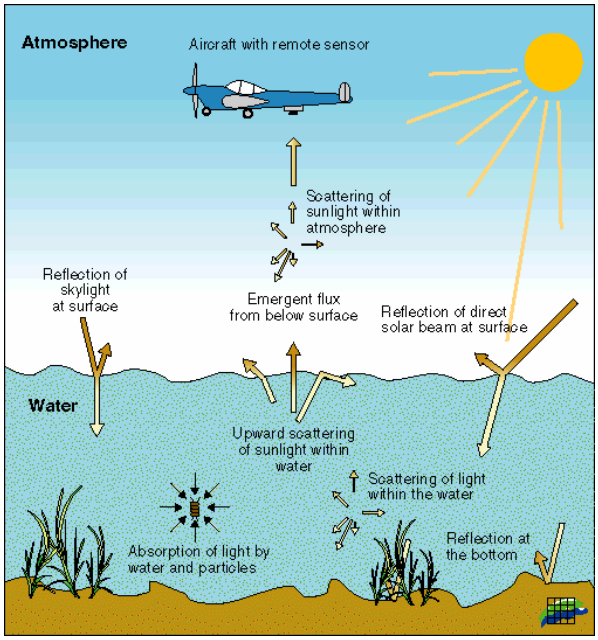

Figure 8.1. Remote sensing of bottom features. From http://www.dmu.dk/rescoman/Project/Backgrounds/challenges.htm.

a mapping frequency should be defined. The relevant frequency depends on potential impacts to the ecosystems and their health status. In highly impacted systems mapping should be done relatively often, e.g. once a year, whereas in weakly impacted systems a mapping interval of 5-10 years may be sufficient.

Method evaluation: The choice of method for mapping seagrass beds depends on the objectives of the monitoring. When the objective is to catalogue the presence/absence of seagrasses or coarsely assess the area distribution, the choice is for macro-scale maps of low resolution. By contrast, when the objective is to provide detailed data on distribution and change in seagrass areas or to estimate the biomass, the best choice is high-resolution maps. All the mapping methods mentioned have been used to map macroalgae and seagrasses in the intertidal at a scale of 2 to 20 m, but when it comes to finer scales (centimeter) no civil remote sensing technique is yet available. If a finer scale mapping is necessary to monitorize special areas, a differential GPS can be used to delineate at patch level. This method is very accurate (centimeters) but requires the use of a DGPS and the possibility to walk around the seagrass patches. All the methods are non-destructive and can be repeated over time, but prices, expertise and hardware requirements vary markedly and affect the choice of method.

Figure 8.2. Aerial photo of shallow-water seagrass landscape in Øresund, Denmark. Seagrasses patches appear as distint dark features on the sandy seabeds.

Aerial photography is the most common remote sensing method for seagrass mapping studies and for monitoring over time, while satellite data are valued for large-scale localisation investigations. Cost and accuracy varies between satellite and airborne sensors, satellites being the least accurate and least costly (Table 8.2). Usually, aerial photos are taken from aeroplanes, but a helium-inflated blimp or a kite with a standard remotely controlled 35-mm camera constitutes another method for aerial photography. This is a low-cost remote sensing technique with reasonable resolution and precision. Airborne scanners may also provide a high accuracy and this technique is becoming gradually more competitive.

Mapping seagrass beds on the basis of remotely sensed images is best achieved in clear, shallow water where seagrasses grow in dense meadows and constitute the only dark features on a sandy bottom. Turbid or deep waters limit image interpretability, and other dark features like mussel beds, stones or macroalgae may also confuse interpretation. For example, depth limits of seagrass meadows are often not visible in remotely sensed images though they may be so in images acquired by CASI scanner or in aerial photos taken on a clear day. Moreover, meadows of low density may not always be detected and the sensitivity of the mapping is therefore higher for dense meadows than for sparsely vegetated meadows. The main advantages and disadvantages of aerial photography are summarised in Table 8.3 and those of satellite imagery in Table 8.4. A guidance system for choosing remote sensing methodology is also

Table 8.2. Cost and accuracy considerations for mapping seagrass beds with satellite and airborne sensors. (Adapted from Mumby et al. 1997). * Satellite sensors.

	Landsat Thematic Mapper*	SPOT XS *	CASI (Compact Airborne Spectro-graphic Imager)	Aerial photography
Accuracy of the map (%)	<60	<50	<90	<70
Coverage per scene (km)	185 x 185	60 x 60	variable	variable
Cost (€ km^{-2} $scene^{-1}$)	0.12	0.71	8.11	16.07

available at the homepage of the project 'Rescoman' (see reference list and elsewhere in remote sensing text books).

Indicative potential: Mapping of presence/absence and area distribution of seagrasses provides the best perception of the habitat to administrators and the public. Remote sensing techniques combined with ground surveys are suitable for assessing status and change in seagrass area distribution in large areas and over long time scales. This approach provides an overview of the seagrass beds and allows assessment of status and changes in the distribution of the meadows. Mapping may also help identify conspicuous impacts on seagrass meadows like sediment redistribution and colonisation by other species. Based on information on the depth distribution range of seagrasses it is also possible to assess the maximum potential distribution area of seagrasses in a given bay and thereby evaluate the potential of the seagrasses for expanding further.

If time series of seagrass area distribution are to be used to forecast seagrass status in future water quality scenarios, relations between water quality and area distribution are needed. A relation between the maximum potential area distribution and water quality may be established based on empirical relations between water quality and depth limits combined with information on the bottom area available for seagrass colonisation at different water depths in the area in question. Historical information on seagrass area distribution under different water quality regimes in the area may also provide useful input for forecasting future distribution.

Colonisation depths

The lower colonisation depth, also known as the depth limit of seagrasses, is defined as the maximum water depth at which seagrasses grow. Depth limits can either be defined as the maximum depth of well-defined meadows or as the depth of the deepest-growing shoots.

Response pattern: The depth limit of seagrasses is primarily determined by water clarity, which, again, is closely related to nutrient levels (see chapter 4). These relationships are the basis for empirical models relating depth limits to water transparency and nutrient concentrations. Such models can be used to predict the expected average depth limit at given levels of transparency and nutrients though precise prediction is difficult due to considerable scatter in the relation (Nielsen et al. 2002). Some of this scatter may be caused by other regulating factors. New research shows a potential role of oxygen deficiency in the regulation of seagrasses, since seagrass die-backs have been documented in connection with periods of anoxia and high temperature (see chapter 4). A clear cause-and-effect relationship between seagrass die-back and anoxia still needs

Table 8.3. Advantages and disadvantages of mapping seagrasses with aerial photography (Modified after Orth et al. 1991, Short and Coles 2001).

Advantages	Disadvantages
• High spatial resolution • Spatial resolution (as determined by the scale) can be selected based on project objectives • Flexible acquisition – imagery can be planned captured at the most optimal time of day and under the best environmental conditions • Low technology information extraction – seagrass maps can be made from aerial prints or diapositives with little technical hard- or software resources, but in most cases aerial photos should be digitised and rectified before mapping. • Stereometry can greatly enhance mapping.	• Cost – The fine spatial resolution provided by the photographs comes at the cost of obtaining a large number of frames • It is produced in an analogue format and must be scanned if any computer enhancement, image processing or rectifying is anticipated • Distortion – The nature of the camera lens and position, roll, yawl and tilt of the plane introduces some distortion into the imagery. A problem if not corrected by rectifying. • Lack of light can make interpretation difficult in deep and turbid waters • Highly variable sun-glint reflection from all directions in image. • Clouds.

Table 8.4. Advantages and disadvantages of satellite imagery in mapping seagrass meadows.

Advantages	Disadvantages/potential errors
• Enables differentiation between objects whose colour would appear identical to a photo-interpreter • High spatial resolution • Digital from acquisition • Large coverage, easy to georectify	• Photographic distortion • Photo-interpretation • The spectral output of seagrass beds may vary over very short distances due to: - growth of epiphytes - "health" of the grasses - water depth - optical properties of overlying water • Low radiometric resolution • Clouds are a big problem

to be documented, however.

Method description: Colonisation depth can be determined by scuba divers swimming along a depth gradient to the maximum depth of the population. Depth limits may show considerable variation within and among sites and several subsamples within each site and coastal area are therefore recommended. Instead of having one observation per depth gradient, the diver may swim along the lower limit of the meadow perpendicularly to the depth gradient and record depth limits at several points. The diver records the depth limit using a high-precision depth recorder. The water depth at the time of observation is subsequently corrected to average water levels, for example using a locally calibrated tidal table.

Determination of colonisation depths of seagrasses should be carried out in the growth season (which depends on seagrass species and location) and preferably at the same time of the year in multi-year comparisons. This is particularly important when the depth limit is determined on the basis of individual plants or seedlings, whose limited long-term survival may cause depth limits to decline gradually over the summer.

Method evaluation: Depth limits can be estimated with relatively high precision if good depth sensors are used and if water depth is corrected depending on the tidal level at the sampling time. Other advantages are that the method is non-destructive and allows repeated measurements at the same location. It is important, however, that the term "depth limit" is well defined in the monitoring programme. It must be clear whether sampling refers to the depth limit of meadows or of individual shoots and, if the former is the case, the depth limit must be defined precisely e.g. as the maximum depth where seagrasses cover a given fraction (e.g. 10%) of the bottom.

Indicative potential: Due to its well-described relationship with water clarity and the relative ease with which it can be estimated precisely, colonisation depth is one of the best-known seagrass indicators of water quality. Moreover, in terms of assessing the general environmental status of an area, depth distribution has the advantage of being immediately comprehensible and easy to present.

The empirical relations between depth limits and water quality allow depth limits to be used to forecast seagrass status in future scenarios of water quality. The existing empirical models are well-suited for predicting average depth limits at given levels of water quality, but less suitable when it comes to predicting seagrass responses to moderate temporal changes in water quality in specific areas, due to considerable scatter in the data. Historical information on seagrass depth limits under different water quality regimes in the area may also provide useful input for forecasting depth limits upon a return to former nutrient levels.

Indicators of seagrass abundance

The abundance of seagrasses shows a characteristic depth dependence, the highest abundance typically being found at intermediate water depths where levels of exposure and light are moderate (Figure 8.3). The decline in seagrass abundance from the depth of maximum abundance towards greater water depths depends, at least partly, on light attenuation in the water column and is therefore sensitive to changes in water quality (Duarte 1991, Krause-Jensen et al. 2003).

As seagrass abundance changes markedly on an annual basis, it is important for all indicators of abundance that comparisons between years are based on samplings performed at the same time

of year – for example in August/September where

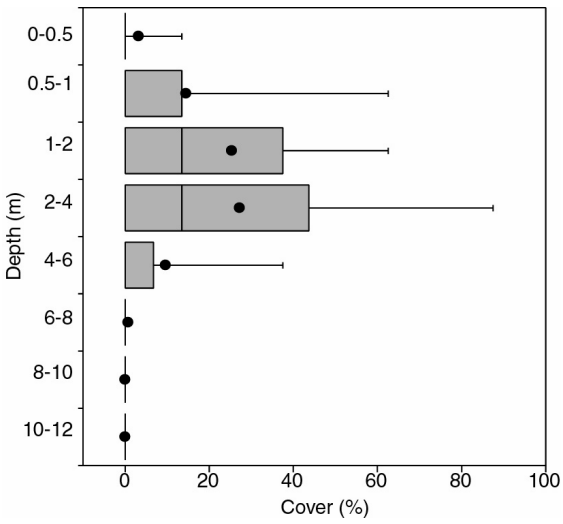

Figure 8.3. Eelgrass cover as a function of water depth. Circles show averages, vertical lines meadians, boxes 25-75% percentiles and whiskers 10-90% percentiles of cover observations. From Krause-Jensen et al. 2003. Permission to reproduce the figure has been granted by the Estuarine Research Federation.

biomass usually attains its annual maximum.

Cover

Seagrass cover describes the fraction of sea floor covered by seagrass and thereby provides a measure of seagrass abundance at specific water depths. Depending on sampling strategy, seagrass cover may reflect the patchiness of seagrass stands or the cover of seagrass within the patches – or both aspects. Measurements of cover have a long tradition in terrestial plant community ecology and are also becoming widely used in aquatic systems.

Response pattern: Light climate and exposure levels are the main factors regulating seagrass cover along depth gradients. Quantitative models show that eelgrass cover in shallow water is markedly influenced by exposure to wind and waves while cover in deeper water is mainly regulated by light. Eelgrass cover in deep water is therefore a better indicator of water quality than cover in shallow water. Similar patterns are likely to hold true for other seagrass species.

Method description: As cover is depth dependent, any measure of cover must be related to water depth. The study area can be either coarsely defined as a corridor through which the diver swims, or be more precisely defined as quadrates of a given size. Percent cover of seagrasses is usually estimated visually by a diver as the fraction of the bottom area covered by seagrass. Both shoot density and shoot length affect this estimate and, consequently, meadows consisting of dense, short shoots may have the same cover as meadows of sparser but longer shoots. The cover can be estimated directly in percent or assessed according to a cover scale. When stones constitute part of the bottom substratum it is important to define whether seagrass cover is assessed relative to the total bottom area or relative to the sandy and silty substratum where seagrasses can grow.

In the Danish national monitoring programme, eelgrass cover is assessed within corridors of about 2 m along depth gradients. A diver swims along the depth gradient and estimates percent cover at intervals of 5-10 m along the depth gradient. The diver uses underwater communication, and the regular observations of cover are recorded on the boat together with automatically logged information on position and water depth. Based on the raw data, the average cover within depth-intervals of 1 m is calculated. This method of assessing cover was found to be the most repeatable, precise and cost-efficient of several methods tested.

Method evaluation: Visual estimates of percent cover provide a simple, although rather coarse, non-destructive way of quantifying seagrass abundance. Some studies have found visual cover estimates to be sensitive to observer bias while other studies have shown these methods to be relatively robust. However, as cover estimates are based on visual observation, there is undoubtedly a risk that they may be made subjectively, and it is important, therefore, that the divers making them are trained. Training can be carried out at different levels varying from computer training programmes to training in the field.

Indicative potential: Cover estimates are coarse but well suited for surveys at the landscape level and they scale well to environmental factors such as light, wave exposure and littoral slope that operate at scales of $10\text{-}10^3$ meters.

Seagrass cover is a more sensitive indicator of eutrophication at intermediate water depths and in deep water, where light plays a major regulating role, than in shallow water, where physical exposure has a marked influence. Cover is, however, less sensitive to changes in light climate than is shoot density, because the general decline in shoot density with depth, which tends to reduce cover, is accompanied by an increase in shoot length with depth, which tends to increase cover.

