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NATURA 2000

MANAGEMENT of Natura 2000 habitats * Posidonia beds (*Posidonion oceanicae*) 1120

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environment

The European Commission (DG ENV B2) commissioned the Management of Natura 2000 habitats. 1120 **Posidonia* beds (*Posidonion oceanicae*)

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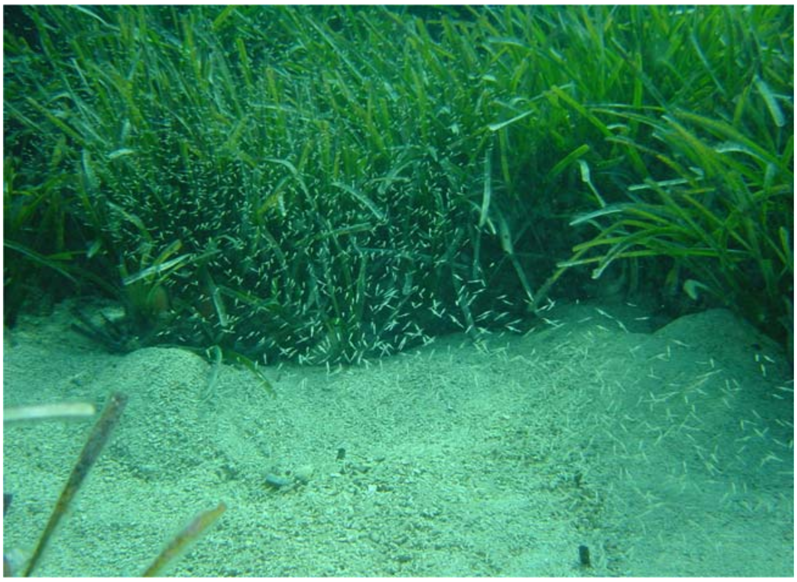

Posidonia oceanica meadow edge, with biogenic sands and mysida crustaceans. Photo: C.M. Duarte.

11 - Open sea and tidal areas

EUNIS Classification:
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* Priority habitat

Summary

P. oceanica is an endemic species to the Mediterranean Sea that forms dense and extensive green meadows whose leaves can attain 1 meter in height. These underwater meadows provide important ecological functions and services and harbour a highly diverse community, with some species of economic interest.

P. oceanica meadows are identified as a priority habitat type for conservation under the Habitats Directive (Dir 92/43/CEE). They require transparent, nutrient-poor waters and sediments devoid of labile organic matter. Over the last decades, following increased coastal urbanisation and industrialisation, many *Posidonia* meadows have disappeared or have been altered. It is estimated that 46% of the underwater meadows in the Mediterranean have experienced some reduction in range, density and/or coverage, and 20% have severely regressed since the 1970s.

Current main threats to the habitat are related to: water and sediment enrichment (eutrophication), the disruption of the sedimentation/erosion balance along the coast and direct destruction by human modifications of the coastline, degradation by boat trawling and anchoring, salinity increase in the vicinity of water desalination facilities and the proliferation of invasive algal species.

Conservation management is mainly focused on protective measures through the installation of artificial reefs and seagrass-friendly moorings for boats, in order to reduce the erosive pressure of otter-trawling and free anchoring in shallow meadows. The control of invasive especies (*Caulerpa taxifolia*, *C. racemosa*) has also been performed recurrently in some *P. oceanica* beds.

There is a need to further develop regulations for activities that have a negative impact on the *Posidonia* beds and other coastal ecosystems (e.g. pollutants level limits and allowed minimum distances of impact sources to meadows) and to implement it through the setting of a vigilance system. Such system could be coordinated with the seagrass monitoring networks already in place.

Seagrass monitoring is a fundamental tool for measuring the status and trends of meadows and is also essential to assess the effectiveness of any protective or recovery initiatives. The number of monitoring programmes on *P. oceanica* meadows has increased in recent years.

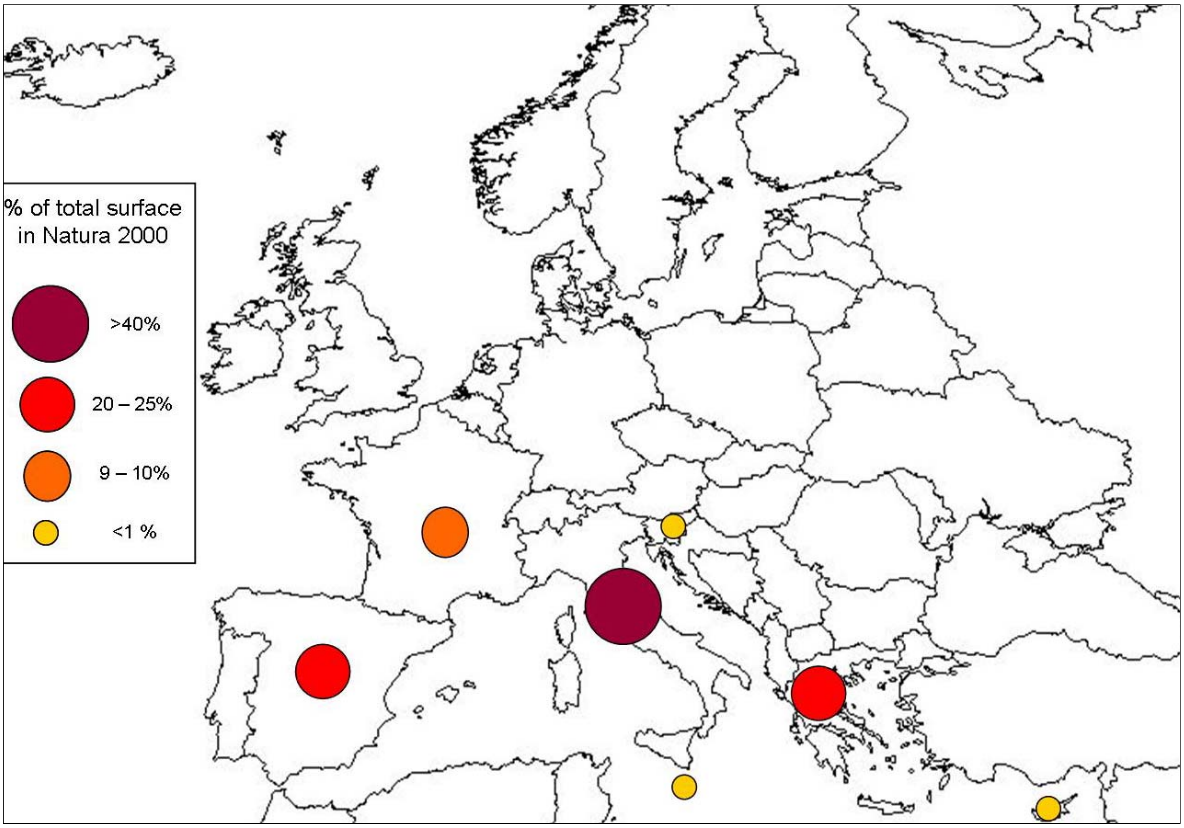
The slow growth of *P. oceanica* beds makes difficults recovery, which can take centuries, once the cause of habitat perturbation is eliminated. Recovery measures, like remediation of seagrass sediments enriched with organic matter, or transplanting of *P. oceanica*, are in experimental stage and need further development.

1. Description of habitat and related species

The species *Posidonia oceanica* is endemic to the Mediterranean Sea. It forms extensive underwater meadows that grow on rocks and sandy bottoms in clean water at a depth from less than 1 meter to over 40 meters (Hemminga and Duarte 2000). The rest of the *Posidonia* species (of which there are 7-8) are found in Australian waters, which illustrates the antiquity of the genus (den Hartog 1970). This seagrass is a strictly marine species and is never found in estuaries or salt-marshes.

Distribution

P. oceanica meadows are identified as a priority habitat type for conservation in the Habitats Directive (Dir 92/43/CEE). *Posidonia* beds are present in all Mediterranean countries and collectively occupy 2.5 -4.5 millions ha (Pasqualini *et al*. 1998). This constitutes nearly 25% of the Mediterranean basin having a water depth of less than 50 meters.



Percentage distribution of the total surface of Posidonia beds in Natura 2000

Posidonia beds in Natura 2000 sites

The following data have been extracted from the Natura 2000 Network database, elaborated by the European Commission with data updated on December 2006. The surface was estimated on the basis of the habitat cover indicated for each protected site and should be considered only as indicative of the habitat surface included in Natura 2000 (approximately 6 to 12% of the total habitat surface).

Biogeographical region	Nº of sites	Estimated surface in Natura 2000 (ha)	% of total surface in Natura 2000
Mediterranean	322	276,668	100
Countries	**Nº of sites**	**Estimated surface in Natura 2000 (ha)**	**% of total surface in Natura 2000**
Italy	157	122,049	44.11
Spain	70	70,029	25.31
Greece	71	57,514	20.79
France	18	25,999	9.40
Cyprus	4	952	0.34
Malta	1	118	0.04
Slovenia	1	6	0.01
TOTAL	**322**	**276,668**	**100**

Main habitat features, ecology and variability

P. oceanica is a large, slow-growing seagrass with wide and persistent rhizomes. It forms dense green meadows whose leaves can attain 1 meter in height during the summer. Old leaves are shed throughout the year, but especially in the autumn. In winter, the canopy appears shorter and sparser (10 to 40 cm high). Meadow density is maximal in shallow water (when it may attain more than 1000 shoots m^{-2}) and decreases exponentially with depth (70-80 shoots m^{-2} at 30 m). Enhanced sedimentation, combined with vertical rhizome growth, produces characteristic reefs called "matte". The matte is a network of dead rhizomes with shell/organic debris and sediments which accumulate over centuries to attain several meters in height (Hemminga and Duarte 2000).

