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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.

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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 km², which can be considered a minimum estimate (Green and Short 2003). For Europe the minimum estimate is 6,340 km², but a recent minimum estimate of the distribution area of *Posidonia oceanica* meadows in the Mediterranean would increase this European estimate by 25,000 km². The World Atlas also provides a rough maximum estimate of the global seagrass area of 500,000 km², 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 km². 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 km² (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 km² in 1919 but at only 5 km² in 1971 and 2 km² 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 km² of the area but in 2000 only 4 km². Fishing and anchoring activity most likely contribute to limit the present distribution area (Glemeréc 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

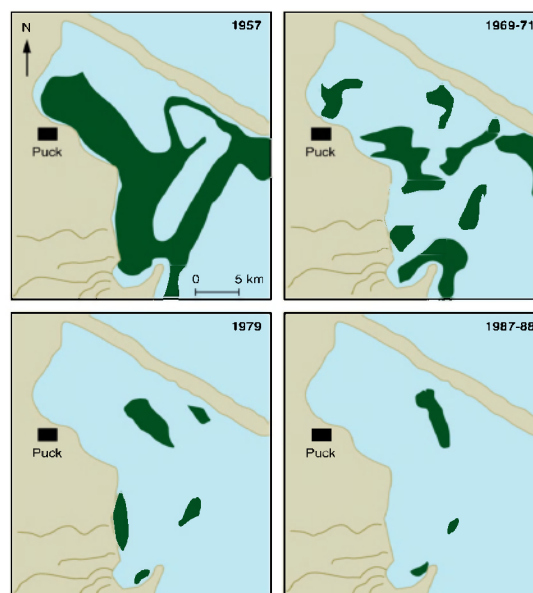


Figure 6.1. Eelgrass area distribution in Puck Lagoon, Poland: 1957, 1969-71, 1979, 1987-88 . Redrawn from Kruk-Dowgiallo 1991.

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,

paralleled 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 km² and covering about 25% of the sea bottom between 0 and 45 m in the Mediterranean basin (Pasqualini *et al.* 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 cartographies 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%

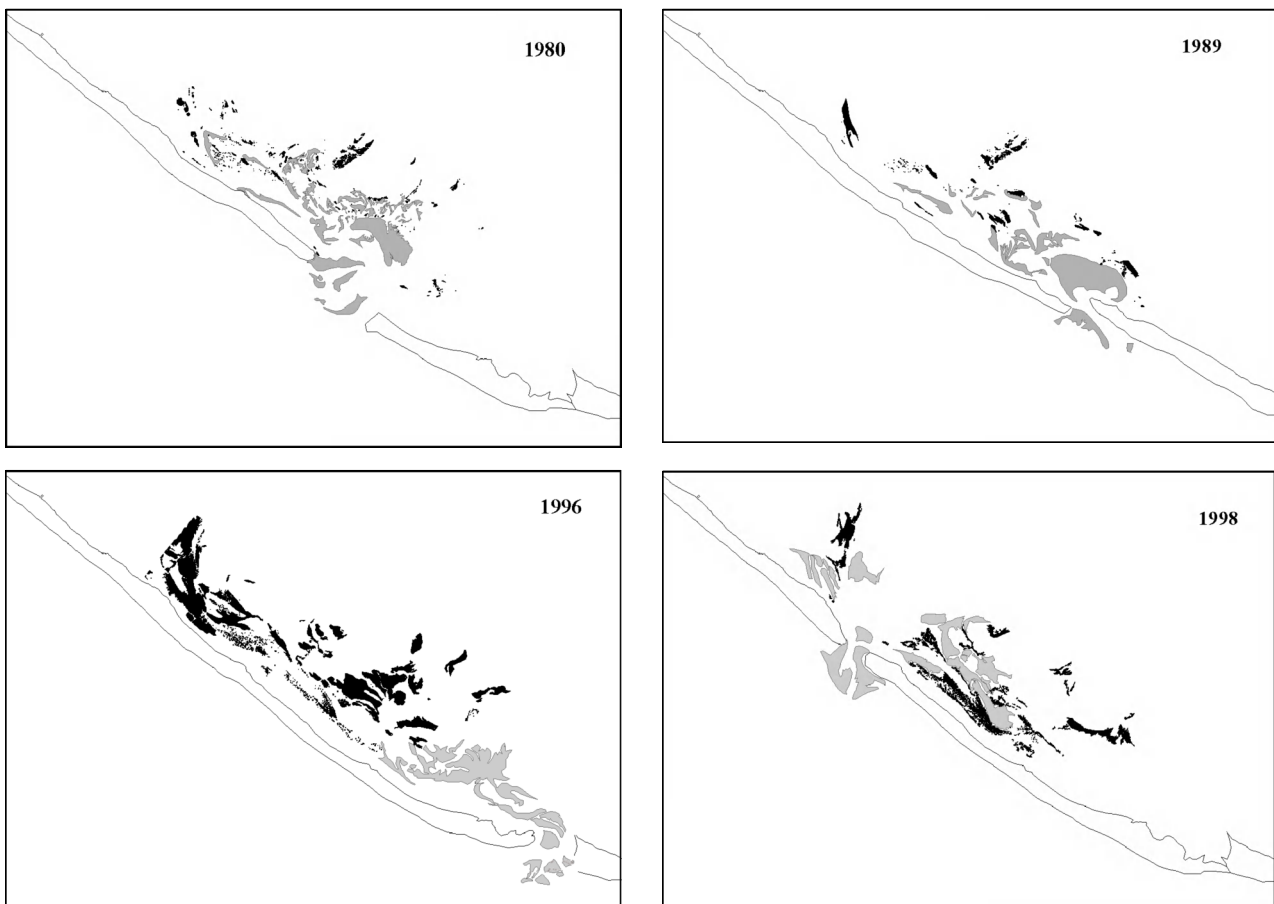


Figure 6.2. Changes in area distribution of *Z. noltii* in the Ria Formosa, Portugal as assessed by aerial photography. Black features: seagrass meadows; grey features: tidal delta. From Cunha *et al.* (submitted).

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 m², 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 m² 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 used 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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