Existing empirical models relating seagrass cover to water quality can be used to make coarse forecasts of seagrass cover under future water quality regimes, but precise prediction is not possible.

Biomass

Seagrass biomass is the weight (measured as fresh weight, dry weight or ash-free dry weight) of seagrasses per m^2 and thereby provides a measure of seagrass abundance along depth gradients. The measure refers to either the total biomass or the aboveground biomass of the seagrasses.

Response pattern: Seagrass biomass tends to decline exponentially from the depth of maximum abundance towards the depth limit, thus paralleling the decline in light availability with increasing depth.

Method description: As biomass is depth dependent, any measure of biomass must be related to water depth. Biomass is measured by harvesting either the aboveground or the total biomass within sampling frames. It is recommended that samples be taken randomly within stands rather than including samples from bare areas, because this sampling strategy reduces the variability of the estimates. Some sampling programmes even recommend that samples be taken randomly within the densest stands in order to reduce the variability further. The number of sub-samples and monitoring sites needed depends on the spatial variability of seagrasses in the area (see later). In the laboratory, the samples are rinsed, dried to constant weight, weighed and related to the area of the sampling frame.

Method evaluation: The method provides a relatively precise measure of seagrass abundance, and is repeatable if the sampling strategy is well defined. An intercalibration of the method applied in Øresund, Denmark underlined the importance of defining precisely where to sample within the eelgrass stands.

The method has the disadvantage of being destructive and is relatively costly, requiring sampling in the field as well as subsequent laboratory work.

Indicative potential: The indicator is useful for detailed analyses of changes in seagrass abundance. The method can also be used in connection with area distribution measures to estimate the standing stock of seagrasses in a given area. Moreover, if combined with relations between biomass and production, the method can be used to estimate total eelgrass production in a given area.

As seagrass biomass is related to water clarity, it is possible to forecast seagrass biomass in future water quality scenarios. Some quantitative models have been developed for different areas, but the available models are not as universal as those relating depth limits to water quality.

Shoot density

Shoot density is the number of seagrass shoots per m^2 and thereby provides a measure of seagrass abundance along depth gradients.

Response pattern: As shoot density is depth dependent, any measure of shoot density must be related to water depth. The maximum shoot density at given water depths shows a clearer exponential decline with depth than do biomass and cover, indicating that shoot density is regulated in a more direct and deterministic manner than the other abundance variables.

Method description: Shoot density can be measured in connection with biomass measurements by counting the number of shoots in the harvested samples before the samples are dried (see above). Shoot density can also be measured in a non-destructive manner by counting the number of shoots within given sub-areas in the field (Figure 8.4).

Method evaluation: The method provides a relatively precise measure of seagrass abundance. Counting shoots in harvested samples requires less laboratory work than processing of biomass samples but the method is still relatively time-consuming. Counting shoots in the field increases the sampling time in the field

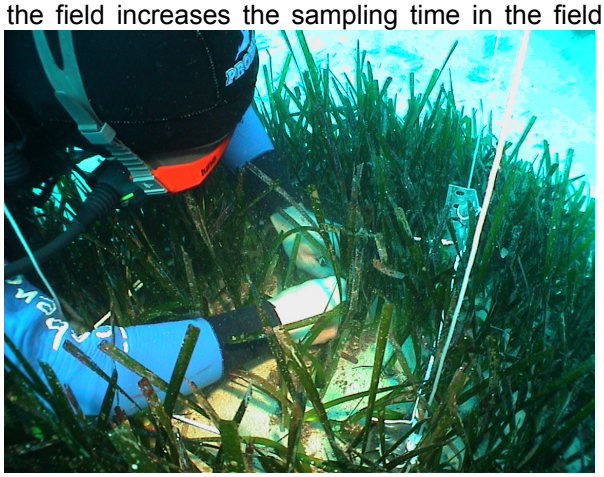

Figure 8.4. Underwater sampling of *Posidonia oceanica* using quadrats. Photo: Jaume Ferrer.

Box 8.1. Sampling strategy for eelgrass abundance - an example

An intensive and ambitious eelgrass-monitoring programme was conducted in order to assess possible effects of the construction of a bridge and tunnel across Øresund, connecting Denmark and Sweden. The method used in this monitoring programme is presented here as an example.

The sampling area was divided into an 'impact zone' close to the construction works and a 'control area' further away (Figure 8.5). The programme included 19 depth gradients of which 11 were located in the impact zone and 8 were located in the control area. 3-4 permanent monitoring sites were placed at specific depths along each depth gradient.

Sampling was conducted at the time of annual biomass maximum in August-September each year. The programme included assessment of the total eelgrass distribution area using aerial photography and image analysis. Moreover, aboveground biomass and shoot density were determined in six sub-samples of 1/16 m^2 at each site and percent cover was visually assessed at each site.

Possible effects of the construction works were evaluated by comparing data from the construction period to data from the period prior to the construction works. Data were aligned according to the so-called BACI-design (Before-After-Control-Impact), which compares the temporal trend in the impact zone with the trend in the control area.

but requires no laboratory work. As shoot density of eelgrass in shallow water may amount to about 2500 shoots per m^2, counting dense stands in the field is only feasible if small sub-areas are used.

Indicative potential: The clear exponential decline in maximum shoot density with depth suggests that shoot density responds faster than biomass and cover to changes in light climate and consequently is the more sensitive of the seagrass abundance indicators. It should therefore be possible to forecast seagrass shoot density under future water quality regimes with higher precision than cover and biomass.

Sampling strategy

Once the relevant indicators of seagrass distribution and abundance are chosen, the next step is to define a sampling strategy, i.e. choose the number of sites within the sampling area and the number of subsamples at each site and define where to place the sampling sites (see Andrew and Mapstone 1987, Short and Coles 2001). In order for the monitoring to be efficient in detecting possible changes in seagrass distribution and abundance it is important that the variability of the estimates is as low as possible. The lower the variability of the estimate, the smaller the identifiable year-to-year differences in seagrass parameters. Permanent sampling sites help reduce the variability of sampling results relative to random sampling sites. Moreover, if the sampling area contains gradients, e.g. a nutrient gradient from inner towards outer parts of an estuary, a stratified sampling design may help reduce the variability of the sampling results. A stratified sampling design infers that sampling sites are distributed within separate strata in the sampling area, e.g. in inner, central and outer parts of the estuary and that sampling results are calculated as means for each stratum rather than being calculated as means for the entire estuary. Similarly, it may be an advantage to conduct seagrass sampling within depth strata because many seagrass parameters change markedly with water depth.

The optimal number of sites and subsamples to include in a monitoring programme depends on

Figure 8.5. Sampling design for eelgrass monitoring under the "Authorities' control and mopnitoring programme for the fixed link across Øresund". Depth gradients and sampling sites are shown with black lines and dots. Redrawn from Krause-Jensen et al. 2001.

the variability in seagrass parameters in the area. In areas showing large variability within sampling sites as compared to among sites, a sampling strategy involving few sites with many subsamples each will be an advantage. In contrast, many sites with few subsamples each are appropriate if the between site variation is large relative to the within site variation.

The monitoring programme conducted in Øresund, Denmark in the late 1990s to detect possible effects of the construction of a bridge and tunnel connecting Denmark and Sweden is an example of a programme using a stratified sampling design and a dense net of sites with many subsamples (see Box 8.1.).

Conclusions

Monitoring of seagrass distribution and abundance range from coarse assessments of presence/absence or area distribution of seagrasses in large areas – based on remotely sensed data and presented as macroscale maps, to fine-scale diver assessments of depth limits and of cover, biomass or shoot density along depth gradients. These indicators all respond to changes in water quality, though with different sensitivity. The lower depth limit of seagrasses and their abundance in deep water are the indicators most directly coupled to water clarity as they are primarily light regulated. These indicators therefore have a high priority in monitoring programs aimed at assessing effects of changes in levels of eutrophication and siltation. Seagrass abundance and area distribution in shallow water are less affected by changes in light climate and more subjected to physical disturbance like wind- and wave exposure and sediment redistribution. Area distribution of entire seagrass populations therefore responds less predictably to changes in water quality than do deep populations, but distribution maps have the advantage of providing large-scale overviews of entire populations and are useful and easily eligible supplements to the more detailed monitoring.

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9 How are seagrass processes, genetics and chemical composition monitored?

Growth rates, recruitment and mortality rates, chemical and isotopic composition, genetic diversity all provide different indications about the status, susceptibility and change within a seagrass community. These indicators all require relatively much man power and some of them, in addition, require access to advanced and expensive equipment. Hence, they should only be applied where important specific questions have been clearly defined.

By Hilary Kennedy, Stathis Papadimitriou (UWB) Nuria Marbà, Carlos M. Duarte (IMEDEA) Ester Serrao and Sophie Arnauld-Haond (CCMAR)

There are many natural and anthropogenic causes that can contribute to the deterioration of seagrass meadows but it is the overall effect of any impact that is the most critical parameter to determine. However, the consequence of any impact on the seagrass ecosystem is more difficult to assess because it depends, in part, on the ability of the meadow to withstand any or further change. This aspect has to be addressed because seagrass restoration is both costly and problematic and so we must have accurate knowledge about the status of seagrass meadows and indicators of the nature and rate of any seagrass decline. Thus process indicators, which point to potential or actual differences over the course of time, are an important addition to any monitoring strategy.

In this chapter we provide an illustration of some of the process indicators that are available, to indicate the growth and demographic history of the meadow, to detect incipient or on-going change and to evaluate the ability of a population to adapt to a changing environment. Ecological, chemical and genetic indicators are used. Each technique varies in terms of type of information it provides, the facilities required, their accessibility and cost. All of these factors must be taken into account and must have consequences for their use.

Process indicators

Recruitment and mortality of shoots

The persistence of seagrass meadows depends on the flux of shoots in the population, which results from the rates of shoot recruitment and mortality, both processes highly sensitive to anthropogenic disturbance. The difference between shoot recruitment and shoot mortality rates is a direct estimate of net population growth rate, hence revealing if the meadow is expanding, declining or in steady-state, as well as providing the rate of change.

Response pattern: Seagrass shoot recruitment and mortality rates vary about 2 orders of magnitude across seagrass flora, due, to a large extent, to the large differences in clonal growth rates across species. Moreover, seagrass shoot recruitment and mortality rates vary seasonally, achieving, in *Zostera marina* beds, maximum rates during summer. Shoot mortality rates tend to increase with increasing water depth in *Cymodocea nodosa*, suggesting that it responds to light availability. Seagrass shoot mortality rate increases in response to burial. However, seagrasses are expected to respond to sediment accretion by increasing shoot recruitment, since shoots surviving burial, increase both vertical and horizontal branching rate. Seagrass shoot recruitment and mortality rates are also sensitive to sediment deterioration. For instance, shoot recruitment of *Posidonia oceanica* decreases and shoot mortality increases in carbonate sediments

with reducing conditions derived from sulphide re-oxidation.

The magnitude of the response and the sensitivity of shoot recruitment and mortality rates to environmental change also depend on the species involved (e.g. response to burial).

Because of the higher recruitment and mortality rates experienced by small, relative to large European seagrasses, meadows of small species exhibit larger changes in population abundance, and hence are more vulnerable, than those of large ones for similar relative population growth changes. Hence, seagrass population decline is easier to be detected in meadows of large, slow-growing species than in those of small, fast-growing ones. Indeed, *P. oceanica* shoot population may be in decline for decades before the meadow is lost, whereas shoot demography would be hardly able to detect declining *Zostera spp.* meadows because the changes in net population growth experienced would have already led to large-scale seagrass loss.

Method description: Seagrass shoot recruitment, mortality and net population growth rates can be assessed using retrospective techniques and repeated shoot census. The selection of the methodological approach depends on the species involved in the study, and the time scale at which shoot population dynamics is examined.

Demographic parameters can be estimated retrospectively from the age distribution of shoots. This technique, described in detail in Duarte et al. (1994), consists of harvesting and ageing at least 200 intact living shoots (i.e. clusters of standing leaves plus the attached vertical rhizome since its insertion in the horizontal rhizome) per population. For *P. oceanica* and *C. nodosa,* shoot age (in yr) is estimated by counting the number of leaf scars on the vertical rhizome plus the number of standing leaves, and then dividing the number of leaves produced by the shoot during its life by the average number of leaves produced annually by a shoot. The number of leaves annually produced by a shoot is estimated retrospectively from the seasonal changes in vertical internodal length imprinted along vertical rhizomes. Seagrasses with vertical rhizomes produce the longest vertical internodes in spring-summer, whereas the length of the internodes is the shortest in those internodes produced during winter. Because seagrasses produce one internode for each leaf produced, the number of vertical internodes (i.e. leaves) in between two consecutive maxima (or minima) in vertical internodal length equals the number of leaves produced per shoot per year. The annual shoot recruitment rate is calculated as the natural lograthm of the total number of aged shoots in the sample minus the natural lograthm of the number of shoots older than 1 yr. Shoot mortality rate is derived from the exponential decline in the abundance of living shoots with time, assuming constant mortality over shoot age classes and years.

Shoot population dynamics can also be assessed through direct shoot census in permanent plots with marked shoots. The size of the plots depends on the number of shoots expected to recruit and die in between consecutive visits. At the beginning of the study, all shoots present inside the plots are counted and tagged with a plastic cable tie placed around their vertical (for *P. oceanica* and *C. nodosa*) or horizontal (for *Zostera* species) rhizomes. During the following visits to the plots, the number of surviving shoots (i.e. tagged shoots) and newly recruited shoots (i.e. young, untagged shoots) are counted. The repeated shoot census allows direct estimates of shoot density, absolute (shoots m^{-2} yr^{-1}) and specific (yr^{-1}) rates of shoot mortality and recruitment and net population growth rates.

The specific net population growth rate (yr^{-1}) using both approaches is estimated as specific recruitment rate minus specific mortality rate. Similarly, absolute recruitment rate minus absolute mortality rate provides an estimate of absolute net population growth.

Method evaluation: Retrospective approach has been used to assess shoot demography of *Posidonia oceanica*, *Cymodocea nodosa*, *Zostera marina*, and, to some extent, *Z. noltii*. Retrospective analyses, however, provide average demographic estimates during the life span of shoot population, which, across seagrass flora, ranges from several weeks (*Z. noltii*) to decades (*P. oceanica*). Hence, assessment of seagrass shoot demography at annual scales requires to sample shoot populations several times per year in species with an average shoot life-span shorter than 1 yr. Conversely, retrospective techniques allow evaluation of the average shoot population dynamics occurred for the last 1-2 decades in meadows of long-living species such is *P. oceanica* from a single visit. Yet, retrospective examination of shoot demography of long-living species provides little information about the present shoot dynamics in the seagrass bed.