P. oceanica meadows are able to support a relatively wide range of temperatures, as deduced from the wide latitudinal range of its distribution, from 31ºN in the coasts of Lybia to 45ºN in the Gulf of Trieste (Green and Short 2003). Therefore, although there are not specific experiments, from this latitudinal distribution we can deduce that the plant endures temperatures from approximately 10ºC to 29ºC. *P. oceanica* needs transparent, oligotrophic and oxygenated waters to survive. The depth to which the meadows grow is often limited by light (Duarte 1991, Duarte *et al.* 2007). The minimum light requirements of this plant are 0.1 - 2.8 mol PAR photons day^{-1} m^{-2} (Gattuso *et al*. 2006). *P. oceanica* also supports a narrow range of salinity, from 33‰ to 39‰ (Fernández-Torquemada and Sánchez-Lizaso 2005), however it is possible that East Basin populations may support higher salinities.

The seagrass uses the underwater substrate for anchorage and nutrient uptake. Sediment has to be relatively oxic (ie oxygenated). *P. oceanica* can grow on rocks or sandy bottoms. Muddy substrates are however not suitable as the plant is unable to attach itself and the water is too murky for plant growth. The sedimentation/erosion balance also often limits meadow development. As a result, *P. oceanica* never grows near river mouths or in confined waters (e.g. hypersaline coastal lagoons). Wave action is another important criterion: in sheltered bays, meadows can grow up to the water surface, forming fringing reefs, but in open coasts they usually start growing several meters below the surface (3-10 m).

There are slight morphological and genetic differences between *P. oceanica* meadows from different regions. In particular, there is a genetic cleavage between the Eastern and Western Mediterranean meadows which suggests that these meadows were temporally isolated from each other during last glaciations (Arnaud-Haond *et al.* 2007). Nevertheless, there are no clear geographical differences in meadow structure and function between the two basins, and the morph type differences disappear after some years of acclimatisation when transplanted to another site (Meinesz *et al.* 1993).

P. oceanica meadows constitute one of the main climax stages of Mediterranean coastal ecosystems. They harbour a highly diverse community, which varies according to depth, shoot density, adjacent communities, physico-chemical conditions and even historical events linked to larval recruitment (Hemminga and Duarte 2000).

Some species indicate seagrass perturbation: the overgrowth of epiphytic algae and especially the episodic formation of dense mucous layers of filamentous algae (*Ectocarpales* and *Crysophyceae*) on the meadow canopy is associated with water eutrophication and reduced hydrodynamics (Lorenti *et al*. 2005). The green algae *Caulerpa spp* invade declining sparse meadows, especially when the sediment is enriched with organic matter (Terrados and Marbà 2006). When nutrient inputs to the bed are too intense, sea urchins *Paracentrotus lividus* (normal densities 0-5 urchins m^{-2}) become over abundant (may attain 30 urchins m^{-2}) on meadows that grow near rocky substrates and consequently overgraze *P. oceanica* leaves (Ruiz *et al.* 2001). Their excess is therefore indicative of habitat eutrophication. Fire worms (e.g. *Hermodice carunculata*) also appear in degraded meadows with an excess of labile organic matter.

Species that depend on the habitat

A conspicuous and complex epiphytic community lives on the leaves of *P. oceanica.* This community is composed of large quantities of micro- (mainly cyanobacteria and diatoms) and macro-algae (over 94 species described). In healthy meadows, the red algae *Fosliella* spp. and *Hydrolithon spp.,* and brown algae, like the complex *Giraudio-Myrionemetum orbicularis* Ben, 1971, cover the tips of the leaves. Sessile animals, such as hydroids (over 44 species identified such as the obligate taxa *Sertularia perpusilla* and *Plumularia obliqua posidoniae*), or briozoa (more than 90 species, like the obligate taxa *Electra posidoniae* and *Lichenopora radiata*) are also a common component of the leaf epiphytic community. Microscopic foraminifera are also very abundant, specially in the less illuminated leaf-sides (e.g. *Quinqueloculina spp., Planorbulina mediterranea, Nubecularia massutiana* or *Conorboides posidonicola*). Some species and associations of foraminifera are exclusive of *Posidonia* meadows and are currently used by palaeontologists to diagnose the existence of ancient meadows in geologic strata (Colom 1974).

Within the rhizome substrate, some sessile species are only found on healthy *P. oceanica* meadows. This is true of the foraminifer *Miniacina miniacea* (whose shells are responsible for the characteristic pink colour of Mediterranean biogenic sands) and the large fan mussel *Pinna nobilis*, which, due to their filter feeding habits, longevity and slow growth, are good indicators of water quality and mechanical stability within the meadows. Other sessile species, such as the filterer worm *Sabella spallanzanii* are also indicators of water quality, but they are not restricted to *Posidonia* meadows (they also appear in rocky habitats). Algae adapted to low levels of light intensity (more than 74 species described, mostly red algae) colonize the rhizomes (e.g. *Peyssonnelia squamaria* and *Udotea petiolata*). Light dependent algae like *Jania rubens* may appear on meadow borders.

Mollusca (more than 185 species described) and *Crustacea* (more than 120 species of *Copepoda, Decapoda* and *Amphipoda*) are the most abundant faunal groups in *P. oceanica* meadows. There are some obligate taxa like the perfectly cryptic *Idotea hectica* and *Limnoria mazzellae* (Isopoda) or *Hippolyte inermis* and *Palaemon xiphias* (Decapoda). The *Polychaeta* are very abundant also (more than 182 species, like *Platynereis dumerlii* and *Syllis* spp.), although most species are ubiquitous. Sponges are abundant in the rhizome substrate (more than 15 species, like *Clathrina contorta* and *Sycon ciliatum*).

Among the Echinoderms, irregular detritivore sea urchins like *Echinocardium* and *Spatangus spp.* and regular herbivore sea urchins like *Spharaechinus granularis* and *Paracentrotus lividus* are common. The rare *Centrostephanus longispinus* can also be found in deep, rocky meadows. There are also *Crinoidea* (*Antedon Mediterranea*) and sea stars (e.g. *Ophioderma longicaudum* or the endangered, obligate species *Asterina pancerii*) but the most abundant Echinoderms are sea cucumbers (16 species described) which play an important ecological role as sediment filterers. Among them, *Holothuria tubulosa* predominates in dense, sandy meadows, while *H. polii* is more prevalent in sparse or degraded meadows, although it is very difficult to distinguish these two species. At night, many mobile species living within the rhizomes migrate to feed in the canopy.

Many fish species live in the *P. oceanica* meadows during their juvenile stage. There are also resident species, the most common of which are *Gobius* spp. (living on rhizomes), as well as *Labrus merula*, *L. viridis* (cryptic, specialist), *Symphodus spp.*, *Diplodus spp*, *Sarpa salpa*, *Coris julis* and *Chromis chromis*. There are also some obligate species living within the leaf canopy, like the cryptic species *Opeatogenys gracilis* and *Syngnathus typhle*. The endangered species *Hippocampus hippocampus* is also found within the canopy. Finally, the herbivorous green turtle *Chelonia mydas* feeds on tender seagrass leaves. Today it is an endangered species, but only a century ago, its population was probably several orders of magnitude larger. Their ecological role as seagrass grazers could have been of great importance in the past, judging by the effects they have on tropical underwater meadows (Jackson 2001).

Related habitats

In soft bottoms, *P. oceanica* meadows are usually surrounded by fine-grained sand detritic communities (Natura 2000 codes 1160-3, 1110-5). They are often combined with facies of the smaller and sparser seagrasses *Cymodocea nodosa* or *Zostera noltii* (the latter is more frequent in northern regions) or with the green alga *Caulerpa prolifera*. Sandbanks and dune systems accumulate large amounts of seagrass litter and biogenic sand, which is formed by the calcareous and siliceous debris of epiphytic fauna and flora. In turn, sediments may be exported from the beach to the meadow, determining their erosion-siltation balance (Medina *et al*. 2001). Therefore, the two habitats are strongly linked.

Saltmarshes can also regulate nutrient inputs into seagrass meadows (Valiela and Cole 2002). On rocky coasts, *P. oceanica* meadows are usually preceded by rocky algal communities (Natura 2000 code 1170). Deep meadows may be followed by soft-bottom maërl (red coralline slow-growing algae) assemblages or by coral/gorgonian communities. The former are important and sensitive communities which still lack formal protection (only two maërl-forming species are protected in the Annex V of the Habitat Directive: *Phytomatolithon calcaereum* and *Lithothamnion corallioides*, their Mediterranean counterparts are not).

The environmental requirements of submersed associated habitats are similar to those of *P. oceanica* meadows, although *C. nodosa* and *Z. noltii* support a wider salinity range.

Ecological services and benefits of the habitat

Posidonia oceanica meadows are key ecosystems within the Mediterranean Sea. The high rate of plant production (0.25 ± 3 kg (dry weight) m^{-2} $year^{-1}$ (Ott 1980, Pergent-Martini *et al.* 1994), mainly due to annual leaf growth, and the abundance of epiphytes (which can reach up 20–30% of the biomass of leaves), support a high secondary production *in situ* and in detritivore compartments of other communities (around 80% of total production, Cebrián and Duarte 2001), thereby sustaining complex food webs from beaches to bathyal areas.

A moderately wide (1 km) belt of *P. oceanica* meadow may produce litter in excess of 125 kg of dry seagrass material per meter of coastline each year (mostly during Autumn). This material accumulates on the beach, developing cushions up to 4 meters high, which can in turn sustain a complex invertebrate food web, protect the shoreline from erosion, deliver sand in the form of carbonate and silica shells and, when transported further inland by the wind, act as seed material for dune formation (Borum *et al*. 2004).

In daylight, *P. oceanica* meadows oxygenate coastal waters (Bay 1984), producing net oxygen releases to the atmosphere above the meadows. Due to the slow decomposition of lignified rhizomes and roots, the reef structure or "matte" acts as a long-term carbon sink (e.g. Gacia *et al*. 2002). The leaves and rhizomes increase the surface available to sessile species and offer shelter to mobile species, thereby sustaining a diverse community (Templado 1984). *Posidonia* beds are especially valuable as nursery grounds for several commercial species (Francour 1997).

The leaf canopy increases particle retention (e.g. Terrados and Duarte 2000), so enhancing water transparency. This function, combined with the active formation of calcareous and silica sand from shelled organisms (Canals and Ballesteros 1997) and cushions of seagrass litter, all contribute to reducing shoreline erosion. Finally, *P. oceanica* meadows are excellent indicators of environmental quality as they can only grow in clean unpolluted waters. Moreover, their rhizomes concentrate radioactive, synthetic chemicals and heavy metals, recording the environmental levels of such persistent contaminants.

Trends

Over the last decades, following increased coastal urbanisation and industrialisation, many meadows have disappeared or have been altered (e.g. Meinesz and Lefevre 1978). A sample of 39 studies in 135 sites shows that 46% of the underwater meadows in the Mediterranean have experienced some reduction in range, density and/or coverage, and 20% have severely regressed since the 1970s. In European coastal waters, the most dramatic losses have occurred in the northern Adriatic Sea where meadows that were present at the beginning of the 20th century have almost disappeared (Zavodnik and Jaklin 1990).

Given the extremely slow growth rate of this species (1-6 cm yr-1), such losses are virtually irreversible. Moreover, underwater meadows may decline more rapidly below a certain shoot density threshold. Shoot mortality exceeds recruitment in 60% of the Spanish Mediterranean meadows, yielding a median exponential decline rate of 5% yr^{-1} (19 meadows analysed, Marbà *et al*. 2005), which is more than double the 2% yr^{-1} global rate of decline in seagrass ecosystems (Duarte *et al*. in press).

In 7 of these sites, there is no evident human perturbation (two of them are in pristine areas of an MPA). This suggests the existence of a background decline trend, maybe related to general changes in the climate of the Mediterranean Sea (Duarte *et al*. 1999). If these trends are maintained, most of the *P. oceanica* meadows are predicted to halve in density over the next 20 years. Nevertheless, 6 pristine meadows, growing in MPAs around the Mediterranean were analysed through reconstructive techniques and showed positive net population growth in the last, indicating that there is no background decline trend linked to global factors, and that the decline observed throughout the Mediterranean would be the product of cumulative effects of natural and anthropogenic local processes (González-Correa *et al.* 2007a)

However, a slight recovery has been observed in some meadows following corrective measures. For example *P. oceanica* meadows off Marseille have experienced a partial recovery in density and extent during the last decade following the installation of a sewage treatment plant (Pergent-Martini *et al*. 2002). However, other cases indicate that meadow recovery is limited by the plant's slow growth and by the altered environmental conditions which often persist well after the cessation of the impacting activity. The projections for the total recovery of meadows undergoing protective measures are in the order of centuries.

Threats

Water and sediment eutrophication

P. oceanica meadows are very sensitive to water and sediment enrichment with organic matter and nutrients. Meadow decline accelerates when organic matter and phosphorus benthic inputs surpass 1-2 g (dry weight) m^{-2} day^{-1} and 0.04 g m^{-2} day^{-1} respectively (Diaz-Almela *et al*. in press.). This occurs through a series of cascade effects. When dissolved nutrients are high, epiphytic algae grow much faster and shadow the seagrass leaves, reducing seagrass light harvest and enhancing leaf grazing (Ruiz *et al*. 2001). Together with trawling, nutrient loading is the greatest cause of deterioration in seagrass beds.

The source of organic matter is often the same as those for nutrient loading, but they usually don't spread as far a-field. Labile organic matter increases sediment microbial activity, producing anoxia and increasing sulphate-reduction rates in the sediment. The excess hydrogen sulphide rapidly reacts with oxygen pumped through the seagrass roots, and may even penetrate the plant tissues, enhancing *P. oceanica* mortality (Frederiksen *et al*. 2007). Sediment hydrogen sulphide concentrations surpassing 10μM increase shoot mortality over 5% yr^{-1} (Calleja *et al* 2007) Reduced sediment conditions persist years after the organic inputs have ceased, prolonging meadow regression (Delgado *et al*. 1999). Therefore, untreated sewage outlets, fish-farm effluents or runoff from fertilized agricultural areas are serious threats to neighbouring *P. oceanica* meadows. In bays with low water exchange, even small amounts of nutrient and organic input from houses or boats may induce seagrass decline (Marbà *et al*. 2002).

Disruption of the sedimentation/erosion balance

P. oceanica meadows can cope, through vertical rhizome growth, with sedimentation rates that do not exceed 4-5 cm yr^{-1} (Gacia and Duarte 2001), and are very sensitive to erosion. Coastline transformation, with the proliferation of roads and houses and the regulation of continental river-flow, sharply reduces sediment inputs to the submersed coastal habitats, thereby promoting meadow erosion in their area of influence. Piers and other coastal constructions destroy the underlying communities and may alter the pattern of coastal currents thus passing on the effects of siltation or erosion to other meadows. Dredging and sand reclamation activities close to meadows have a high risk of direct meadow removal and may produce bed siltation or erosion. Finally, beach re-filling (Medina *et al.* 2001) may change sediment conditions and produce long-term siltation of the adjacent underwater meadow, slowing seagrass recovery (González-Correa *et al.* 2007b). On the other hand, removing the seagrass leaf litter from the beach may produce the reverse effect, enhancing shallow meadow erosion.

Direct erosion by boat-trawling and boat anchoring

Fishermen have been complaining about the effects of bottom trawling gear on the marine environment since at least the 18th century. Otter trawling is one of the most important causes of large-scale degradation of *P. oceanica* meadows, particularly in deep meadows (e.g. Ardizzone and Pelusi 1984, Erftemeijer and Robin Lewis 2006). The repeated use of trawl gear over the seabed pulls up *P. oceanica* leaves and rhizomes (100,000 to 360,000 shoots hour^{-1}, Martín *et al.* 1997)), largely reducing plant density and cover. As the trawl passes over the seabed, it also re-suspends the sediment and alters the substrate structure, increasing turbidity and nutrient concentrations in the water column. Reduced plant cover and the altered sediment interact in a negative way to maintain silted conditions. The slow regrowth of seagrass further prolongs the impact of trawling which can sometimes run into decades (González-Correa *et al*. 2005).

In sites frequently visited by pleasure boats, there is significant removal of seagrasses by boat anchors (Francour *et al*. 1999). Also moorings consisting of a dead weight lowered to the seabed, attached to a partially crawling chain, form characteristic bare circles in *P. oceanica* meadows. These clearings persist for many years. If the anchoring density and frequency are too high, the subsequent erosion may be accelerated by enhanced hydrodynamics.

Salinity increase in the vicinity of water desalination facilities

*P. oceanic*a is especially sensitive to increases in salinity levels (Fernández-Torquemada and Sánchez-Lizaso 2005). Salt concentrations above 39 p.s.u. induce rapid plant death (Sabah *et al*. 2003). Thus, the brine (40-80 p.s.u.) from water desalination facilities, poured directly onto *P. oceanica* meadows, can produce diebacks across large areas. Moreover, pipelines constructed to divert the brine to offshore areas destroy considerable meadow surfaces. The present and projected increase in coastal desalination facilities is therefore an emergent threat to *P. oceanica* meadows.

Proliferation of invasive algal species

The sustained increase in global marine transport favours the proliferation of exotic species which harm existing communities (Galil 2007). In the Mediterranean Sea, around 100 exotic macrophytes have been introduced in the last decades, of which at least 10 have an invasive behaviour (Ballesteros 2007). Those that most affect *P. oceanica* meadows are the green algae *Caulerpa taxifolia* and *C. racemosa*. Although these species do not apparently penetrate into dense healthy meadows, they may, when associated with other perturbations (e.g. eutrophication, bottom trawling), enhance meadow decline, since they compete for space and light and increase the contents of labile organic matter in the sediment.