Direct shoot census is particularly useful to examine the recent shoot population dynamics in seagrass beds of long-lived species, such as *Posidonia oceanica*. For this species, an inventory of permanent plots may be made at annual or bi-

annual time intervals. Direct shoot census approach is difficult to use in *Zostera sp*. meadows, as a consequence of high shoot density, fast shoot turnover rate, and rhizome fragility.

Indicative potential: Shoot recruitment and mortality rates, assessed retrospectively in species with shoots with intermediate life-span and by direct shoot census in long-living species, allow evaluation of the present status of seagrass meadows, and detection of on-going population decline. The use of direct shoot census is particularly useful to assess recent population dynamics of slow-growing seagrass species, such as *P. oceanica*, where other approaches, such as shoot density monitoring using random quadrats, are ineffective. Therefore, the incorporation of assessment of shoot mortality and recruitment rates to monitoring programmes will help them provide the early-warning signals necessary to support management decisions to conserve seagrass meadows.

Where and when it is useful: Retrospective examination of shoot demography is useful to assess seagrass population status across broad spatial scales and that of remote meadows. Direct shoot census requires much greater diving and man-power effort than retrospective assessments, restricting their use to smaller spatial scales, unless involving participation of larger number of personnel in the study. Direct shoot census is useful to detect on-going meadow decline of slow-growing seagrass species.

Use of the indicator for forecasting: Net population growth rate allows forecasting the expansion or declining rate of the meadows when assuming that shoot mortality and recruitment rates experienced by the meadows during the study, and, hence, environmental conditions leading to present population change, will be maintained in the future. If these assumptions do not hold the indicator cannot be used for forecasting.

Leaf elongation

European seagrasses are characterized by strap-like leaves that are produced continuously by a leaf meristem ("leaf-replacing"). Leaf production is a major component of seagrass productivity, and the growth rate of the leaves of "leaf-replacing" species can be easily estimated by measuring their rate of elongation. Leaf elongation is an integrative indicator of how favourable environmental conditions are for seagrass growth.

Response pattern: Water temperature, light, and nutrient availability are the main environmental factors determining seagrass growth (Chapter 4). The first two variables have a strong seasonality in European coastal waters which drives a strong seasonal component of leaf elongation rates, with minimum values during winter time, and maximum values in summer. Additional variance in elongation rates among locations is set by differences in nutrient availability. Hypoxia and sulphide also reduce the elongation of seagrass leaves. Human-associated activities in the coastal zone can modify light and nutrient availability, and promote hypoxia in the water column and exposure of seagrasses to sulphide, environmental changes that will affect leaf elongation of seagrasses. The response of leaf elongation rates of seagrasses to environmental change will depend on the relative effects of the different components of change on seagrass growth.

Method description: Leaf elongation is estimated using leaf-marking techniques. First it is necessary to estimate the time interval between the generation of two consecutive leaves in shoot, the leaf plastochrone interval. At time 0 all leaves present in a seagrass shoot are marked by making two holes using a syringe needle or pin in the lower part of the outer leaf sheath. At time ‘t’ the marked shoot is retrieved and the number of "new" leaves (those unmarked and longer than the distance between the base of the sheath and the height at which the pair of mark holes were made) is counted. A minimum of 15-20 shoots should be marked. The leaf plastochrone interval is calculated as the number of days since marking divided by total number of new leaves produced. The length (cm) of the youngest mature leaf of each marked shoot is measured from the leaf meristem at the base of the shoot to the leaf tip (only leaves with unbroken tips should be used). Leaf elongation (cm $shoot^{-1}$ day^{-1}) is calculated as leaf length divided by the leaf plastochrone interval.

Accurate measurements require that the duration of the marking interval be longer than leaf plastochrone interval, which duration varies depending on the species (7 to 51 days for European seagrasses). If it is not possible to set a marking interval longer than the leaf plastochrone interval, leaf elongation can be also estimated by measuring the length of the new part of the leaves present at the time of marking and that of the new leaves. Marking holes are located in each leaf and the distance between them and the height corresponding to the holes marked in the sheath of the oldest leaf in the shoot (reference height) are measured. The length of the new leaves (i.e., the distance between the tip of each unmarked

leaf and the reference height) is also measured, added to the new length of the marked leaves, and divided by the marking interval (in days) to calculate leaf elongation (cm shoot^{-1} day^{-1}).

Method evaluation: Leaf marking will be difficult in locations where the sediment is silty, because resuspended sediment will decrease water transparency and when the width of the leaves is very small (some *Zostera noltii*). Except for scuba diving equipment (not essential for intertidal species if shoot marking and retrieval are performed during low tide), materials needed are rather inexpensive and easy to acquire.

Indicative potential: Several factors affect leaf growth which makes it difficult to explain differences of leaf elongation rates among locations unless the estimates are obtained simultaneously and information of environmental conditions (water temperature, light availability, and others) in each location is available/collected.

The potential of leaf elongation to evaluate growth conditions for seagrasses is higher when the locations compared are spatially close. The effect of point sources of disturbance (i.e., sewage emissaries, aquaculture facilities) can be effectively evaluated by estimating leaf elongation along the generated environmental gradients.

Nutrient and carbohydrate storage in seagrass rhizomes, and translocation between the shoots connected by the rhizome may uncouple leaf elongation rates from local environmental conditions for short (days) to medium (weeks) periods of time.

Where and when it is useful: This indicator can always be used.

Use of the indicator for forecasting: The forecasting power of this indicator is poor.

Vertical rhizome growth

Some seagrass species present vertical rhizomes in addition to horizontal ones. Vertical rhizomes keep the leaves above the sediment surface. *Posidonia oceanica* and *Cymodocea nodosa* are the only European seagrass species with vertical rhizomes. The meristem of vertical rhizomes is located at the junction between new leaves and vertical rhizome, and vertical rhizomes elongate simultaneously with production of new leaves. The maximum life span of European seagrass vertical rhizomes varies from 5-7 years in *C. nodosa* to 4 decades in *P. oceanica*.

Response pattern: The growth of seagrass vertical rhizomes is sensitive to sediment accretion and environmental conditions such as sediment quality. Seagrass vertical rhizomes respond to sediment burial by enhancing their growth, whereas they grow at minimum rates when sediment is eroded. Growth of vertical rhizomes, particularly of *P. oceanica*, declines in response to deterioration of sediment quality (e.g. enrichment of organic matter, increase of sulphate reduction rate). In addition, variability in vertical rhizome growth exhibits an important seasonal component, the longest internodes being produced in late spring-summer and the shortest ones in winter.

Method description: Seagrass vertical growth can be estimated retrospectively from the sequence of internodal lengths along vertical rhizomes. Summarizing, the sequence of vertical internodal length is measured under a dissecting microscope along 5-10 old vertical rhizomes. The seasonality imprinted on the length of vertical internodes allows calculation of annual vertical rhizome elongation, as the sum of internodal lengths between consecutive maxima or minima, and identification of annual vertical rhizome elongation across the life span of the rhizome.

Method evaluation: The method is destructive. However, one sampling visit provides estimates of seagrass vertical rhizome elongation for particular years during shoot life span (i.e., 5-7 yr for *C. nodosa*, 40 yr for *P. oceanica*). Conversely, this method does not allow elucidation of seagrass vertical growth responses at shorter than 1 year time scales, because of the seasonality in vertical rhizome internode production.

Indicative potential: Examination of the pattern (e.g. cyclical, sustained trend, discontinuities) and magnitude of interannual variability in vertical rhizome elongation allows identification of the chronological time when seagrasses have been disturbed, and provides insights on disturbance forcing. For instance, cyclical fluctuations in vertical rhizome growth reveal the passage of natural disturbances across seagrass beds often associated with sediment coastal dynamics (e.g. subaqueous dune migration, climatic change). Conversely, sustained or abrupt declines in seagrass vertical rhizome growth indicate the presence of anthropogenic effects (e.g. sediment erosion, deterioration of sediment quality) disturbing plant growth.

Where and when it is useful: This method is useful to reconstruct coastal sediment dynamics occurring at time scales within 1 yr and 7 yr if the area is colonised by *C. nodosa*, and within 1 yr and 40 yr if the area is colonised by *P. oceanica*. In addition, temporal changes in vertical growth,

particularly in *P. oceanica*, reveal the existence of antrhopogenic impacts on sediment quality, and, thus, they allow detection of coastal sediment deterioration.

This method has been useful to quantify the migration velocity of subaqueous dunes across a shallow *Cymodocea nodosa* bed, to examine the relative importance of climatic vs anthropogenic disturbances along the Spanish Mediterranean coast, and to assess the impact of fish farm activities on *P. oceanica* meadows.

Use of the indicator for forecasting: Rates of vertical rhizome elongation close to minimum values are indicative of growth cease (due to erosive conditions and/or sediment quality deterioration), and, hence, a possible increase of shoot mortality. Similarly, extremely high vertical growth rates are indicative of intense sand burial, and eventual shoot mortality. However, examination of the temporal trend of vertical rhizome growth variability will help (1) to elucidate the type, and natural/anthropogenic origin of the disturbance deteriorating the plants (and possibly the coastal zone), and (2) to evaluate the capacity of plant growth to recover based on past growth dynamics, both contributing to forecast the future of the meadow.

Chemical and isotopic composition indicators

Nutrient content of seagrass leaves

Leaf tissue contents of the macro-nutrients nitrogen (N) and phosphorous (P) are indicators of the nutrient status of seagrasses determined by nutrient availability at the locality, which in turn is, at least partly, a function of anthropogenic loading.

Response pattern: The contents of N and P in seagrass leaves are functions of the nutrient availability in sediments and water column at the growth site. Hence, leaf nutrient contents reflect the nutrient richness of the site, which in turn is determined by the background nutrient status (natural nutrient richness) and any anthropogenic nutrient loading. At low nutrient contents seagrass growth may be nutrient-limited, and at high nutrient contents the tissues accumulate nutrient stores reflecting that nutrient availability have exceeded plant nutrient requirements for growth possibly due to anthropogenic nutrient enrichment. Excess nutrient content in seagrass leaves most often has no negative impact by itself on seagrass performance.

Method description: Comparison of nutrient status is best done during the growth season (e.g. late summer when nutrient contents are at a minimum) and samples must be collected at the same time of the year and at the same depth at the different localities.

Nutrient contents are determined on dried leaf material. Standardised leaf samples (e.g. the four youngest leaves of each shoot) are collected at the selected localities. Epiphytes are scraped off the leaves with a scalpel. The leaves are dried for 24-48 hours at 60°C and homogenised by grounding. A minimum of 3 replicates is required.

Nitrogen can be measured automatically in a CHN elemental analyser or by standard manual methods for determining nitrogen contents in organic matter. Phosphorous can likewise be determined by standard manual methods. Modified techniques can be found by searching the seagrass literature.

Method evaluation: The method is a destructive and not awfully sensitive but repeatable and feasible technique to assess seagrass nutrient status of all four European species and approximate nutrient richness of sampling sites.

Apart from being dependent on environmental nutrient richness, however, tissue nutrient contents are also highly dependent on seagrass nutrient requirements for growth determined by species, season and depth. Therefore, comparisons of nutrient contents among species, season and depth must be interpreted with caution.

The analysis of nitrogen is relatively costly in terms of consumables if conducted on an elemental analyser. Manual analysis of both N and P is time consuming but in the order of 20 samples can be analysed by one person per day. To this comes handling time for sampling, cleaning and drying the leaf material.

Indicative potential: Due to the low sensitivity and high seasonal and spatial variability, the seagrass nutrient contents in leaves cannot be recommended as routine parameters within monitoring programmes.

Where and when it is useful: The indicators are useful for assessing whether low nutrient availability may limit seagrass colonisation and growth in a specific area. Nutrient limitation may occur at nitrogen contents below 1.8 % of dry weight and at phosphorus contents below 0.2 % of dry weight. However, severe nutrient limitation is rarely seen because seagrasses are relatively

slow-growing plants adapted to nutrient poor environments.

Use of the indicator for forecasting: The indicator can be applied to examine areas for differences in nutrient richness as a feasible scanning technique because seagrass nutrient contents are integrating parameters of nutrient richness. However the forecasting power of this indicator is poor.

Stable isotope composition of nitrogen and sulphur in seagrass tissues

Seagrasses are linked closely to the environmental quality of their habitat through nutrient exchange with the surrounding water. Seagrass plants can record perturbations in environmental quality in the chemical composition of their tissues with respect to nutrient elements. Nitrogen and sulphur are essential nutrients for growth, and are taken up by the plants in the form of their dissolved inorganic ions present in the aquatic environment. Nitrogen and sulphur occur naturally in different atomic forms, which are known as isotopes. The abundance of the stable isotopes of sulphur and nitrogen in seagrass tissues can be measured, providing the potential for exploratory information about the environmental status of seagrass meadows, specifically:

- in certain cases, detection of anthropogenic nutrient load in the habitat, and
- detection of sulphide penetration events into the plants through their root and rhizome tissues.

The above cases are linked because nitrogen that is derived from anthropogenic sources may have a characteristic stable isotope ratio that will be reflected in that of the aquatic plants, and may be linked to eutrophication of the water column in seagrass habitats. Prolonged eutrophication may cause insufficient oxygen supply to the benthic environment to counteract the toxic sulphide produced in the underlying sediments. This will eventually stress seagrass plants and cause irreversible decline and eventually loss.

Response pattern: The ratio of the stable isotopes of nitrogen in seagrass tissues can mirror that of the nitrogen source to seagrass habitats because of the low standing stock of dissolved inorganic nitrogen in coastal waters and its seasonal availability to aquatic plants.

The main source of sulphur to seagrasses is the pool of dissolved sulphate ions in seawater, with a relatively invariant stable sulphur isotope ratio of about +20‰. The stable sulphur isotope ratio of seagrass tissues can be expected to be close to this value within ± 2‰. Dissolved molecular oxygen is essential for the maintenance of well oxygenated roots. Insufficient root oxygenation due to low oxygen availability in seagrass habitats receiving large organic inputs can lead to development of diurnal events of root anoxia. Under such circumstances, the concentration of dissolved sulphide produced in the sediments builds up in the rhizosphere and can invade the plant through the roots. Sulphide is a potent phytotoxin, and its invasion into plant tissue is often associated with die-off of seagrasses. Sedimentary sulphide has a much lower stable sulphur isotopic composition than seawater sulphate (see above) which can extend to highly negative values, allowing detection of its presence in seagrass tissues. The stable sulphur isotope ratio of seagrass tissues, which have been affected by sulphide penetration, is lower than +20 ± 2‰ and can be as low as –15‰.