Recently, the invasive red alga *Lophocladia lallemandii* has been shown to induce *P. oceanica* shoot mortality (Ballesteros 2007). *L. lallemandii* settles on rhizomes and old leaves along the edges of meadows and in low density patches where it grows rapidly, producing disc-like holdfasts along the thalli that enable the formation of a mat of red algal filaments intermingled with *P. oceanica* leaves. This mat can become so thick that leaves confined within may display chlorosis, and shoots eventually die.

Cosmopolitan filamentous algae (belonging to *Phaeophyceae* and *Crisophyceae* families) may also behave as an invasive species, forming dense mucous layers on *P. oceanica* canopies in calm periods usually of 1 to 3 months, and, in so doing, reducing light availability to the seagrass (Lorenti *et al*. 2005). Some studies indicate that such episodes affect *P. oceanica* growth and survival, while others show no evident impacts. Their effect probably depends on their persistence and frequency as *P. oceanica* can resist shading for several months (Ruiz and Romero 2001). Finally, *Acrothamnion preissii*, a new exotic red algae that invades *P. oceanica* rhizomes has no apparent effects on seagrass integrity but it does displace most of the autochthonous rhizome epiphytes which reduces the meadow's species diversity and habitat complexity.

Climate change effects

High temperatures and prolonged heatwaves reduce *P. oceanica* shoot growth (Mayot *et al*. 2005) and increase shoot mortality (Díaz-Almela *et al.* 2007). Sexual recruitment may be enhanced by temperature, but the balance is still negative.

The observed trend in Mediterranean sea warming and the expected increase in the number of heatwave episodes (Cubash *et al*. 2001), as well as other observed trends in Mediterranean sea climate (e.g. a general reduction in water transparency and the greater frequency of severe storms, Duarte *et al.* 1999) suggest that *P. oceanica* meadows will have to cope with enhanced climatic stress in the coming decades.

2. Conservation management

General recommendations

P. oceanica meadows are identified as a priority habitat type for conservation in the Habitats Directive (Dir 92/43/CEE). Moreover, the species and/or the habitat are under specific legal protection in several European countries (see Gravez and Boudouresque 2003). They are capable of self-maintenance without human intervention provided their physical and chemical environmental requirements are met. This apparently simple pre-condition is however increasingly difficult to secure in this day and age. Such requirements as transparent, nutrient-poor waters and sediments devoid of organic matter are incompatible with present human activities along coastal ecosystems.

This is further exacerbated by the slow re-growth of damaged *P. oceanica* beds. Once the cause of habitat perturbation is eliminated (already a difficult task) the meadows should recover but as their growth rate is so slow it could mean that this recovery takes centuries. Thus, the most effective form of management is one that aims to prevent meadow loss in the first place by maintaining suitable water and sediment conditions, and preventing large-scale erosion or siltation. Sustainable coastal development and an adequate control of negative external influences is the best means of preserving *P. oceanica* meadows and securing their important role in maintaining a healthy marine environment.

Active management

Protection of reefs against otter trawling

Bottom trawling has a heavy impact on almost any benthic ecosystem but an effective regulatory framework can help to protect especially sensitive habitats, like *P. oceanica* and maerl beds. In Spain, Italy and France, restrictions on trawling over meadows have been reinforced in the last decade by the deployment of protective artificial reefs. These reefs are usually installed in Marine Protected Areas (MPA), but, given the special protection status of *Posidonia* habitats, they could also be installed in any area with *Posidonia* meadows that suffers from illegal trawling. Protective reefs are heavy concrete constructions (usually cubic or pyramidal, Fig. 1c), which can be armoured with extruding steel bars. Any trawling gear passing over these structures will get entangled and break. To be effective however the reefs must be built to scale, taking into account both the power of the trawling boats (which are often 2 to 5 times higher than the officially declared) and local environmental conditions (weight resisted by the substrate, current speeds).

Artificial reefs are relatively cheap to construct but require careful installation: they need to be lowered to the seabed from medium or large boats using cranes and under diver supervision. It is customary to install several rows of artificial reefs on the seafloor perpendicular to the coast (and to trawl trajectories) at the same depth as the meadow under threat. Ideally, the reefs should be placed on local meadow clearings to avoid secondary impacts (such clearings are abundant in meadows affected by trawling).

The protective rows may be linked through a line of artificial reefs along the deep meadow limit, or form polygons surrounding meadow patch ensembles (e.g. Maro-Cerro Gordo MPA, Montoro 2007); the polygons are also designed to cut sand trains, the intensity of which has increased following coastal alterations and meadow erosion). The distance between reef modules has to be short enough to impede the passage of trawlers, whilst the distance between successive rows should be 1-2 times the cable length used by trawling vessels. In a lattice net, the most efficient disposition would be one in which modules in one row would not be right in front of the modules in adjacent rows, but, instead, displaced to half the distance between row modules. The location of artificial reefs is usually widely publicised in order to discourage trawling activities in the area.

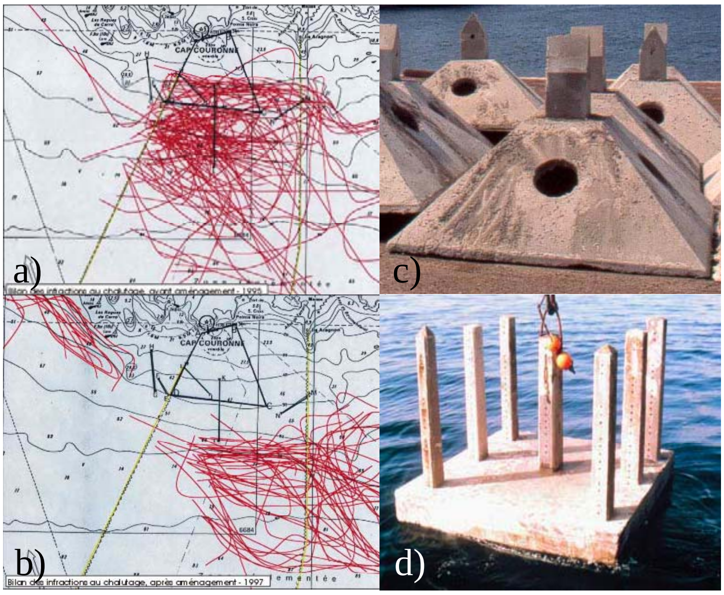


Figure 1. a) trawler trajectories before installation of protective reefs in Cap Couronne and b) trawler trajectories after protective reefs installation. Pictures kindly provided by Mr. Frédérick Bachet (Director, MPA Parc Marin de la Côte Bleue). c) Protective reef model "Sea-rock" and d) "Sotrape" reef.

Protective reefs last for decades and do not need heavy maintenance. However, annual or bi-annual checks are needed to review the working state of the protective reefs and their correct positioning (with side-scan sonar or by divers): experience shows that the reefs installed to date are often too small (4-8 Tm) compared to the real trawler power. As a result, fishermen can move the reefs and continue fishing in the meadows. Reef surveys therefore should be complemented by the monitoring of the protected meadow in order to evaluate their conservation status and assess their eventual recovery (see the seagrass monitoring section). Many reef installation programmes performed to date unfortunately lack such monitoring protocols which makes it difficult to evaluate their effectiveness.

One of the earliest experiences of *P. oceanica* meadow protection with artificial reefs was initiated in 1992 at El Campello and Villajoyosa (Alicante, SE Spain, SW Mediterranean Sea). The seabed in this area was altered by illegal otter trawling: 40% of the *P. oceanica* beds between depths of 14 and 28 m (290 ha, over 7 km of shoreline) had been damaged as a result. In 1992, 540 ha of meadows were effectively protected from trawling activities by the installation of 358 anti-trawling reefs, laid out in 300 meter rows. Reef installation was complemented by a follow-up research program. Eight years later, partial meadow recovery through rhizome growth could be observed. Nevertheless, rhizome growth was 5 times slower than in adjacent non-impacted meadows at the same depth because the light intensity in the impacted meadows was still 4 times lower than in non-impacted areas due to the altered sediment structure. Such low rates of vegetative growth may prolong the time needed for recovery to 100 years (González-Correa *et al.* 2005).

Protective reefs have also had a positive effect on fish populations and on fish yields in surrounding areas. In the Marine Reserve of Cap Couronne (210 ha, France), the deployment of 91 protective concrete reef modules in 1996 was done at a total cost of €102,539 (8 Tm each 2.5×2.5×0.45 m^3, displayed in 5 rows 50 meters apart; their form makes them difficult to detect by boat sonar). These reefs, combined with daily surveillance and partnerships with fishermen's associations, put a stop to trawling activities in the meadow itself, but a part of these activities have since moved into an unprotected area of meadows that was not previously exploited (Fig. 1b). This stresses the need for increasing the spatial scale of protective reef deployment, and of protective strategies in general, to the whole *Posidonia* habitat area, and adjacent sensible habitats like maërl (Hall-Spencer and Moore 2000).

The protective reefs at Cap Couronne, together with some fish-production reefs, have nevertheless increased the faunal diversity in the area as well as the abundance and size of fish catches outside the reserve (Bachet 2006).