Method description: For the determination of the ratio of the stable nitrogen isotopes in seagrasses, intact shoots (n = 5) should be collected randomly from the meadow. Careful removal of epiphytes from leaves using a scalpel blade is critical to obtaining representative measurements. Analysis can be done in approximately 30 – 40 mg of finely ground material following standard isotope analytical procedures. No significant difference can be expected in the stable nitrogen isotope ratio of above- and below-ground seagrass tissues. However, for consistency, routine measurements for monitoring purposes should be done on the same part of the plant, preferably the leaves, as this method of sample collection is potentially the least destructive. Seasonal differences do occur, and sample collection may, therefore, take place twice a year, in winter and in summer, or should be consistently done within a season, e.g., in the summer when standing biomass is at its maximum. It is also suggested that, at any one time, five shoots be removed randomly from a site, and the youngest leaf analysed, resulting in one person week per sampling interval per site.

For the determination of the ratio of stable sulphur isotopes in seagrasses, intact shoots (n = 5) should be collected randomly from the meadow. Careful removal of epiphytes from leaves using a scalpel blade and thorough removal of sediment particles are critical to obtaining representative results. The ratio of stable sulphur isotopes in the roots and rhizomes can additionally be determined as these tissues are more sensitive indicators of sulphide ingress, however complete

removal of sediment particles is problematic. Analysis should be done of roots, rhizomes and leaves separately in approximately 5 to 10 mg of finely ground material following standard isotope analytical procedures.

Method evaluations: The reproducibility of measurements of the ratio of the stable nitrogen isotopes in seagrass tissue can be very high on small spatial scales up to meadow size. The variability of these measurements begins to increase on large temporal (annual) and spatial scales. Spatial variability depends on the frequency of sources and the rate of their dilution by local waters. In areas where spatial eutrophication gradients have been established (e.g. estuaries, fjords), widely different values can be found along the gradient due to different nitrogen sources and or gradual mixing with other water sources of very different signature. In these situations, the stable isotope ratio of nitrogen in seagrass tissues can still be distinguishable at different points along the gradient.

The timeframe from sample collection to analysis for the stable sulphur isotope measurements is similar to that for the stable nitrogen isotope measurements, but the analyst should beware of sulphide mineral contamination of seagrass tissues and of the need for a higher sampling frequency. This is because the within site, between sites and temporal (annual) variabilities of the stable sulphur isotope measurements can be high and comparable (up to 6‰), hence seasonal sampling will only give a measure of sulphide stress (if any) at the time of sampling.

Where and when they are useful: Measurement of the ratio of stable nitrogen isotopes in seagrass tissues has the potential to help elucidate the origin of nitrogen that enters the estuarine and coastal zone. In multi-source cases, the prerequisite is that nitrogen sources are isotopically distinct from each other. For example, open ocean, land, airborne and anthropogenic nitrogen contribute significantly to the nitrogen load in coastal waters. These sources of nitrogen can have a distinct stable nitrogen isotope ratio, such as +5 to +6‰ for deep ocean nitrate, +2 to +8‰ in groundwater influenced only by atmospheric deposition, –3 to +3‰ in fertilisers, and +10 to +20‰ in human and animal waste (sewage). Despite the complex natural cycle of nitrogen isotopes, in areas where cultural eutrophication conditions have been long standing, the ratio of stable nitrogen isotopes in seagrass plants can extend over a much wider range of values than in pristine coastal areas. However, it is pointed out that interpretation in such cases is qualitative and not quantitative, i.e., the values of the ratio of nitrogen isotopes are not related to the actual nitrogen load of the habitat.

The stable isotope ratio of sulphur in seagrass tissues can provide a proxy for sulphide invasion into seagrass plants and, hence, a detection tool of conditions potentially detrimental to the seagrass habitat.

Use of the indicators for forecasting: The stable isotope ratio of nitrogen in seagrass plants can serve only as an evaluator of nitrogen sources to the habitat and not as a forecasting tool of its status and fate. The stable isotope ratio of sulphur in seagrass tissues offers information primarily about the source of this nutrient to the seagrass plants. Shifts in the stable isotope ratio of sulphur in seagrass tissues are linked directly to processes that can be detrimental to the health status of seagrass meadows (i.e., toxic sulphide invasion into the plants). Sulphur isotopic measurements can therefore contribute to forecasting the health status and fate of seagrass habitat.

Diversity-indicators

Genetic diversity

Genetic diversity is a fundamental component of biodiversity, forming the basis of species and ecosystem diversity. Since species are composed of populations that exist somewhat independently of each other, genetic diversity exists both within and among populations of each species. Seagrass habitat degradation may cause loss of genetic diversity, consequently lowering the potential for populations to survive and to adapt to changing environmental conditions.

Response pattern: Measurement of genetic diversity with molecular markers that are not under selection can be used to understand not only the genetic composition of each population but also at what scales there is dispersal of seeds and pollen, and what is a management unit (population genetically differentiated from others). Characterisation of the geographic differentiation of populations is critical for understanding whether meadow recovery can rely on the neighbouring populations as sources for recovery, or to choose source populations for restoration. A recent reduction in genetic diversity or in gene flow between populations can be indicative of deteriorating environmental conditions.

Method descriptions: Sample selection: A pilot study is necessary to be able to estimate the

number of samples needed for a representative sample of the population but it is often suggested that 50 samples per population are sufficient to give a reliable estimate of the genotypic diversity.

From the samples DNA is extracted. From every sample genetic markers are used to determine the genotype of the sample at each locus (i.e., determining the multilocus genotype). The genetic markers yielding the best resolution to identify distinct clones on the basis of their multilocus genotypes are usually microsatellite loci, because they are hypervariable markers (i.e. show more distinct alleles in the population than other markers). Having this data, various population genetic parameters are estimated (genetic diversity, population differentiation and gene flow at various scales), for which a variety of software packages are available.

Since seagrasses are clonal organisms, low resolution of genetic markers can lead to the erroneous conclusion that the population is highly dependent on clonal propagation rather than sexual reproduction, just because the markers cannot distinguish genotypes. So it is essential to choose markers that are capable of distinguishing different genotypes as opposed to clonal repeats.

Evaluation of the method: Microsatellite markers are very sensitive for determination of genotypic diversity and its spatial and temporal variability. However this requires a large number of polymorphic microsatelite loci to be used, and therefore high costs. The feasibility of the method is limited by the requirement of rather specialised expertise and equipment. It is the most repeatable PCR-based technique for nuclear DNA genetic studies. The method is destructive in the sense that cells must be obtained (and therefore destroyed) from the organism, but it is non-destructive for the whole organism, because only a small portion of material is sufficient for the analysis of the genotype.

- Development of microsatellites is necessary for each species, and it is expensive. However microsatellites have already been developed for all four European seagrass species and primers are available in the literature. The method can be used for all habitats (as long as sampling of seagrass material is possible).

Indicative potential: This indicator provides an assessment of the current status of genetic diversity of any seagrass species from any habitat, but this current status is the consequence of long term past events and dynamics. Its interpretation is difficult because it is influenced by so many factors. It is not a general early warning indicator, but under certain conditions changes can take place in this indicator before other warning signs can be detected, such as restriction to gene flow due to habitat fragmentation. Due to the high cost in equipment, consumables and man-power, and the high level of specialised training that this indicator requires, it is not recommended as a general use indicator, but only when more thorough population studies are needed/possible.

Where and when it is useful: This indicator is useful in any meadow at any time, depending on what population parameters are required to be understood. For example, within-population genetic diversity (e.g., if it has been reduced or increased following some perturbation), or scales of gene flow (e.g., if barriers to dispersal have been created).

Use of the indicator for forecasting: The forecasting power of the indicator is poor in terms of short term predictions, because the genetic composition reflects events that took place over a long time scale, and the factors shaping it may have changed considerably for the time period for which one is attempting to make predictions. It is more an indicator of the past history. However, understanding past history can certainly be helpful in predicting what will happen if the same factors/events that took place in the past will continue to occur.

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10 How is seagrass habitat quality monitored?

Seagrass monitoring programmes can benefit from including variables on habitat quality in addition to seagrass indicators. Such variables affect the seagrasses, and information on their level may therefore help identify reasons for status and changes in seagrass indicators and suggest corrective measures.

By Dorte Krause-Jensen (NERI), Nuria Marbà (IMEDEA), Elvira Alvarez Pérez and Antoni Grau (DGP)

A wide range of variables of natural and human-induced origin affect the status of seagrasses. The variables include water- and sediment quality, climatic variables and biological interaction. Information on such variables may therefore help interpret results of seagrass monitoring by identifying probable causes of a given status or change in seagrass parameters. Moreover, such information may be used to isolate the effects of human pressure from those of natural variations in climate etc. and thereby help suggest corrective measures.

Water quality and climatic variables

Nutrient concentrations and light attenuation in the water column are the most important water quality parameters affecting seagrass growth. Another habitat characteristic, salinity, may also play a role (see chapter 4). These variables therefore constitute the primary list of variables to measure in connection with seagrass monitoring programmes:

- Light attenuation – can be measured simply by using a Secchi disc or more precisely using a light meter to measure actual light levels at different positions in the water column and then calculate the light attenuation per meter water column.
- Nutrient concentrations – inorganic concentrations are often low and difficult to detect in summer so it may be a better choice to measure inorganic nutrient concentrations in winter and/or total nutrient concentrations in summer.
- Salinity – can e.g. be measured automatically using a probe or manually using a refractometer.

More specific water quality variables can also be useful in special cases. Examples are listed below:

- Concentrations of particulate and dissolved organic matter – relevant if details on the main light attenuating variables are needed.
- Concentrations of O_2 and H_2S - relevant in connection with detailed analyses of reasons for sudden seagrass diebacks.

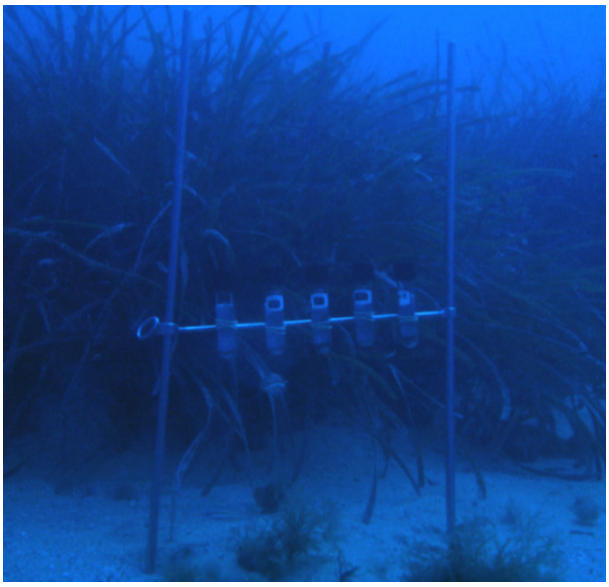

Figure 10.1. Benthic sediment trap deployed within a *Posidonia oceanica* bed. Photo: Carlos M. Duarte.

- Concentrations of toxic substances – relevant in case of suspicion of seagrass diebacks due to toxic substances.

Due to large temporal variability in water quality parameters, they must be measured frequently. It is not sufficient to measure them along with an annual seagrass sampling. In a routine monitoring programme, water quality can be measured at a site placed centrally in the bay or coastal lagoon of interest and representing the general water quality of the area. In a more detailed monitoring programme aiming at identifying spatial gradients in water quality and seagrass indicators within a given bay, more sampling sites are obviously needed.

Several climatic variables may also affect seagrass growth (see chapter 4) and may therefore provide useful information in the interpretation of monitoring results. The most relevant climatic variables to include are:

- Water temperature
- Rainfall or freshwater run-off
- Insolation
- Wind velocity/direction

Water temperature is usually measured along with the measurements of water quality while data on rainfall, insolation and wind may be obtained from meteorological institutes.

The classification of ecological status under the Water Framework Directive (WFD) is one example of a monitoring strategy that includes hydromorphological and physicochemical elements to support the biological elements.

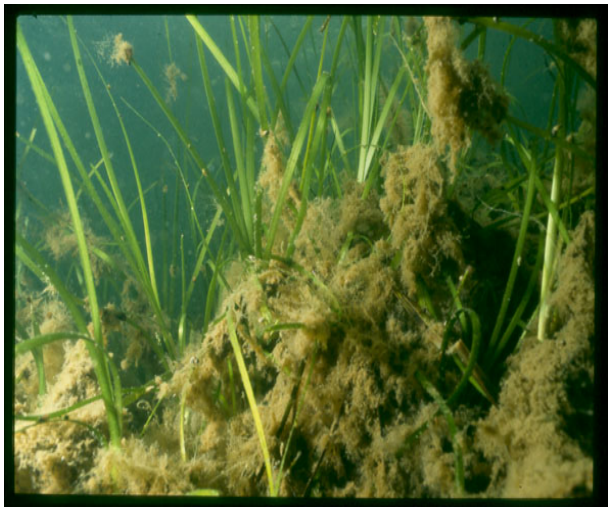

Figure 10.2. Epiphytes and macroalgal blooms covering *Zostera marina*. Photo: Michael Bo Rasmussen.

Another example is the 'Helcom Combine Programme' for monitoring eutrophication and its effects in the Baltic Sea. The national Danish monitoring programme also includes both biological indicators, water quality variables and climatic variables. The programme attempts to correct the biological indicators for climatic influence in order to be able to relate trends in the biological variables more directly to changes in water quality. The idea is that e.g. an unusually sunny summer may have a positive effect on the vegetation indicators that may be misinterpreted as an indication of improved water quality. Analyses of climatic influence would allow correcting for this effect of climate so that any changes in biological indicators can be related more directly to changes in water quality.

Sedimentation

Human activities in the littoral zone increase the inputs of organic matter to the sediment and the growth and survival of seagrasses decrease as this input increases. Seagrasses growing on carbonate-rich, and iron deficient, sediments are particularly sensitive to sediment organic matter enrichment. In addition, high inorganic sediment accretion enhances vertical rhizome growth, in those species that possess them. Hence, knowledge on sedimentation rate of total and organic suspended particles will help to assess the status of seagrass meadows.