Installation of seagrass-friendly moorings

In order to reduce the erosive pressure of free anchoring and mooring in shallow meadows, ecological moorings are increasingly being provided to boat users. When such moorings are available, sailors usually prefer to use them because they are more secure than free anchoring. However, in areas with high tourist pressure, mooring deployment does not suffice in itself, and has to be reinforced by a ban on free anchoring and free mooring. This prohibition and the use of seagrass-friendly moorings require effective control and mooring management. For this reason, they are usually installed within marine protected areas (MPA) that have foreseen specific control measures as part of their management plans. Even in these areas, mooring installation usually requires the approval of the national or regional administrations responsible of coastal management.

Moorings installation must be preceded by detailed preliminary studies to identify the meadows suffering higher boat anchoring pressure and to record the size, distribution and numbers of boats present. Studies on the resistance and thickness of the substrate, and on local hydro-dynamics, help to further optimize mooring locations and design. A useful guide to permanent ecological moorings was recently published to assist site managers (Francour *et al.* 2006). According to this guide, moorings should be preferably installed on meadow clearings, if available. On sandy patches, sand screws are suitable. They consist of a galvanized steel device, made of a shaft with one or several helix-shape discs (Archimedes screw, Fig. 2a).

Sand screws are usually 0.8 to 3 meters long. Beyond this size, their installation becomes more difficult and requires larger boats. It may be more appropriate to install several small sand screws in a row which are connected to form a more resistant mooring. The impact of the sand screw on the sand and mud environment is extremely low because the area occupied by the device is very limited, with only the head sticking out a few cm from the substrate. In addition, no movement of material is produced during the installation of the anchor. These can therefore be readily removed and re-installed in new locations. On large sandy patches, the traditional dead weight moorings may also be used (Fig. 2b) but they require large boats, cranes, etc. to handle and lower them to the seafloor. They also occupy a larger area and there is a risk of sliding and ripping, especially in areas affected by currents, because the volume of the dead-weight above the sea floor generates hydraulic turbulences and produces scouring effects.

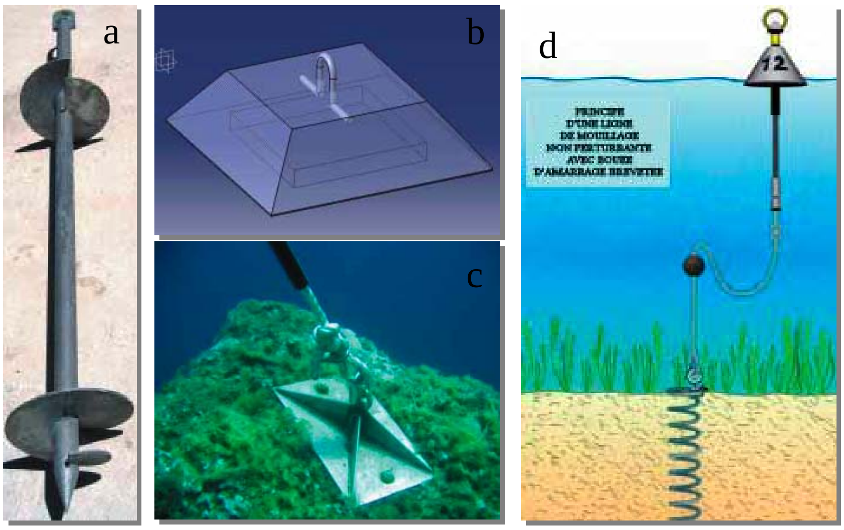


Figure 2. a) Sand screw, b) dead weight, c) grouted anchor, d) Harmony P® anchor penetrating the meadow mat, and intermediate elements of an ecologic mooring. Images extracted from Francour et al (2006) with permission of the authors.

On rocky patches, grouted anchors can be installed. Each one consists of a plate (Fig. 2c) or a single anchor ring (galvanized steel of A4 quality stainless steel) with one or many threaded rods, or ringbolts, resin-bonded into the rock with an under-water injected grout. The rock has to be compact and without fractures. Its impact can be considered negligible because the area occupied by this device is very small (standard plate: 0.15 m^2). However, this anchoring system is not reversible and the anchor parts are not reusable.

In meadows without clearings but with a well developed matte, a special ecological anchor device can be used: this consists of a steel coil anchor, Harmony type P, made of a galvanized steel coil (in the shape of a corkscrew, Fig. 2d) which is designed to penetrate the meadow matte and give a strong anchoring point. The wire of this giant corkscrew finds its own path through the rhizome and debris network without cutting, crushing or destroying the matte. Scouring is avoided if the cork is screwed well enough into the substrate. Therefore, on meadows with a compact matte, the Harmony P anchor would not have negative effects (Francour *et al.* 2006), although, secondary effects of shading and nutrient inputs could be observed. Like the sand screw, its installation is reversible and it can be removed and reused.

The installation of sand screws, grouted anchors and Harmony P anchors does not require important nautical equipment and does not involve heavy equipment or techniques, which helps to reduce secondary damage. They do however require specialised divers: grouted anchors require a hole of the correct diameter and length to be drilled into the rock using an underwater drilling gun (ideally hydraulic). Sand screws and Harmony P anchors need manual screwing, sometimes assisted by hydraulic machines. Using divers has the added advantage of ensuring precise anchor installation at optimal sites (e.g. a small patch of sand within the meadow).

Intermediate mooring elements which attach the boat to the permanent anchor must also avoid impacting on the seabed. Ecological moorings use an immersed float attached at mid depth onto the mooring line. The float's pull ensures that the lower part of the mooring line is taught and vertical and so preventing any contact with the substrate and the plants (Fig. 2d). Additionally, the pull of the float acts as a first shock absorber, while the length and the weight of the hanging chain or elastic rope absorb larger jerks.

The earliest experiments with seagrass-friendly moorings were performed in the 1990s in the National Park of Port Cross (France). In the Cabrera National Park (Spain), free anchoring was forbidden and 50 moorings (consisting of 1.3 to 2.5 meters-wide concrete dead weights, with ecological intermediate elements) were installed over the meadow and in sandy patches, in 1993, in the sheltered bay of Es Port, which supports nearly 380 visitors per day in summer. The global cost of the installation was of €200,000, and the annual maintenance and management costs are around 15% of this initial installation cost (Moreno 2006).

In the Cerbère-Banyuls MPA (France), 30 moorings were installed in 2003 (grouted anchors and sand screws), but free anchoring was not forbidden as the park had a relatively low boating pressure. Two years later, free anchoring was reduced by 90% (Licari, 2006). The installation, maintenance and legal responsibilities for these moorings were contracted to a private enterprise. The costs of installing 15 moorings (project and installation) was of €63,000 (€3,315 per installation of grouted anchor, €3,980 per installation of sand-screw or Harmony P), with an annual maintenance cost of €260 per mooring. Sand screws are removed in Autumn and re-installed each Spring. The use of the moorings is free.

In the strictly protected area of Medas islands (Spain), 54 moorings have been installed and free anchoring has been banned in 1994. The costs of installation and management are largely covered by a small fee of €3.5 per mooring user (Medas 2006).

Recently, a large-scale project (EU Life-*Posidonia*, LIFE00/NAT/E/7303) aimed at protecting seagrass from boat moorings has been implemented around the Balearic Islands. Since 2006, 400 permanent moorings (sand screws, dead weights and grouted anchors) have been installed in the coastal waters of Mallorca, Menorca, Ibiza and Formentera. There is a system for assigning moorings to boats. This task and the review of moorings every year has been contracted out to a private enterprise. However, given the high boat pressure during the summer months, this strategy needs to be reinforced by a clear prohibition of free anchoring and free mooring which is effectively controlled through regular surveillance.

The early mooring experiences in sheltered bays such as Port Cross and Es Port de Cabrera have revealed a secondary problem associated to intensive visits: boat waste which increases organic and nutrient inputs into the surrounding waters and sediments, especially in bays with slow water exchange. Permanent moorings may aggravate this problem because the solid organic waste sinks repeatedly to the same place and the area is frequently shadowed by the moored boat. Meadow decline around moorings in these areas have been observed (Marbà *et al*. 2002). Therefore, the use of permanent moorings, at least in sheltered bays, should be restricted to boats having wastewater holding tanks.

An additional, negative impact could come from the accumulation of fragments of anti-fouling paints from the bottom of moored boats. The poisonous effects of these products have been demonstrated for many species, although they are not clear yet for *P. oceanica*. Nevertheless, a large accumulation of anti-fouling residue may affect the fauna of this habitat.

Management of stranded seagrass litter

Beach management consisting of litter removal may be detrimental to the stability of both the beach and of the shallow meadows. Some alternative practices have been implemented with apparent success. The best practice is a no-removal policy of beach cast material. However, in heavily-used beaches this is not always possible. In such places, no-removal periods should be set to encompass the largest amount of time possible, taking advantage of the low tourist-season.

At the same time, public information campaigns on the benefits of seagrass litter and on the fact that it is an indicator of good environmental quality can be developed, for example through the distribution of leaflets among beach users. Ideally, these efforts should reduce the pressure to remove the seagrass litter. This demand can be monitored using questionnaires that assess the level of materials users are willing tolerate before considering it a nuisance.

When removal is inevitable, heavy machinery should be avoided and the material should not be moved away from the beach. The least damaging practice may involve mixing the seagrass material with sand so that it can be buried out of sight, thereby avoiding losses of beach sand and seagrass-associated minerals.