The rate of suspended particle deposition on seagrass sediments can be measured by deploying benthic sediment traps. There is a wealth of sediment trap designs; one is illustrated in Figure 10.1. In this model, each trap is built by a stainless steel bar supporting five cylindrical glass centrifugation tubes. The benthic sediment

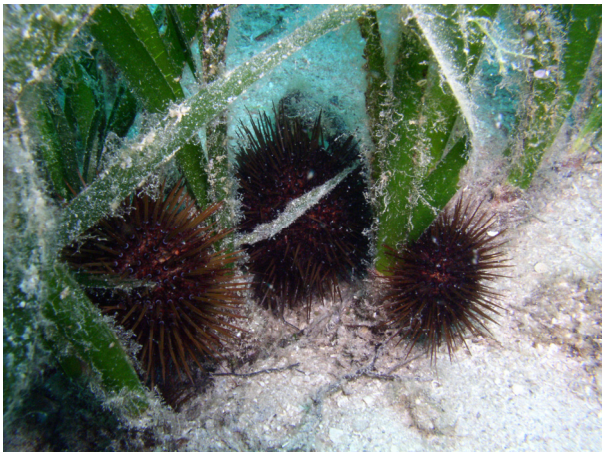

Figure 10.3. Seaurchins in seagrasses. Photo: Elvira Alvarez Pérez.

traps are placed at 20 cm above the sediment surface by SCUBA divers. Details on sampling methods can be found in e.g. Gacia et al. (2003)

Epiphytes and macroalgal blooms

Epiphytes and (particularly green) macroalgal blooms may be a prominent component of seagrass ecosystems when ambient nutrient concentrations are high (Figure 10.2). Abundance and species composition of seagrass epiphyte communities also depend on seagrass leaf life-span, which is species-specific (see chapter 3) and determines the age of the epiphyte substrate. Hence, macroalgal and epiphyte abundance and species composition in seagrass meadows, after taking into account the seagrass species involved, can be used as a proxy of nutrient richness in coastal waters. In addition, the presence of green macroalgae of the genus *Caulerpa*, which grow on organic rich sediments e.g in the Mediterranean, or of the widespread genera *Ulva* and *Enteromorpha* which also thrive under nutrient rich conditions can be used as an indicator of deterioration of sediment quality for seagrass growth. Macroalgal blooms and epiphytic biomass may vary markedly over time both because the organisms grow fast and because epiphyte biomass is regulated by wind exposure and can be decimated after a storm. When these organisms are included in monitoring programmes, sampling must therefore be repeated several times during the growth season to represent the site properly. This render sampling of epiphytes and macroalgal blooms costly. Details on methods for sampling epiphytes can be found in e.g. Borum (1985).

The abundance of macroalgal blooms can be measured either as cover or as biomass using the same methods as described for seagrasses (see Chapter 8).

Figure 10.4. Pinna (*Pinna nobilis*) in seagrass bed. Photo: Carlos M. Duarte.

Key fauna

Seagrass meadows host a large number of animal species. Information on key fauna associated to the seagrass beds can be directly relevant for the interpretation of seagrass monitoring results in cases where the fauna grazes the seagrasses. Moreover, the fauna species associated with seagrass meadows often reflect plant health and may also add to the general understanding of the importance of seagrass beds for coastal biodiversity.

Relevant key fauna to measure in connection with seagrass monitoring programmes are listed below:

- Sea urchins – are often important grazers of seagrasses (Figure 10.3). Herbivory by sea urchin occasionally (overgrazing events) can be so intense that it may even result in the elimination of extensive seagrass patches. The density of sea urchins increases with increasing nutrient concentration in plant tissues, and, hence, in the environment. An increased grazing activity by sea urchin has for example been observed in *P. oceanica* meadows situated under fish cages. Sea urchins can be sampled with underwater visual census.
- Fish – there are fish species that are “permanent residents” in the seagrass meadows. Examples are pipefish and sea sticklebacks and in the Mediterranean also *Sarpa salpa* (L.). Exclusive feeding on living seagrass leaves is a rare feature; in general fishes that feed fresh leaves also depend on other food resources, such as epiphytes or small invertebrates. Fish can be sampled with underwater visual census.
- Molluscs – some big species, like the Mediterranean bivalve *Pinna nobilis*, are exclusively dependent on seagrasses (Figure 10.4), and are therefore affected by physical impacts on the meadows, e.g. boat anchoring. Presence of *P. nobilis* is a characteristic of healthy seagrass meadows. *Pinna nobilis* can be sampled with underwater visual census. Some snails (e.g. the genus *Rissoa*) are also frequent on seagrasses.
- Birds – can be major seagrass consumers in the intertidal zone, e.g. the mid-west

European coastal areas, with *Zostera marina* and *Z. noltii* populations are wintering areas of some species of birds: brent goose, pintail, widgeon and mallard. Swans also graze on seagrasses. The abundance of birds can be assessed by population density surveys.

The underwater visual census is a quantitative estimation of the abundance of fishes and large epibenthic invertebrates by transects in clear water environments. There are other techniques available for assessing the abundance and biomass of fishes and epibenthic invertebrates, such as gill nets, drop nets, etc., in turbid waters.

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11 What can be done to prevent seagrass loss?

The presence and distribution of seagrass beds depend on many complicated environmental factors and conditions. Nevertheless, the main factors responsible for loss of seagrass beds can often be clearly identified and addressed in the management of coastal zones. Efficient remedial actions may sometimes involve high-cost regulation of land use but in other instances can be based on simple, low-cost means or be taken into account during planning and implementation of human activities.

By Jens Borum, Tina M. Greve, Thomas Binzer (FBL) and Rui Santos (CCMAR)

The most important factors for seagrass growth and distribution have been identified as: water column light conditions affected by nutrient loading and siltation, water column and sediment oxygen conditions affected by organic loading, chemical pollutants in the form of e.g. pesticides and antifouling agents and, finally, physical disturbance generated by coastal constructions, fisheries, boating, clam digging, etc. (see chapter 4 and 6). In addition to these common disturbances, local conflicts between human activities and seagrass conservation may arise in connection with e.g. cooling water from power plants or high salinity water from desalination plants. At the other end of the spatial scale, global climate changes may in the future have marked influence on the abundance of seagrasses.

A wide range of management tools are available to prevent or reverse seagrass loss, but their efficiency and costs vary substantially, and remedial actions must be selected depending on the nature, source and strength of the human disturbance causing the loss of seagrass beds.

Nutrient loading

Nitrogen and phosphorous are the most important nutrients regulating planktonic algae, and hence water transparency and light conditions for seagrasses. Nutrients also stimulate growth of algae living on seagrass leaves causing additional shading. Nitrogen and phosphorus derive from a variety of sources, some of which are of regional importance, others of local importance. The most important sources of nutrient loading to coastal areas are urban sewage outlets, industrial outlets, runoff from fertilized agricultural areas and atmospheric deposition of nutrients originating from agriculture and fossil fuels. Fish farms, small point sources from industries, houses and from boats are often minor contributors to nutrient loading but may be of large local importance in lagoons or embayments with low water exchange.

Nutrient loading is beyond comparison the most important factor responsible for deterioration of seagrass beds on national and regional scales. As an example, the effects of anthropogenic nutrient loading from urban sewage and agricultural runoff can be traced in all marine waters in the Baltic along the coast of northern Germany, Denmark, Sweden, southern Norway and even in the open part of the North Sea. This nutrient loading has resulted in a decline of seagrass depth penetration and area distribution of 40-50% within the last century. Hence, it is obvious to address nutrient loading as a first priority issue in coastal management.

Efficient remedial actions against regional nutrient loading may require high-cost intervention against sewage disposal, agricultural runoff and the use of fossil fuels. Urban and industrial sewage must be diverted to sewage treatment plants with efficient means of nutrient removal (Fig. 11.1). The techniques are well known but investments and running costs are high. Similarly, efficient reduction of agricultural runoff requires major, integrated remedial actions with limits on the use of fertilizers and restrictions on land use in catchment areas. As an example, narrow zones of uncultivated soils along streams and rivers together with undisturbed wetlands have significant potentials for intercepting nutrient runoff before reaching the water and, therefore, such zones should be established and protected through legislation. Such remedial actions are

being implemented in several North European countries. Marsh areas similarly function as extensive and important buffer zones with substantial capacity to capture nutrients, and marsh areas must be protected against disturbance and exploitation.

Smaller point sources with nutrient loading to lagoons and embayments can be addressed by legislation. Fish farms, industrial outlets and sewage disposal should not be allowed in lagoons and embayments but must be re-located to areas with more efficient water exchange or preferably be diverted to sewage treatment plants. Even sewage from small boats may constitute a problem in popular anchoring sites, especially in nutrient poor Mediterranean embayments, and should be avoided through legislation on sewage containment aggregates and by implementing codes of conduct.

Loading of organic matter

Inputs of organic matter consume oxygen in the water column and sediment. Seagrasses need oxygen for respiration in leaves and roots and as a protection against invasion of toxic compounds from the sediment (see chapter 4). During daytime the plants produce oxygen by photosynthesis, but in the dark, oxygen is supplied from the water to the leaves. Therefore, poor oxygen conditions due to organic loading are important stress factors for seagrass growth and survival.

The sources of organic loading are often the same as those of nutrient loading but the impact of the organic loading is more local and often less severe. Sewage treatment plants may efficiently remove organic wastes at relatively low cost by simple mechanical treatment means (filtration or sedimentation tanks). However, outlets of untreated sewage are still very common in some contries, even from larger cities and industries, but most untreated sewage originates from numerous small settlements and smaller industries. Where sewage treatment cannot be applied, remedial actions may be taken to ensure that outlets are positioned outside the seagrass beds and in areas with high water exchange to ensure efficient dilution of the organic matter.

Figure 11.1. Negative impacts of nutrients, organic matter and industrial wastes on coastal waters, including seagrass beds, can be substantially reduced by treating waste water in advanced sewage treatment plants (with permission from Lynettefællesskabet I/S).

Fish farms constitute a local but very substantial source of organic matter because of inefficient utilization of the fish feeds. The sea floor below fish farms receives large amounts of organic matter and as a general rule the farms should be placed outside areas with seagrass beds. If this is not possible, addition of iron to the sea floor or to the fish feeds may be considered as possible means to reduce the negative impact on seagrass beds, because iron has a positive effect on plant performance and survival. Organic loading from small boats may locally, in areas with low current and water exchange, have a substantial negative impact on seagrass beds.

Siltation

Siltation is the process where fine silt particles (mud and clay) originating from land or from the sea floor are suspended in the water column creating turbid water and poor light conditions for seagrass growth. Hence, the consequences of siltation are almost the same as those of nutrient loading. In addition, the silt may settle on the seagrass leaves, and at very high siltation rates, the plants may even be buried.

Siltation originating from land is the result of land use (agriculture or deforestation) or construction activities (e.g. road construction) in the catchment areas or coastal zone. Siltation may, however, also arise, at least temporarily, during human disturbance of the sea floor in connection with dredging and sand reclamation.

Agriculture, forestry and construction activities in coastal areas must be planned and conducted taking actions to minimize siltation to streams and rivers or directly to coastal waters. For agriculture, uncultured buffer zones along streams and rivers can, like for nutrients, function as efficient filters for runoff of silt. With respect to deforestation there are well-known procedures such as strip cutting to reduce soil erosion and subsequent siltation from forests growing on high slope

grounds. In contrast to the land based siltation sources, it is impossible to avoid siltation in connection with dredging, sand reclamation and marine constructions. However, sand reclamation can be conducted in areas outside the seagrass beds and dredging can be conducted with equipment minimizing silt loss. In addition, dredging and sand reclamation can be restricted to short, intensive periods as seagrasses can overcome shorter periods (days) with poor light conditions without major problems.

Mechanical disturbance and coastal constructions

Direct mechanical disturbance and uprooting of seagrasses have long-term impacts on seagrass beds because seagrasses are in general slow-growing plants requiring long periods for recolonization. Mechanical disturbance is caused by dredging, sand reclamation, land reclamation for agriculture or clam culture and by trawling within the seagrass beds. Anchoring and boat propels may also give rise to scars in the stands. Coastal constructions, such as bridges and piers, directly on seagrass beds are obvious and fatal mechanical disturbances but even small scale disturbances such as clam digging inside intertidal seagrass beds may have substantial negative impacts on seagrass growth and survival.

Dredging is often necessary to ensure boat access, but dredging channels should be placed outside the most important seagrass beds if possible (Fig. 11.2). Similar considerations can be taken when constructing bridges and piers although public interest in infrastructure often will have first priority. In contrast to this, sand reclamation within seagrass beds can easily be prohibited by legislation without any socioeconomic consequences.

Figure 11.2. Dredging and sand reclamation from shallow coastal areas with seagrass beds has major detrimental impacts due to direct destroyment of the beds and due to resuspension of sediments and reduction of the light conditions for plant growth. Photo: Rui Santos

Disturbances caused by boating, anchoring, trawling and clam digging may similarly be controlled through legislation, local regulations or codes of conduct based on awareness. Damage caused by anchoring may seem to be a minor problem but in popular anchoring sites, e.g. with beds of the slow-growing *Posidonia*, anchoring scars are a major problem with long-term consequences. In such areas permanent anchoring buoys can be established and damage to the seagrass bed completely avoided. Trawling and clam digging activities should to the widest possible extent be held outside the seagrass beds.

Heat and salinity stress

Like all other marine organisms, seagrasses are adapted to certain temperature and salinity regimes and anomalies compared to these regimes may result in decline of plant performance or even mortality.

Salinity anomalies in the form of too low or too high salinity may occur when water exchange between lagoons and open sea is regulated through dam construction or floodgate control. There are several examples from lagoons adjacent to the Wadden Sea and the North Sea of how control of water exchange has reduced salinity to an extent where seagrasses are not able to survive, and large seagrass beds have been destroyed. In areas with low precipitation and low freshwater runoff, removal of water exchange may result in too high salinities similarly resulting in seagrass mortality. Very high salinities also occur around outlets of desalination plants operating to extract freshwater.

Heat stress may occur in connection with power plants using salt water as a cooling agent. In addition to these local human disturbances, the ongoing global climate change will affect temperatures with the possibility of increased heat stress in the southern and eastern parts of Europe, while a potential cooling of the climate in northern Europe due to changes in oceanic circulation may move the northernmost distribution limits of the *Zostera* species southwards.

Actions to prevent occurrence of heat and salinity stress on seagrass beds consist of means to ensure sufficient water exchange with the open

Text box 11.1. The most important actions to prevent seagrass loss are:

- Control and treatment of urban and industrial sewage to reduce the loading with nutrients, organic matter and chemicals
- Regulation of land use in catchment areas to reduce nutrient runoff and siltation due to soil erosion
- Regulation of land reclamation, coastal constructions and downscaling of water exchange between open sea and lagoons
- Regulation of aquaculture, fisheries and clam digging in or adjacent to seagrass beds
- Create awareness of the importance of seagrasses and implement codes of conduct to reduce small-scale disturbances

sea or ensure fast dilution of high salinity or high temperature outlets. Outlets with high salinity from desalination plants or cooling water from power plants should not be positioned in shallow waters inhabited by seagrass beds but instead be diverted to deeper waters with higher currents and dilution capacities. To counteract effects of global climate change, however, calls for actions at the international scale.

Contaminants

Pollution with man-made chemicals, such as pesticides, polyaromatic hydrocarbons, oil spills or anti fouling agents, may in certain areas constitute a substantial problem to seagrass performance and survival. These compounds have different effects on the physiology of seagrasses and other organisms and should in general be avoided through proper sewage treatment, regulation of industrial waste disposal or substitution of hazardous compounds.

The sources of chemical pollutants are numerous with crop cultivation as main responsible for pesticide contamination, industrial activities responsible for the disposal of a very large number of chemicals and ships and ship building industries as responsible for release of anti-fouling agents.

Actions to reduce pollutants must target the sources directly through legislation to implement adequate treatment of disposals and to substitute chemicals with unintended environmental impact by alternative and less toxic agents. These actions typically require national and international legislation while few means exist on regional or local scales. The fact that many chemicals may be transported over long distances in the atmosphere further stresses the importance of international legislation or agreements.

Invasive species and diseases

Seagrass loss may occur as a consequence of biological invasion of non-native species either competing with or feeding on the seagrass beds. Sudden outbreaks of mass destructive diseases are another potential cause of seagrass loss (chapter 6). There is an ongoing debate on the negative impact of the Mediterranean invasion of the green algae, *Caulerpa taxifolia*. Another example is the invasion of the Asian exotic mussel, *Muculista senhousia*, which can build up very high densities and have a negative impact on eelgrass beds. Finally, the outbreak of the eelgrass “wasting disease” caused by a slime mold in the early 1930s had dramatic consequences for eelgrass stands along the coasts of North America and Europe.

Introduction of new species with potential negative impacts on seagrass beds is an increasing threat due to the continuous increase in shipping transport between different regions of the world, mariculture based on foreing species and import of exotic species for aquaria. There are efficient means to reduce the risk of introducing invasive species with ship ballast water and a few contries (e.g. Australia and New Zealand) have implemented strictly enforced regulations to prevent intentional import of new species. Such regulations should also be implemented within the European Community. However, invasions of non-native species and outbreaks of diseases are very difficult to prevent and control. Fortunately, most introductions are non-successful because the non-native organisms are less fit than their native competitors. However, under conditions where the seagrasses are under environmental pressure due to human stress factors, they may be less able to overcome pressure from invading species. Accordingly, the best defence against the threat from invading species and diseases is to ensure otherwise undisturbed environmental conditions for seagrass growth.

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12 How long time does it take to recolonize seagrass beds?

The time scales for seagrass recolonization after events of disturbance are important information for seagrass managers, because, based on this information, they can decide what actions need to be taken for each scenario. Seagrass recolonization potential vary considerable among species. The potential for seagrass colonization is a function of both rhizome elongation, which determines patch growth, and reproductive effort, which sets the potential for formation of new patches. Additionally, environmental conditions are important factors in the recolonization times.

By Alexandra H. Cunha (CCMAR), Carlos M. Duarte (IMEDEA) and Dorte Krause-Jensen (NERI)

Seagrass landscapes are often patchy with patches from less than one meter to several kilometres wide. Patch-forming mechanisms operate at spatial and temporal scales controlled by intrinsic and extrinsic factors. The dynamics of seagrass patches is the basis for the maintenance of this structure and it is important to understand the changes that occur in these meadows. The potential for seagrass colonization is a function of both rhizome elongation, which determines patch growth, and reproductive effort, which sets the potential for formation of new patches. Clonal propagation is considered to be the most important process for the maintenance of seagrass meadows. Recruitment from sexual reproduction depends on the flowering probability and survival of the seeds, which are very variable between species.The environmental conditions play here an important role that is not quite understood yet.

It is difficult to estimate the recovery of a seagrass meadow based on their growth capacity, but some simulation techniques have been used to produce models. These models are based on the growth of the horizontal rhizome and the rate of formation of new patches assuming the rate of patch formation to be constant. The predicted recovery times are particularly short for fast-recovering species (e.g. *Cymodocea nodosa* – within 1 year) but can exceed a century for *Posidonia oceanica* due to the slow rate of patch formation of this species.

The aim of this chapter is to provide an overview of the recolonization capacity of European seagrass beds on long and short time scales and on large and small spatial scales. We searched for examples that would contribute to the understanding of the capability of seagrass to recolonize new areas.

How do seagrasses recolonize?

Gaps within seagrass meadows formed by natural events, such as storms or by anthropogenic disturbances, such as anchor scars, are primarily recolonised by the extension of the rhizomes of the plants located in the periphery of the gaps. The effectiveness of this process is limited by the horizontal extension rates of the species present. Hence, gaps in the order of 1 m^2 can be closed within one or a few years, whereas gaps tens of meter across require a decade or longer to be recolonised. Horizontal rhizome extension (i.e. clonal growth) is not an effective mechanism for the recolonization of larger gaps, which must necessarily involve sexual reproduction to initiate new patches within the denuded area.

Seagrass patches can also be established from drifting fragments, if they are anchored or trapped within the area to be recolonised. Also, following storms, or intense anthropogenic activities such as clam digging or boat anchoring, detached pieces of flowering shoots can float and release seeds far from the original area. Whenever an entire meadow is lost, due, for instance to disease related decline, recolonization is dependent on the arrival of propagules as seeds or viable vegetative fragments from neighbouring populations. Among the European seagrasses, only *Posidonia oceanica* has buoyant seeds able of long-range (10's of km) dispersal. The seeds of the remaining species are negatively buoyant, and the seeds of *Cymodocea nodosa* are produced at the base of

Figure 12.1 – Runners of *Cymodocea nodosa* in Ria Formosa, southern Portugal. Seagrass patches grow by the horizontal expansion of the rhizomes that extend out of the patches centrifugally. Photo: Alexandra H. Cunha.

the shoots and are often positioned at, or just below, the sediment surface. These seeds are, therefore, not likely to disperse far. The dispersal range of seagrass seeds is a very poorly studied aspect of their reproductive ecology, and robust estimates of dispersal events are only available for *Zostera marina* populations, for which 95 % of the seeds are retained within 30 m from the source, and robust estimates of dispersal events are only available for *Zostera marina* populations.

Once in the sediment, the seeds of some seagrass species can remain dormant for some time, with a documented dormancy period of a few months for *Zostera marina* and 7-9 months for *Cymodocea nodosa*, thereby building a rather ephemerous seed bank. Indeed, seed survival for more than one year appears to be a rare situation in seagrasses. While high seed production (e.g. > 50,000 seeds $m^{-2}yr^{-1}$ for some *Zostera marina* stands) and dense seed banks (e.g. > 1000 seeds m^{-2} for *Cymodocea nodosa*, and *Zostera marina*) have been reported in some meadows, seedling density is always comparatively low (typically about 1 to 10 % of the seeds produced) providing evidence of high seed losses. These losses are due to many factors including lack of viability, physical damage, export to unsuitable areas and burial. Moreover, seeds are also lost through predation, as documented by the experimentally estimated 65 % loss of *Zostera marina* seeds to predators. Seagrass germination does not appear to be too demanding, so that germination rates are typically high under field conditions. Seedling mortality is very high with a reported survival probability of only a few percent of the germinated seedlings over the first year for most species, although it may occasionally be very high as the exceptionally high seedling survival rates reported for *Posidonia oceanica* seedlings established on dead rhizome mats.

Successful recruitment from sexual reproduction is a rare event in most seagrass species, particularly in *Posidonia oceanica*, where seedlings have been reported only in a few meadows. Low flowering probability and low survival rates of seeds are major bottlenecks. Even when sexual reproduction culminates in the establishment of a new plant, this process is still an ineffective way to propagate because of the short dispersal distances that most species seem to exhibit. The inefficiency of sexual reproduction highlights the importance of clonal propagation as the main process responsible for the maintenance of seagrass meadows.

Small scale recolonization in seagrass meadows

Small scale recolonization in seagrass meadows depends on rhizome growth and patch formation rates. Seagrass patches grow by the horizontal extension of the rhizomes that extend out of the patches centrifugally (Fig. 12.1).

These rhizomes, which colonise unvegetated substrata, produce longer internodes and thereby grow faster than rhizomes growing in the interior of the meadow. The growth of the colonizing rhizomes is supported by the translocation of resources from the older shoots. The available data on rhizome elongation rates shows a considerable variability among and within species (Table 12.1). Species with low elongation rates as *Zostera marina* may attain a high potential for colonization by balancing with a high reproductive effort.

Besides the species capacity for rhizome growth, patch growth is subject to considerable variability among and within patches. The growth rate of seagrass patches is initially slow but accelerates as the patches grow larger due to the proliferation of branching. In addition, seagrasses modify their surroundings by reducing currents, stabilizing sediments and reducing sediment resuspension

Table 12.1.Rate of horizontal elongation rate (m $year^{-1}$) of European seagrass species

Species	Rate of horizontal
Zostera marina	0.1 – 0.9
Zostera noltii	0.44 - 1.68
Cymodocea nodosa	0.09 – 2
Posidonia oceanica	0.02 – 0.08

within the patch. Also as the patch grows, a positive feedback in seagrass growth takes place such that seagrass patches form a "mutually sheltering" structure as they grow.

The rate of formation of new seagrass patches has been examined for a few seagrass populations (Table 12.2).

Patch growth of Zostera marina

Zostera marina has declined extensively during the last decades following eutrophication of coastal regions of Western Europe. Sharply declining mortality with increasing patch age and size was observed in a population of *Zostera marina* growing in a protected embayment in Limfjorden, Denmark. This observation was associated to improving anchoring, mutual physical protection and physiological integration among the shoots as the patches grow larger. Here, the patch size distribution was dominated by small patches, which were formed by seedlings at high rates during spring (0.16 to 0.76 m^{-2}). The seedlings were subjected to high mortality and only 24% survived. The lateral growth rate of established patches was low (16 cm yr^{-1}on average) and, therefore, acreage expansion will be faster in systems with many small patches then in systems with few large patches. In large scale systems, successful recovery will be dependent on seed production, seed establishment and subsequent patch development.

Patch growth of Zostera noltii

Patch growth of *Zostera noltii* has not been investigated as yet. However, this species combines a very high branching rate with a moderately fast rhizome elongation rate suggesting that it must be capable of fast patch growth. The use of patch growth models to simulate *Zostera noltii* patch growth suggests that patch growth is fast and compact due to the high branching rate. Observations of fast recovery of gaps created by disturbance in *Zostera noltii* stands support the conclusion that recolonization rates are fast for this species.

Patch growth of Cymodocea nodosa

In the Mediterranean a *Cymodocea nodosa* community growing over highly mobile sandy sediments demonstrates that patch development proceeds by colonization by seedlings (0.047 m^{-2} yr^{-1}) and subsequent patch development by the surviving seedlings (<10%). Seedling distribution was highly aggregated and tended to be greater in areas where previous patches had died. There was considerable seedling mortality in the first year after germination (>70%). Rhizome development, necessary for patch formation from established seedlings, was observed in less than 50% of the seedlings. Patch growth was subject to considerable variability among and within patches, which demonstrates the influence of the environmental conditions. Difference in growth among patches of different sizes are attributed to a positive feed back effect accelerating patch growth as patch size increases.

Time scales for recolonization in seagrass meadows

It is difficult to provide an estimate of the recovery rate of seagrass species based on their growth capacity. However, estimates of the time scales for seagrass recovery have been produced using simulation techniques as well as estimates derived from projections of observed patch growth rates. In one such model, the interplay between the horizontal growth and the rate of formation of new patches was combined to provide a first-order estimate of the likely colonization time of different seagrass species. The model results predict recovery times ranging from < 1 year to centuries depending on species. The predicted recovery times for fast-growing species (*Zostera noltii* and *Cymodocea nodosa*) are particularly short, so that even with a moderate rate of new patch formation, a meadow can be established - provided suitable habitat is available - within 1 or a few years. In contrast, the recovery time for *Posidonia oceanica* involves several centuries, which would be in a permanent stage of overall decline if the return time of disturbance is shorter than a century. Extrapolations derived from the

Table 12.2.Examples of rates of formation of seagrass patches.

Species	Rate of formation	Source
Zostera marina	1000 patches ha^{-1} yr^{-1}	(Olesen & Sand-Jensen 1994)
Cymodocea nodosa	45 patches ha^{-1} yr^{-1}	(Duarte & Sand-Jensen 1990)
Posidonia oceanica	3 patches ha^{-1} yr^{-1}	(Meinesz & Lefèvre 1984)

observed growth rate of seagrass patches concur with model estimates to indicate that recolonization time scales for *Posidonia oceanica* are in the order of several centuries. Hence, any large-scale loss of *Posidonia oceanica* must be considered almost irreversible on managerial time scales.

Some seagrass populations exposed to periodical disturbance (e.g. transit of sand waves, etc.) are permanently in a stage of local loss and recovery resulting in a characteristic patchy landscape. Whether the substrate is devoid of seagrass, supports a dynamic patchy landscape or a continuous seagrass meadow depends on the frequency and magnitude of the disturbance relative to the capacity to resist disturbances and the recovery time of the species. Such dynamic equilibria have been demonstrated for *Cymodocea nodosa* growing on a Mediterranean bay swept by sand waves 10 – 20 cm in height. The progression of the sand waves causes the mortality of the buried patches as the rhizomes become exposed and colonised by drilling organisms following the passage of the sand wave, but the time interval in between the passage of consecutive sand waves was sufficient to allow the formation and development of new patches. *Posidonia oceanica* even survives larger (> 30 cm), frequent (> 1 yr^{-1}) sand waves without experiencing mortality. Hence, a persistent patchy landscape is only possible under combinations of disturbance magnitude and frequency causing only partial mortality and allowing the partial recovery of the species involved.

Zostera marina

Nature provided a large-scale test on the recolonization capacity of eelgrass when the worldwide wasting disease eliminated the majority of the European eelgrass populations in the 1930s. Fortunately, the process proved to be at least partly reversible. Assessments of eelgrass area distribution through the latter half of the 19th century provide evidence from many areas that considerable recolonization had occurred within 2-3 decades. However, recolonization after the wasting disease has not led to a complete reestablishment of the former distribution and abundance of eelgrass. In many areas various kinds of anthropogenic disturbances hinder full recolonization and cause further decline of especially deep populations (see chapter 5 for details).

Zostera noltii

Quantitative observations of recolonization dynamics in *Zostera noltii* are few if not non-existing. However, indications on the recolonization capacity following local and larger-scale disturbance have been conducted at Ria Formosa (S. Portugal). Clam digging often creates gaps within *Zostera noltii* populations. However, these have been observed to be recovered within a few months, and there are indications that the disturbance stimulates flowering and seed production, which may further contribute to the fast recolonization. The migration of sand bars has also been shown to disturb *Zostera noltii* at larger scales (ha) in the Ria Formosa. Recovery from such large-scale disturbance was slower, but still relatively fast, in the order of a few years (see chapter 5 for details).