Control of invasive species

Given the large growth potential of invasive species, total eradication is almost impossible but the control of their populations to limit their negative impact can be attained with recurrent management strategies. In the case of *Caulerpa taxifolia* in the Mediterranean basin, recurrent management actions to contain invasion have been carried out since 1992. The European Community supported 4 LIFE projects to monitor and control the expansion of this alien species along the Mediterranean coasts of Spain, France and Northern Italy (LIFE92ENV/F/0066, LIFE92ENV/E/0067 and LIFE92 ENV/IT/0068 "Proliferation of the tropical algae *Caulerpa taxifolia* in the Mediterranean").

The project "Control of the *Caulerpa taxifolia* extension in the Mediterranean Sea" (LIFE95ENV/F/782) involved eighteen partners, including national and regional governments. In this project, various techniques for controlling these seaweeds were developed, tested and proposed at a local, regional and international level.

The following is a description of the strategies used by the National Park of Port Cross (France, Houard 2007), at the local level. This park has a long experience of controlling the invasive alien algae *C. taxifolia* (since 1994) and recently *C. racemosa*:

- Partnerships with fishermen, diving clubs and other sea professionals were created to promote working practices that reduce the risk of spread. The partners also report their incidental observations of new points of growth to a coordinator agency.
- Continuous monitoring *of P. oceanica* meadows (especially areas in decline) was carried out to measure population densities of invasive species and to chart their presence and extent.
- Systematic surveys and eradication programmes were launched every year. Such programmes are performed during the period of maximum algal growth (late Spring-early Summer) by a team of 40-50 people (coastal managers and volunteers, usually from diving clubs). The systematic survey of the target meadows is carried out over 4 to 5 days. Divers swim in prospective fronts, tracing parallel trajectories at 5m intervals from each other (the total front width is around 50 meters). When small patches of *C. taxifolia* (and now also of *C. racemosa*) are found, they are removed and kept in closed bags, taking care not to leave algal fragments behind in the environment (large colonising potential).

  When large patches are found, they are recorded and later covered with opaque blankets which starve the algae and cause them eventually to die. Such eradication programmes cost €12,600 in 2006 (eradication from the Islands of Port Cross and Porquerolles), which included travel, accommodation and food for volunteers, as well as materials and petrol. The work was financed by private agencies (foundation of the French company Total) and public funds. This is a recurrent strategy which does not eradicate the invasive species but maintains their population under a certain level of control.

- Periodic information to the public on the problem of *C. taxifolia* invasion and how to behave when they find them in nature was also carried out.

At the international level, a mathematical model of the spread of *Caulerpa* and a strategy for its control in the entire Mediterranean Sea was developed. This was developed from surveys of sensitive areas and Natura 2000 sites which monitored the spread of the so called "killer alga" and identified new populations.

The project included a large-scale information campaign destined to inform governments and all sectors involved (i.e. fishermen, divers, pleasure craft crew) of the need to control the invasion of *C. taxifolia*. Multi-language leaflets and posters as well as a video were produced and distributed in eight Mediterranean countries (Spain, France, Italy, Malta, Croatia, Tunisia, Algeria and Turkey). The effectiveness of this campaign was remarkable: tourists and residents contributed to the discovery of new colonies of *C. taxifolia* which were subsequently removed, thereby slowing-down the spread of the "killer alga" in the Mediterranean Sea.

The stagnation and, in some places, even regression of *C. taxifolia* populations (maybe produced by the transference of pathogens from the native *C. prolifera*) has further helped to control the spread of this seaweed. Researchers believe that this collapse may be due to the limited genetic diversity of the introduced species. On the island of Mallorca, the General Direction of Fisheries (DGP), responsible within the project LIFE-*Posidonia* (LIFE00/NAT/E/7303) for controlling and eradicating populations of *C. taxifolia*, finally decided to stop eradication efforts and change strategy as the local *C. taxifolia* populations appeared to be stagnating or even regressing around the Island. Instead, the DGP developed a public information campaign and monitoring programme on existing populations of *C. taxifolia* (and recently *C. racemosa*) through the *P. oceanica* Balearic monitoring network, a partnership of volunteers from diving clubs and students coordinated by the DGP.

A different scenario is observed for populations of *C. racemosa*. This species has been spreading actively around Mediterranean coasts since 1990 (like *C. taxifolia* but not limited to *P. oceanica* meadows). The impact of *C. racemosa* on algal assemblages is higher than that of *C. taxifolia*. In contrast to *C. taxifolia*, *C. racemosa* is continuously re-introduced into the Mediterranean from the Red Sea through the Suez Canal. This increases the seaweed's genetic diversity pool in the Mediterranean and enhances the vitality of existing *C. racemosa* populations.

Many programs are now monitoring *C. racemosa* populations. The effects of this new invasion on Mediterranean benthic communities seem to be even more serious than those of *C. taxifolia* (Piazzi *et al*. 2003). The eradication of *C. racemosa* also appears to be more difficult because it has a faster regeneration capacity, and because frequent removal has to be complemented by vacuum removal systems (Cecherelli and Piazzi 2005). Therefore, the control of this invasive species urgently needs a coordinated effort on a scale at least as important as the one deployed for *C. taxifolia* in the past.

Nevertheless, it should also be considered that *Caulerpa* spp. appear to invade only *P. oceanica* meadows that are already in decline. For example, in the National Park of Port Cross, the largest populations of *Caulerpa* spp. are found in areas cleared by boat trawling. Conserving seagrass density and bed size can enhance the meadow's resistance to introduced species.

Some other invasive species, like *Lophocladia Lallemandii,* have also demonstrated deleterious effects on *P. oceanica* meadows, but assays and eradication protocols do not exist yet for these species.

Dredging recovery

In Capo Feto (SW Sicily, Italy) a gas pipeline trench destroyed 150 ha of *P. oceanica* bed and affected adjacent meadow density and growth (Badalamenti *et al*. 2006). The trench was partially back-filled with rubble from dump barges which lead to the formation of a series of rubble mounds on the seabed. The coarse materials prevented siltation and enabled light conditions to return to normal after installation. In 10 years, these mounds became partially colonised by vegetative fragments of *P. oceanica*. In rubble-mound valleys, such fragments coalesced. At 15 meters depth, the valleys even reached shoot densities equivalent to those prior to the installation of the pipeline. But the plants did not progress on the sides and crests of the rubble mounds, due to stronger hydrodynamics in these areas (Di Carlo *et al*. 2005).

Transplanting

Restoration programs have been introduced for other seagrasses, especially for *Zostera marina* in the USA and Australia, with variable degrees of success (Calumpong and Fonseca 2001). In the case of *P. oceanica*, some experiments have been performed in France, Italy and Spain: most have failed due to the slow growth rate of the species and the lack of knowledge. Even if successful, transplant restoration of *P. oceanica* has to be considered over a long time frame, requiring active recurrent management over several decades.

For these reasons, the restoration of *P. oceanica* meadows cannot be considered as a measure, justifying the destruction of existing meadows, as it has been the case in Campomanes (Spain). Here a project to enlarge a pleasance harbour has been accepted in exchange of a "compensatory" project, in which the meadow to be destroyed was going to be transplanted to an adjacent area. Seven months later, shoot survival within the transplanted area was only of 15%, which, at any rate, can be considered a failure (Sánchez-Lizaso *et al.* 2007). Nevertheless, the development of transplanting techniques to accelerate re-colonisation in areas where *P. oceanica* has been already lost is desirable, provided certain guarantees are in place.

Any transplanting project should be considered within a global frame of seagrass management, taking into account: (1) the total existing meadows surface, (2) the annual rate of meadow surface loss and its causes, (3) the annual rate of natural meadow surface progression (if existing), (4) the expected meadow re-colonisation through transplanting in 10 to 50 years and (5) transplanting costs, and comparison with the effects of an identical investment in correcting regression causes (water treatments, anti-trawling reefs etc., Gravez and Boudouresque 2003).

Some preliminary recommendations have been drawn up from successful and less successful transplant experiences performed until now. Moreover, a code of good practice in transplanting projects has been adopted by the EU Parliament, within the STOA program (presented in Corfú, Greece, in September 1993, Boudouresque 2003). A synthesis of all this is given here and in Gravez and Boudouresque (2003), but technical recommendations should be taken with caution, as most conclusions are only based on a few, small-scale experiments, involving 200 to 1000 shoots. As a general rule all sites and strategies considered for potential restoration should always be tested in advance using experimental, small-scale plantings to ensure their suitability before any major restoration projects are launched.

- <u>Evaluation of the site to be restored</u>*:* before any transplantation is performed, it is essential to determine if the environmental conditions of the candidate site can support plant growth again (Calumpong and Fonseca 2001). The plant's requirements concerning light, nutrient levels, sedimentation rate, sediment type and quality (sand and organic matter content, sulphide and oxygen conditions), substrate stability, current intensity, wave exposure, temperature and salinity and potential herbivore pressure, should all be within the range acceptable for *P. oceanica* before any transplant is initiated. It has been shown that the dead mattes of disappeared *Posidonia* meadows are very suitable substrates for transplanting. The meadows adjacent to the area to recover (or the remaining tufts) should be in good health and iniciating a re-colonisation process, not regressing.