Posidonia oceanica

Within the European seagrass flora, *Posidonia oceanica* is the species with the lowest recolonization capacity. The slow rhizome elongation rate (a few cm per year) and the sparse flowering of this species, where the meadows may not flower in several years and when they do, this involves only a small fraction of the shoots, are conducive to a low recolonization capacity. Small (m^2) gaps within *Posidonia oceanica* meadows can remain visible over several years, and large-scale recovery is indeed very slow requiring time scales of centuries (500 – 800 years, depending on patch formation rate). Yet, large-scale decline appears to be widespread in the Mediterranean Sea, involving a number of factors, such as constructions along the shoreline (e.g. ports, wave breakers, etc.), enhanced organic and nutrient inputs from land and from aquaculture activities, coastal erosion and mechanical damage by trawling boats and anchors. The deterioration of sediment conditions due to enhanced organic inputs may slow recolonization even further, as rhizomes cannot extend into anoxic sediments. Recolonization rates can be somewhat accelerated by measures to improve the growth conditions for seagrasses including action on the sources of deteriorated water and sediment quality and measures to stop mechanical damage.

Cymodocea nodosa

Within the European seagrass flora, *Cymodocea nodosa* is the species with the fastest rhizome growth (up to 2 m $year^{-1}$) and this species may flower profusively. However, the very low dispersal capacity of the seeds implies that

recolonization processes are largely dependent on horizontal spread as well as new patch formation from occasional seeds or vegetative fragments that may colonise the area. Observations on the dynamics of *Cymodocea nodosa* landscapes indicate that gap recolonization occurs within a year, and the meadow development takes place in less than a decade.

Conclusion

European seagrasses differ greatly in recolonization time scales due to the vast differences in sexual reproduction, seed dispersal and horizontal elongation rates. Whereas, recolonization of *Zostera* species and *Cymodocea nodosa* seems to be a relatively rapid phenomenon, extending over months for gap recolonization, to a few years for large-scale recolonization, recolonization of *Posidonia oceanica* is much slower involving time scales of a decade for gap recolonization to several centuries for large-scale recolonization. In managerial terms, loss of *Zostera* species and *Cymodocea nodosa* area can be considered reversible provided adequate measures are implemented to facilitate recovery, whereas loss of *Posidonia oceanica* should be considered as an irreversible process. Hence, conservational efforts, which must prevent losses of all European seagrasses, must be particularly strict to protect *Posidonia oceanica* meadows.

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13 Can transplanting accelerate the recovery of seagrasses?

Numerous attempts have been made to meet the tremendous seagrass losses the marine environment has faced worldwide. Artificial transplanting of shoots and spreading of seeds from intact meadows to non-vegetated coastal sediment are the most applied techniques. Planted seagrass beds can function and grow exactly as natural beds. However, favourable environmental conditions must be obtained before artificial restorations are considered. Furthermore, new approaches involving less labour and improved survival success rates must be developed before transplantation techniques can become an effective and widespread tool for seagrass recolonisation in European coastal waters. Education of the public as to the relevance of seagrasses can be one of several important goals in a restoration project.

By Peter Bondo Christensen (NERI), Elena Díaz Almela (IMEDEA) and Onno Diekmann (CCMAR)

As natural colonisation of many seagrass species are very slow (see chapter 3 and 12), it is tempting to speed up the recovery process by actively introducing vegetative plants or seeds into areas previously colonised by seagrasses and thus restore the environmental benefits that marine meadows provide to coastal ecosystems (see chapter 2). Restoration programs have indeed been introduced worldwide, most pronouncedly in the USA and Australia – but some large-scale programs have also been implemented in Europe. Detailed guidelines have been developed and may be of great help for managers considering seagrass transplantation. Such guidelines provide detailed considerations regarding costs and strategies before, during and after transplanting activities, i.e. donor and recipient site selection, transplanting procedure and monitoring of success rates.

Several interests may find expression in restoration programs:

- Introduction of seagrasses into areas that have been denuded of seagrasses and are far away from donor populations
- Speeding up of seagrass recolonisation in areas where it is proceeding already, but at a slow rate.
- Increase of species diversity in sites that have historically supported a diverse array of dense plant populations.
- Improvement of the genetic material in seagrass populations.
- Education of the general public as to the relevance of seagrasses.

Transplanting to sites in which environmental conditions approach those of the donor site as much as possible is the most obvious way to improve the survival chances of the transplants. Furthermore, recipient sites should preferably have supported seagrass in the past. Before any transplantation is performed, it is therefore essential to examine if the environmental conditions can meet the requirements for plant growth and survival (see chapter 4). Among factors to be considered are requirements such as light availability, water turbidity, nutrient levels, sedimentation rate and nature, sediment type and quality (contents of sand and organic matter, sulphide and oxygen conditions), sediment stability (erosion or siltation), current intensity, wave exposure, water depth, temperature and salinity and potential herbivore pressure. If the required environmental conditions can be met, sites considered for restoration should always be tested by experimental plantings to ensure that both environmental conditions and plants are adequate before any major restoration projects are implemented.

Lack of genetic diversity is thought to make populations more uniformly susceptible to diseases and other disturbances. It has indeed been shown that transplanting a genetically diverse population of i.e. Posidonia oceanica increase the chances of survival and genetic data can thus be useful to avoid transplanting shoots

that actually belong to the same genetic individual, i.e. shoots from one clone.

Although information on the local adaptation of seagrasses is still limited, recent information is now available on the genetic similarities across geographically widely distributed populations of all European seagrass species. This information can be useful when selecting populations from which to transplant or restore seagrass beds. Adaptation of seagrasses to their specific environment is reflected in their genetics. The information on genetic structure of European seagrasses obtained in the M&M's project and other works can therefore be a useful tool also for determining recommended maximum distances between donors and recipients.

Transplantation of eelgrass (*Zostera marina*)

Eelgrass is the plant species that has been transplanted most widely in coastal areas. Back in the early 1940s, the first attempts were made to mitigate the massive losses resulting from the eelgrass wasting disease in the Northern Hemisphere. Since then, much effort has been invested in developing different techniques for eelgrass transplantation in particular in USA.

Transplantation of plant material

By the “Plug Methods”, plugs consisting of seagrass and attached sediment are harvested using core tubes of various sizes. Plugs are extracted from the donor bed and transported within the tube to the planting site. At the planting site, another hole must be made to accommodate the planting plug.

By the “Staple Method”, plants are dug up using shovels. The sediment is shaken from the roots and rhizomes. Groups of plants are then attached to staples by inserting the root-rhizome portion of the group under the bridge of the staple and securing the plants with a paper-coated metal twist-tie. The staples are inserted into the sediment so that the roots and rhizomes are buried almost parallel to the sediment surface, as they occur in nature.

In the “Peat Pot Method”, sediment blocks are removed as when using the plug methods. A 3x3-inch sod plugger is used as a standard to cut plugs from existing beds. The sediment-plant plugs are then extruded into peat pots, which are subsequently installed in the donor sediment. Once in the sediment, the sides of the peat pot should be ripped down to allow rhizomes to spread.

TERFS (Transplanting Eelgrass Remotely with Frame Systems) is a modification of the staple method. Several eelgrass shoots are attached with biodegradable paper ties to a metal frame, which is lowered into the water until it rests on the bottom. Once the eelgrass has rooted and the paper ties have dissolved, the metal frames are retrieved.

Transplantation of seeds

Several experiments have tested the use of eelgrass seeds for restoration purposes. This technique has the advantage that the impact on the donor population is minimised, and working with seeds may also be less laborious and time-consuming.

Normally, seeds are harvested by manual collection of mature reproductive shoots from established beds when seeds are being released from the flowing shoots. Shoots are then maintained in large flow-through tanks until the seeds are released from the shoots, after which seeds can be kept in tanks under ambient conditions until use. It has been reported that several divers jointly collected up to 30,000 seeds per hour, thus obtaining a large amount of seeds through relatively low effort.

Several seed planting techniques have been tried out. The seeds must be broadcasted before the onset of natural seed germination, the time of which may vary from latitude to latitude. Seeds have been broadcasted by hand, either from a boat or while wading in shallow water. The seeds are relative heavy and therefore sink to the sediment surface and are rapidly incorporated into the sediment very close to where they were broadcasted. However, the number of seedlings resulting from such broadcasting experiments have been very low, because seeds are either washed out, fail to germinate or are consumed by predators.

Attempts to protect the seeds by planting 15-20 seeds in 5x5x5-cm peat pots did not improve the results. The peat pots were held in greenhouse tanks until after seed germination and then planted in the field. The peat pots were, however, susceptible to being washed out during periods of high wave activity.

The most successful of the experimental seed planting methods is a technique by which seeds are placed in 1-mm burlap bags (5x5 cm) before planting. The burlap bags protect the seeds from

potential predation and minimise burial and/or lateral transport.

Success of Zostera marina transplantation activities

A compilation of data on planting unit survival from 53 reports published in the USA showed a mean percentage planting unit survival of 42% after app. one year. The survival rates may vary considerably with the planting methods applied, but in general many of the planting units are lost, a fact that must be taken into account in the planning of transplantation projects concerning *Zostera marina*.

Transplantation using *Zostera marina* seeds has not been very successful. In several experiments carried out in the USA, less than 10% of the broadcasted seeds germinated and formed seedlings and the density of the seeds did not influence those results. Seeds from eelgrass are very exposed to predation, and measures must be taken to prevent the seeds from disappearing when broadcasted on bare sediment. Furthermore, survival of eelgrass seedlings may be very low, and the low percentage germination and limited initial seedling success are the major challenges facing future research directed at making transplantation of *Zostera marina* economically feasible and environmentally successful.

Transplantation of *Zostera noltii*

Experience in transplanting *Zostera noltii* is very sparse compared to *Zostera marina*. Vegetative transplantation of *Zostera noltii* can be achieved relatively easily at low tide, and *Zostera noltii* has been transplanted successfully i.e. on the mud flats of Southeast England and in the German and Dutch Wadden Sea. The experiments demonstrated that bioturbating infauna like lugworms may prevent growth of transplanted *Zostera noltii* and that the strong hydrodynamics may reduce the density of meadows.

Laboratory experiments have demonstrated that the seeds of *Zostera noltii* potentially have a high germination rate. At a low salinity of 1 ppt and a temperature of 10°C up to 70% of the seeds germinated under laboratory conditions. However, less than 3% of the plants survived seven days at ambient laboratory temperatures. Thus, much research is needed before the use of *Zostera noltii* seedlings in transplanting experiments is considered.

Transplantation of *Posidonia oceanica*

Posidonia oceanica, the endemic and most widespread seagrass along the Mediterranean coasts forms extensive meadows, thousands of years old, at between 0.5 and 40 m depth. Due to the extremely slow rhizome growth rate of this species (1-6 cm/year) natural meadow recovery takes place on a time scale of centuries. This fact alone motivates artificial acceleration of recovery through transplantation. However, for a major transplanting programme to be implemented, thousands of plants are needed, and the slow growth rates of the plants therefore introduce the added problem of source material. If plants are to be supplied in the form of vegetative cuttings, collection of donor plants should be distributed in as many meadows as possible to avoid serious impacts on donor beds.

It is recommended that plant material be obtained on an intra-basin scale. The consequences of mixing Eastern and Western Mediterranean plants, which differ with respect to genetic as well as anatomic characteristics, are still unknown. When choosing donor meadows, the main factors to take into account seem to be environmental health (high rhizome reserves), genetic diversity (heterozygosity and number of alleles) of donors and depth of collection with respect to the trnaplnat site. Plants transferred from lower to higher water depths have a very low survival success while plants from relatively deep water may survive in shallow waters.

Survival rates of *P. oceanica* transplants and seedlings are highest on “dead-matter” reefs, that is structures formed by *P. oceanica* meadow growth over millennia. Those reefs consist of a framework of dead rhizomes and sediment particles that persist years after the disappearance of the *P. oceanica* meadow. Transplants do not survive for long on pebbles or gravel but may develop on rocks covered with epiphytes if currents and waves are sufficiently weak.

Transplantation of plant material

Horizontal rhizomes with an apex and two lateral branches constitute the most active parts of the plants for spatial colonisation (see chapter 3) and are therefore the best vegetative material for transplants of *P. oceanica*. However, as horizontal apexes are less abundant and more vital to donor meadows, vertical shoots can be chosen as alternative transplants. Furthermore, survival rates of vertical shoots are virtually the same as those of horizontal rhizomes, if the vertical shoots bear

Table 13.1. Suggestion for a monitoring programme to measure the success of *Posidonia oceanica* transplantation in the Mediterranean

	Beginning	After some years
Basic parameters	Survival Shoot formation Rhizome elongation Root production	Cover Shoot density Patch size
Frequency	Year 1: every 4 months Years 2 & 3: every 6 months Years 4 & 5: once a year	Until patch coalescence or targeted cover and density : visits every 2 or 3 years
Ecological parameters	Biotope: water turbidity, sedimentation rate, sediment granulometry, organic content and oxic level Biocenosis: composition and abundance of epiphytes on leaves and of associated fauna and flora	
Frequency	At least three times during the project: (1) before transplanting (2) intermediate stages (species-dependent) (3) end of project (when targeted cover is obtained)	

Monitoring item	Parameter	Frequency
Individual plants	Survival Shoot formation Rhizome elongation Root production	Year 1: every 4 months Years 2 & 3: every 6 months Years 4 & 5: once a year
Plant population	Cover Shoot density Patch size	Until patch coalescence or desired cover and density: every 2 or 3 years
Biotope	Water clarity Sedimentation rate Sediment granulometry Organic content Oxic level	At least 3 times during the project: (1) Before transplanting (2) Intermediate stages (species-dependent) (3) End of project (when desired cover is reached)
Biocenosis	Leaf epiphyte composition and abundance Associated flora & fauna	

two leaf bundles. One year after transplantation, most double vertical shoots spontaneously switch to horizontal growth and thus become more active in spatial colonisation.

Some vegetative shoots are naturally detached from meadows by storms, and such shoots have been used successfully as transplant material. The use of detached vegetative shoots does not impact donor meadows and the plant material is available throughout the year. To serve as transplant material, detached shoots should have at least one leaf bundle and show good signs of vitality. The rhizome fragments of the shoots should be at least 8-12 cm to provide sufficient nutrient reserves and antibiotic substances, which is present in the rhizomes, to protect the plant.

Shoots attached to plastic or nylon nets (of 25x25cm or 60x17 cm, 1cm^2 mesh), which in turn are attached to the substrate with metal sticks have been used successfully to transplant vegetative fragments or seedlings of *P. oceanica* in the Mediterranean Sea. Seedlings were protected with cheese clothes before being attached to the nets. The spacing between plants has ranged from 3 to 17 cm, respectively, in different transplanting experiments, resulting in similar rates of success.

Rhizomes secured between two pieces of superposed wire netting on rectangular concrete frames (0.07 m^2) have also been transplanted with success. These frames are easy to handle and could reduce the transplanting cost since they can simply be lowered from the boat and do not need any attachment to the sediment. However, they are not recommended on soft, muddy bottoms, because these heavy structures may sink into the sediment, where the transplants would be unable to survive.

Horizontal growth of *P. oceanica* plants is straightforward at first with low branching angles (see chapters 3 and 10). Therefore, it is be best to arrange the plants within a planting unit with apexes and leaf bundles facing away from the

centre of the unit, so that growth can take place more efficiently from the centre outwards.

Transplantation of seedlings

P. oceanica does not flower every year and seeds do not enter into dormancy, which means that a seed bank is not established below the plants. However, some meadows reproduce sexually more frequently than others – on average every second year. Further, widespread flowering and fruiting occurs especially in years following exceptionally high summer temperatures. Information on such meadows and on temperatures allows planning of seed collection. Fruits of *P. oceanica* float and seeds or seedlings can easily be collected in great numbers along the coasts in spring and summer. The seeds germinate very well in tanks containing seawater (70-80% germination success), where they can survive and grow under suitable conditions and thus make up a plant nursery for later transplantation.

Success of P. oceanica transplantation activities

The first year after transplantation is crucial for the survival of *P. oceanica* plants, as they must acclimate and establish roots during this time (Balestri *et al.* 1998). The best time of the year to transplant *P. oceanica* plants is in late winter after the severe autumn and winter storms have passed. The most active growth period of *P. oceanica* is from February to May, and the plants are therefore able to produce roots and anchor into the sediment before the next stroin autumn storm. Seedlings, in contrast, must be planted in June-July, after collecting the seed and seedlings from March to July. Collected seeds must germinate and the seedlings are preferably kept to grow at least two months in an aquarium before being transplanted.

Large-scale transplanting (1 ha) of vegetative shoots detached from meadows by storms was performed in the Bay of Cannes in 1984 with concrete frames. Ten years later, the plant-covered area had increased by a factor of seven and the number of shoots had increased nine times. The plants formed oblong islets with numerous running rhizomes indicating active colonisation.

However, epiphyte growth on leaves of transplanted plants was three times higher compared with leaves of adjacent non-transplanted meadows. This phenomenon may be density-dependent and destructive to the plants, since considerable epiphyte growth may lower the photosynthetic efficiency of the plants and attract grazers that might damage the transplants.

Pilot studies have shown that *P. oceanica* seedlings transplanted onto “dead matter” have high survival rates (70%) after three years. However, seedlings seem to be less capable of developing branching shoots compared with transplanted vegetative fragments – at least in the short term. Thus, after three years from transplantation, 87% of the vegetative transplants showed branching growth while this number was only 14% for seedlings.

Transplantation of *Cymodocea nodosa*

Experience in transplanting C. nodosa is very limited. Experiments in the southern basin of the Lagoon of Venice have demonstrated that C. nodosa can be transplanted successfully and that the performance of transplanted meadows after only two vegetative seasons were similar to those recorded in a natural meadow.

New thoughts may be considered

Transplantation of seagrasses is a laborious and expensive activity. The costs of transplanting are, of course, site and method specific and may vary dramatically. The price for a transplantation project in New Hampshire, USA, was approximately 250,000 Euro per hectare in 2002 prices. The costs may, however, be considerably higher depending on several factors, such as the need for SCUBA divers (depending on water depth), cold-water planting, soft sediment, low visibility, high disturbance or actual loss of transplanted material (e.g. due to bioturbation, rough seas etc.), which may necessitate replanting.

In addition, survival rates of both vegetative and seed transplants are still rather low and collection of material from existing meadows may affect the donor meadows negatively. This is especially the case when harvesting of vegetation for transplantation purposes leaves bare patches, which are susceptible to erosion. For slowly spreading species like *Posidonia*, such patches may stay bare for many years. Collection of seeds from a population may also be critical, as natural seedling recruitment may be necessary to ensure meadow vitality in the long term. Furthermore, however carefully donor plants are freed from the sediment, some damage to belowground plant components is inevitable, which may weaken the donor plants and lead to lower transplantation

success. Thus, using vegetative shoots that have been naturally detached from existing meadows or collecting floating seeds that have washed up at the coast seems to be the least destructive way of obtaining plant material for restoration purposes. In this light, more research should be applied to the development of laboratory-cultured plants for restoration efforts.

Assessment of transplantation success requires monitoring of the restored site, preferably for several years, and may therefore also be lengthy and costly. The monitoring survey may vary from species to species and from latitude to latitude. In Table 13.1, a monitoring programme for evaluating the success of *Posidonia* transplantation is suggested.

Increased public understanding of the ways in which conditions for seagrasses in coastal waters can be improved is essential. To achieve this goal, closer collaboration between scientists, managers and the general public must be obtained. In the USA, both management and environmental groups have now developed activities directed towards primary and secondary school children. The school groups are provided with necessary materials for raising species of seagrasses in the classroom and are later assisted in transplanting the plants into appropriate habitats.

"Xarxa de Vigilancia de la Posidonia" is the name of a new project with a similar goal that is being established in the Mediterranean. When seeds from *Posidonia* become available around Mallorca and the Balearic Islands, a large number of volunteers can be requested at short notice to collect the seeds. The seeds are then transferred to nursery greenhouses for germination and the seedlings are later transplanted to areas where a new population is needed. The help provided by numerous volunteering divers not only makes the project economically feasible, but also has a very important educational aspect, the objective being to create a much wider public understanding of the importance of protecting existing seagrass covered areas in whichever way possible.

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14 How can beaches be managed with respect to seagrass litter?

Seagrass meadows produce large amounts of leaf material that is shed and eventually may reach the shore, often building important banks of seagrass litter. These deposits often represent a source of concern for the manager, whenever they accumulate in beaches and shorelines used for recreational purposes. This chapter describes the formation of these litter banks, their role on the littoral systems and discusses possible management options.

By Carlos M. Duarte (IMEDEA)

The nearly continuous growth of seagrass leaves implies that each shoot produces a large number of leaves every year, ranging from about 30 in the case of *Zostera noltii* to 6 in the case of *Posidonia oceanica* (see chapter 3). These leaves eventually reach a senescent stage and are shed. A fraction of this material, representing roughly 25 % of the production of the seagrass meadow, is transported to shore and deposited along the coastline (Fig. 14.1). Because the production of seagrass per unit area is in the order of 500 grams of dry weight per m^2, the amount of litter reaching the coastline will depend, besides currents and winds, on the width of the meadow. A moderately wide (1 km) belt of seagrass may deliver seagrass litter in excess of 125 kg of dry seagrass material per meter of coastline each year. Moreover, this is not a continuous process, but occurs mostly in the fall, when loss of material from the seagrass is highest coinciding with enhanced wave action and transport.

The amount of material cast accumulates at the beach, developing cushions of up to 4 meters in height, such as observed in some Mediterranean shores adjacent to large *Posidonia oceanica* meadows. This material has multiple functions, as these cushions protect the shoreline from erosion, delivers inorganic materials in the form of shells and carbonate produced by the plants and their epiphytes, nurishes the beaches, is the basis for a rich invertebrate food web, and, when transported further inland by the wind, may act as seeds for dune formation, also supplying dune communities with organic matter and nutrients.

Yet, European shores are no longer the wild environment they once were, as the use of beaches as recreational sites for society has generated an important demand for clean beaches. Conspicuous seagrass litter, although in fact a fingerprint of a lush adjacent seagrass meadows and, hence, adequate environmental quality, is considered a nuisance by a fraction of the beach users and beach managers and is often removed using heavy machinery (tracks, tractors, etc.). This management option, however, causes considerable damage to the beach and coastline ecosystems, as it disrupts the benefits derived from the cushion of seagrass litter, as described above, and also impinges considerable damage to the physical integrity of the beach. This damage is caused by (1) altered packing of the material due to the transit of heavy machinery on the beach; (2) removal of beach materials because not only the mineral particles associated to the seagrass litter are removed but also large amounts of sand, intermingled with the seagrass litter, which may

Figure 1. Photo of cushions of beach cast material. Photo: Carlos M. Duarte

represent up to twice, by weight, the seagrass litter removed.

Scientifically designed experiments of alternative management options have, to our knowledge, not been attempted. However, some alternative practices have been implemented with apparent success. The best practice is a no removal policy of beach cast material. However, this is not a possible option when materials accumulate in amounts sufficient as to interfere with leisure activities on the beach. In such heavily-used beaches, no-removal periods should be set to encompass the largest amount of time possible, when demands for beach space of users are low or moderate. At the same time, the public should be informed of the beneficial functions of that material and the fact that it should be considered as a fingerprint of good environmental quality offshore from the beach. Ideally, these efforts would result in a reduced pressure by users to have the seagrass litter removed. This demand can be monitored using user satisfaction leaflets or questionnaires, as to assess the thresholds of materials the users consider to reach nuisance levels. When removal must occur, this should avoid use of heavy-duty machinery and removal of the material outside the beach. The least-damaging practice may involve mixing the seagrass material with the sand as to bury the materials below the surface, thereby avoiding losses of beach sand and seagrass-associated minerals.

In addition to materials accumulated on the beach, some managers have been known to remove seagrass growing in shallow waters near the beach. This is indeed a most detrimental practice, as it enhances beach erosion, since the seagrasses removed trap particles, dissipate wave energy and fix the sediments with their rhizome and root networks. Hence, increased beach erosion is the likely consequence of removing shallow seagrass stands. Provided the worldwide tendency towards beach erosion, derived from increased sea level, preserving seagrass beds may prove critical to attenuate the erosion trends.

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Glossary

Alleles: Alternative form of a gene. One of the different forms of a gene that can exist at a single locus.

Blimp: a nonrigid balloon/airship.

Clonal growth: growth process regulating the vegetative proliferation of ramets

Clone: An individual formed by some asexual process so that it is genetically identical to its parent.

DNA (deoxyribonucleic acid): The long, spiralling molecule that rchestrates the cell's daily operations and provides the genetic blueprint for the physical characteristics of all living organisms. When made up of two strands, the strands intertwine like a spiral staircase to form a structure called a double helix. Subunits, called bases, are the rungs of the staircase.

Epiphyte: [Greek. epi: upon; phyton: plant] organisms growing on plants. In its broad definition epiphytic communities include bacteria, micro- and macroalgae and invertebrates.

Eutrophication: The excessive input of nutrients (nitrate, ammonium, phosphate, etc.) to water bodies triggering noxious algal blooms, deterioration of water quality, and the loss of functionality of coastal ecosystems.

Gene: The fundamental physical and functional unit of heredity, which carries information from one generation to the next. A segment of DNA, composed of a transcribed region and regulatory sequences that make possible transcription.

Gene flow: The movement of genes from one population to another by way of interbreeding of individuals in the two populations.

Genetic diversity: The variation at the level of individual genes.

Genetic marker: allele, DNA marker or cytogenetic marker used as experimental probe to keep track of an individual, a tissue, a cell, a nucleus, a chromosome, or a gene. (Genetic markers commonly used to study seagrasses: allozymes, microsatellites, RAPD)

Genetic polymorphism: The occurrence together in the same population of more than one allele or genetic marker at the same locus (with the least frequent allele or marker occurring more frequently than can be accounted for by mutation alone).

Genotype: The specific allelic composition of a cell, either of the entire cell or more commonly for a certain gene or a set of genes. The genes that an organism possesses.

Grain size: Grain size refers to the mean or effective diameter of individual mineral grains or particles.

Internode: Segment of rhizome separated by nodes.

Intertidal: The zone between the extreme high tide and extreme low tide marks; exposed to air within each tidal cycle.

Isotopes: Elements which have the same atomic number and, therefore, similar chemical properties, but differ in atomic weight and, thus, have differences in physical properties. The atomic weight appears as a left-hand superscript in chemical notation. The non radioactive isotopes of a chemical element are known as stable isotopes of this element. The stable isotopes of sulphur are ^{32}S and ^{34}S, of which the 'light' isotope ^{32}S is the most abundant naturally. The stable isotopes of nitrogen are ^{14}N and ^{15}N, of which the 'light' isotope ^{14}N is the most abundant naturally. The abundance of stable isotopes in materials is determined by measuring their ratio, R, e.g., $^{15}N/^{14}N$ and $^{34}S/^{32}S$, in the material of interest relative to their ratio in a global standard. These measurements are expressed in parts per thousand in the delta (∗) notation as $\delta = 1000\ [(R_{material}/R_{standard})-1]$. Materials with a positive ∗ value are enriched in the 'heavy' stable isotope relative to the standard, while materials with a negative ∗ value are enriched in the 'light' stable isotope.

Lacunae : Airfilled spaces/tubes running through the plant from leaves to roots via rhizomes.

Locus : The position of a gene, DNA marker or genetic marker on a chromosome

(Gene locus: The specific place on a chromosome where a gene is located.)

Microsatellite marker : Small repetitive DNA sequences dispersed in genomes.

Molecular marker : A molecular selection technique of DNA signposts which allows the identification of differences in the nucleotide sequences of the DNA in different individuals.

Node : The joint made by leaf scars between rhizome segments.

Nutrients : Elements which are important for biological growth, such as carbon, nitrogen, sulphur and phosphorus.

PCR : Polymerase Chain Reaction. This technique will allow a short stretch of DNA to be amplified.

Polymorphic : A gene with more than one allele variant.

Population : A group of organisms of the same species relatively isolated from other groups of the same species.

Ramet : vegetative unit potentially able to live on its own after becoming physically separated from the parent plant

Remote sensing : Measuring from a distance, e.g. from airplane or satellite

Rhizomatous plant : clonal plant with lignified or herbaceous stems

Rhizome : Below-ground horizontal or vertical stem consisting of segments.

Silt : Defined as mineral particles with effective diameters between 50 µm and 2 µm

Siltation : The excessive input of sediment materials (silt), which deteriorates water quality and leads to excessive sediment accumulation disrupting coastal ecosystems.

Subtidal : The constantly submerged zone immediately below the intertidal zone.

Sulphide : is short for hydrogen sulphide (H_2S). It is formed by the reduction of sulphate (SO_4^{2-}) in marine sediments when no oxygen is present and the redox potential is low