- <u>Donor meadows</u>: when selecting a donor meadow, the main factors to take into account are meadow health (higher rhizome reserves in cuttings) and genetic diversity (heterozygosity and number of alleles). Transplanting a genetically diverse population of *P. oceanica* increases its success (Procaccini and Piazzi 2001). In order to minimize pressure on donor beds and to maximize genetic diversity in the new site, donor plants should be collected from as many meadows as possible. But, as a matter of precaution, meadows should be selected at the intra-basin scale, because of the genetic cleavage observed between Eastern and Western Mediterranean populations (Arnaud-Haond *et al.* 2007). Increasing the number of donor sites, and the distance between them, also increases the cost of the project.

  Of decisive importance is the depth of donor areas with respect to the site to be restored: plants transferred from low water depths to higher depths have shown a very low survival rate while plants from deeper water survive quite well in shallower waters (Molenaar and Meinesz 1992, Genot *et al*. 1994, Piazzi *et al*. 1998).

When a given area is selected as a potential donor site, a preliminary study of its shoot population dynamics should be performed, in order to be sure that it is stable, and to set the maximum number of shoots and/or apices that can be collected. As a general rule, no more than 1% of shoot density (or of apex density) should be collected. This is the level at which the loss can be compensated by the plant's annual vegetative recruitment in healthy meadows. Moreover, shoots should be collected selectively and in a sparse way, to prevent local reductions in shoot density and cover and to prevent the opening of erosion fronts.

This necessitates the use of divers to collect shoots by hand, which increases the overall cost of the transplantation. For this point and the following, it is important to take into account the legal regulations protecting this species and habitat. Before starting the activity, collection permissions have to be obtained from the local authorities.

- Plant material: plants do suffer physiological stress when transplanted, as reflected in a reduction of their chlorophyll and carbohydrate contents (Genot *et al*. 1994). Cuttings with a horizontal apex and two lateral branches constitute the most active parts of the plants and have shown higher survival and branching rates in experimental transplants (Molenaar *et al.* 1993, Piazzi *et al.* 1998) of *P. oceanica.* However, as horizontal apexes are not abundant and are vital for donor meadows, vertical shoots can be chosen as alternative transplants. Survival of vertical shoots with two leaf bundles did not differ from that of horizontal apexes in one experiment (Molenaar *et al.* 1993), but they had to switch to horizontal growth before initiating active growth and branching.

  Vegetative shoots naturally detached from meadows by storms have also been used successfully as transplant material (Augier *et al.* 1996). They are available all year round, their collection is cheaper and, above all, it does not impact on donor meadows; but they should have at least one leaf bundle, at least 8-12 cm of rhizome (carbohydrate reserves and antibiotic substances) and good signs of vitality to serve as transplant material (Meinesz *et al*. 1992). The time elapsed between collecting and planting and the exposure of cuttings to the air have to be minimised to reduce transplant mortality (Calumpong and Fonseca 2001).

  Transplanting seedlings could also avoid impact on donor sites. Unfortunately, *P. oceanica's* sexual reproduction is rare and seeds do not enter into dormancy, which hampers the establishment of a seed bank. Nevertheless, widespread flowering and fruiting generally occurs after especially warm summers (Díaz-Almela *et al.* 2007). Seeds of *P. oceanica* float and can be found in great numbers along coasts in the spring that follows a summer heat-wave (Balestri *et al*. 2006), which makes them cheap to collect.

  Large seeds are more abundant at the end of season and have more carbohydrate reserves. The seeds germinate very well in tanks containing seawater (70-80% germination success). Here they can survive and grow under suitable conditions allowing the development of a nursery for later transplantation (Balestri *et al*. 1998). Experiments show that seedlings and cuttings treated with auxins develop two to three fold more/longer roots, which could enhance their transplanting success (Balestri and Bertini 2003, Balestri and Lardicci 2006).

- Transplanting techniques*:* shoots attached to plastic or nylon nets (of 25x25 or 60x17 cm^2, 1cm^2 mesh), which are in turn attached to the substrate with metal sticks have been used with variable success to transplant vegetative fragments or seedlings of *P. oceanica* in small scale experiments (Molenaar *et al*. 1993, Balestri *et al.* 1998). Seedlings were protected with cheese clothes before being attached to the nets and the apices were made to point out of the net. The number of plants within the net has varied from 5 to 18 plants (spacing between plants ranging from 17 to 3cm) in different transplanting experiments, resulting in similar success rates. The cost of using this technique on a large scale would be high in its present technological state because the plants need to be attached by hand to the net and extra diving time is required to secure the nets to the seabed.

  Planting individual cuttings attached to staples has been assayed with other seagrasses (Calumpong and Fonseca 2001), but has not yet been tried out on *P. oceanica*. Another technique that has been tested consists of cutting meadow blocks of live seagrass and the rhizome net (matte) from donor sites (1m^2 surface, 0.40 m high). However, shoot survival rate in these blocks was very low: 15% after 7 months (Sánchez-Lizaso *et al.* 2007). Moreover, the removal of such blocks may induce erosion in the donor areas.

A technique developed by the Cooper Foundation consisted of securing rhizomes on rectangular concrete frames (0.0676 m^2) between two pieces of superimposed wire netting (Augier *et al*. 1996). These frames are easy to handle and may reduce costs, since they can simply be lowered from the boat and do not need any attachment to the sediment. However, their success was variable. For example, they do not work on soft bottoms because the heavy structures sink into the sediment.

- Transplanting season*:* one experiment indicates that September would be the best month to collect and transplant *P. oceanica* cuttings (Meinesz *et al*. 1992). Their carbohydrate reserves have been accumulating during summer and are maximal in early autumn. The plants could therefore produce more roots and anchor before the next growing season. The best season for transplanting seedlings is unknown but could be similar (seedlings attach naturally from June to October).

- Follow-up: the first year after transplantation is crucial for plant survival as it must adapt to new environmental conditions and develop roots to anchor itself onto the substrate. The largest transplant losses usually take place the first year. Survival rates of *P. oceanica* seedlings were highest on dead matte (70% at 10 meters, 38% at 2 meters, after 3 years) and did not survive on pebbles or gravel (Balestri *et al.* 1998), but had intermediate survival rates on rocks (46% at 10 meters, 0% at 2 meters after 2 years, Piazzi *et al.* 1999). Survival of vegetative fragments on dead matte was similar to that of seedlings (76% of horizontal transplants and 59% of vertical transplants, at 10 meters after 3 years, Piazzi *et al.* 1998). However, seedlings take more time to develop branches compared to vegetative fragments: after three years, 87% of horizontal, 37% of vertical cuttings (Piazzi *et al.* 1998), but only 14% of seedlings, had branched (Balestri *et al.* 1998).

  The follow-up of a restoration project is crucial. Nevertheless, this phase has been neglected or is in most cases too short. For *P. oceanica*, a restoration project requires several decades of follow-up and reinforcement. The largest *P. oceanica* transplanting experiences performed to date were undertaken between 1972 and 1984 by the Cooper Foundation: 70,000 *P. oceanica* shoots were transplanted at different sites. But most of these early trials failed due to lack of experience, or their effectiveness remains unknown because they were not followed up.

  Nevertheless, one of the planted sites (in the Bay of Cannes planted in 1984 where transplants consisted of naturally detached shoots, within concrete frames deployed in 1ha) was revisited 10 years later and showed relative success: the transplanted area had increased seven times and shoot numbers had increased nine times. The plants formed oblong islets with numerous running rhizomes, indicating active colonisation. But leaf epiphytes were three times more abundant than on adjacent meadows (Augier *et al.* 1996).

  Table 1 recommends relevant parameters and follow-up frequency for a *P. oceanica* restoration project.

Table 1. Monitoring transplanting success. Adapted from Borum et al (2004)

	Start-up	Follow-up
Population parameters	-Survival -ramification rate (new shoots) -rhizome elongation -root production (if feasible)	-coverage -shoot density -patch size
Frequency of measures	- 1st year : each 4 months - 2nd and 3rd year : each 6 months - 4th and following years: annual	Until patch coalescence or targeted cover and density: visits each 2 to 3 years
Environmental 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, herbivore pressure	
Frequency	- At least three times during the project : (1) before transplanting, (2) at an intermediate stage, and (3) at the end of the project	

Other relevant measures

Regulation

P. oceanica meadows are protected at the European level, as a priority habitat (Dir. 92/42 CEE 21/05/92 and 97/62/CE 27/10/1997) and as an species (Bern Convention, Annex 1). Bottom-trawling is expressly forbidden on seagrass meadows (Fishing regulation 1626/94). At the national and regional levels, *P. oceanica* meadows are protected in Spain (RD 7/12/1995, BOE nº310) and France. *P. oceanica* is also protected as a species in France (Arreté Ministeriel 19/07/1988) and in Catalonia (Spain, Orden 91.210.098 DOGC nº 1479 12/08/1991), where all seagrass species are protected. In Valencia and the Balearic Islands (Spain), bottom trawling on seagrass meadows is explicitly forbidden since 1993. Aquaculture facilities over seagrass meadows are also forbidden in the Balearic Islands (Orden nº 19611 21/09/1993).

Unfortunately, most legal texts and conventions are too often eluded. The legal protection of *P. oceanica* as a species, has proven more effective and restrictive than its protection as an habitat, in face of the strong pressures often made by local authorities on national an European environmental laws (Crebassa 1992). This is because the minimal meadow surface to be considered an habitat worth protecting is not clear, and this is used as way to elude the law, while with the legal protection of the meadow-forming species, there is no minimal size to be formally protected.

The specific protection of the millenary barrier reefs of *P. oceanica* still remaining has been recommended and settled in some sites, like the barrier reef of the Port Cros National Park (France) and another in Roquetas the mar (Spain).

EU regulations on urban and industrial effluent treatments, coastal construction and meadow protection are still not well developed and/or implemented in most Mediterranean countries. So, only the meadows situated in marine protected areas are under active specific protection. The new approach of the EU Water Framework Directive, which aims to maintain water body standards at levels that are compatible with surrounding habitats, could be a great regulative tool if applied correctly.

A first priority in coastal conservation should be to maintain benthic nutrient sedimentation rates on meadows under 0.03 Nitrogen and 0.04 g Phosphorus m^{-2} day^{-1}, respectively (Díaz-Almela *et al*. in press.). In order to meet these objectives, urban and industrial sewage must be systematically diverted to treatment plants which have an efficient means of removing organic matter and nutrients. The establishment of narrow zones of uncultivated soils along streams and rivers, together with the protection of undisturbed wetlands capable of intercepting agricultural nutrient runoffs and reducing deforestation/erosion-derived siltation, would contribute significantly to preserving many coastal ecosystems, among them *P. oceanica* meadows.

To prevent loading from fish farms to meadows, cages should not be allowed in bays. In open deeper waters, a minimal security-distance of 800 meters between fish cages and the meadow border should be respected (Marbà *et al*. 2006). In any case, distance and effluent loads should be regulated so that organic inputs into *P. oceanica* meadows never surpass 1 and 2 g m^{-2} day^{-1} in rocky and sandy bottoms respectively (Díaz-Almela *et al*. in press.).

Outlets with high salinity from desalination plants (which are proliferating in Mediterranean countries) should not produce salinity levels in or near a *P. oceanica* meadow above 38.5 psu in more than 25% of meadow water samples (and never higher than 40 psu in more than 5% of water samples (Autores varios, 2006). Outlets of cooling seawater from power plants should not increase meadow seawater temperature by more than 1ºC above the mean seawater temperature in the coastal region.

Although coastal infrastructures of public interest often have priority, measures to minimize impacts on coastal currents and sedimentation/erosion balances are imperative. Dredging and sand reclamation on or close to meadows is banned in most EU national legislations but recent experiences indicate that sand reclamation activities should be accompanied by a more effective control in order that the security distances are respected. In order to settle an effective regulation system, complete cartographies of the habitat 1120 are needed.

Meadow monitoring

Seagrass monitoring is a fundamental tool for measuring the status and trends of meadows and environmental conditions. In the case of the slow-growing *P. oceanica* seagrass, it is crucial to early detect decline trends. Monitoring is also essential in any protective or recovery initiative, in order to address its effectiveness. Programmes have been launched since the 1980's to address the spectacular seagrass losses witnessed across the world. Presently, more than 40 countries have developed monitoring systems for 31 seagrass species. Some programs are even trans-national. The number of monitoring programs on *P. oceanica* meadows has increased in recent years but they are less developed than for other species, like *Zostera marina*, and, in general, remain regional or national initiatives. Their differing methodologies also make it difficult to obtain a comprehensive view of the general status and trends of the underwater meadows across the Mediterranean.

Information on existing *P. oceanica* monitoring networks in the EU, their protocols, results and contact details are available on several web pages (summarised in table 2). All of these monitoring programmes rely on a combination of volunteers, technical personnel and scientists. Volunteer-based networks create a culture of community support for seagrass protection and for the wise management of coastal habitats. This social role is as important as the information that these programmes deliver. Nevertheless, they require a clear leadership if they are to be sustained over time. Volunteers must also be motivated by the prompt delivery of results and activities that encourage communication among them.

The first basic tool for a seagrass monitoring system is an inventory of the meadow's location and distribution. Meadow maps are basic managerial tools, providing an overview of the habitat's status to administrators and the public. Presently, there are systematic cartographic maps of *P. oceanica* beds off Liguria (Italy), Mediterranean French and Spanish coasts, as well as other maps around the rest of the Mediterranean. Successive mappings enable meadow changes to be detected on a large scale and help to identify conspicuous impacts, like sediment redistribution or colonisation by other species. Linked to other environmental data-bases (water quality etc.), they may also help to ascertain the cause of meadow decline thereby facilitating corrective actions, or to forecast meadow distribution in future environmental scenarios.

Table 2. Summary of existing P. oceanica monitoring networks

Monitoring network	Web site
Murcia (Spain)	http://www.carm.es/neweb2/servlet/integra.servlets.Control Publico?IDCONTENIDO=1327&IDTIPO=100&RASTRO=c494$m
C. Valenciana (Spain)	http://ecologialitoral.com/volunt.htm
Islas Baleares (Spain)	http://lifeposidonia.caib.es
Catalunya (Spain)	http://www.gencat.net/darp/faneroga.htm
GIS-Posidonie, (France)	http://www.com.univ-mrs.fr/gisposi/

There is a wealth of methods for seagrass mapping. Most need cumbersome or expensive techniques and have to be performed by trained personnel. *In situ* charting can be made through systematic diver observations, grab samplings, video or sonar. The first method is worthwhile only for a small scale (<1ha), providing detailed data on distribution and change within a small meadow. Other methods are needed to chart seabeds at medium or large scales (1 to 100 km^2). Remote-sensing is particularly useful: aerial photos enable detailed meadow mapping on a broad range of scales (<1ha- <100km^2). Coarse presence/absence and area distribution maps can be obtained from scanners (1ha->100 km^2) and satellites (1 to>100 km^2). Remote sensing data does however need to be complemented by ground surveys to confirm the meadow's characteristics. The delimitation of deep meadows is often not visible in remote-sensing images, but may be seen on data acquired using a CASI scanner or side-scan sonar. A guide to choosing appropriate remote sensing methodology is available on the homepage of the EU LIFE project Rescoman (http://www.dmu.dk/rescoman).

Monitoring of the upper and lower meadow limits delivers robust indicators of overall meadow distribution as most stresses will usually be detected first along meadow borders (water clarity affects the limit of depth and erosion or burial affects the upper limit). The deep meadow limit is a comprehensive indicator and has a high priority in monitoring programs as it can help assess the effects of eutrophication and siltation on *P. oceanica* meadows. However, when meadows reach their deepest species range (45m in the clearest Mediterranean waters), monitoring may require the use of professional divers.

Deep limits can be monitored swimming along the meadow border, installing permanent metal sticks at regular distances and recording their precise depth. It must be clear whether the border refers to the limit of meadows or to the deepest individual shoots. In the case of the former, the meadow must be defined precisely (e.g. the maximum depth where seagrasses cover 10% of the surface). In subsequent visits, the new meadow limit with respect to the sticks is recorded and, if it has changed, another metal stick is installed at the new site. Deep limits can also be monitored using bathyscaphs and precise geo-referencing methods.

Seagrass abundance (biomass, coverage, shoot density) shows a characteristic exponential decline with depth and is therefore also a good monitoring indicator. This pattern is sensitive to changes in environmental conditions. The most widely used abundance parameters in *P. oceanica* monitoring are coverage and shoot density along random or fixed transects (corridors of a fixed width through which the diver swims) and/or quadrates of a fixed size. However, when these parameters are measured at random within the meadow, they show a large degree of uncertainty due to the natural patchiness of seagrass growth. In this case only relatively abrupt changes (>20%) can be identified. For the slow-growing seagrass *P. oceanica*, a change of this magnitude is already too big as it may take many years for this seagrass to recover from large impacts.

Some *P. oceanica* monitoring programmes, such as those used by regional networks in Murcia and the Balearic Islands, have adopted a new strategy consisting of measuring shoot density changes through direct shoot counts in permanent quadrates. This allows the detection of early decline in slow-growing species (Marbà *et al.* 2005), because it eliminates the uncertainty of patchiness. However, it is usually done on a very small fraction of the meadow (0.016-0.025 m^2 per plot). In large meadows, several quadrates may need to be installed at different depths/areas so that the net population growth rate of each area can be estimated.

Thus, in these monitoring networks, there is an inventory of quadrates visited at annual or bi-annual intervals. In some of them, a more detailed picture of shoot population dynamics is obtained by performing shoot censuses where all shoots within the permanent plots are marked with small plastic cable ties. In the subsequent visit the young, unmarked shoots can be detected and marked with a new colour. This makes it possible to estimate the rate of shoot recruitment and mortality within a population and within shoot cohorts, which in turn provides a finer diagnostic level for assessing meadow dynamics. However, these detailed censuses require a lot of diving hours and well-trained personnel.

The long-lived vertical shoots of *P. oceanica* (up to 4 decades) elongate each time their apical meristem produces a new leaf, leaving a rhizome internode. Internodal length has an annual cycle and responds to sediment burial, erosion and other perturbations, like heatwaves or pollution (Duarte *et al.* 1994). Therefore, an examination of the pattern (e.g. cyclical, sustained trend, discontinuities) and magnitude of inter-annual variability in vertical rhizome elongation allows the identification of periods when seagrasses have been disturbed, and consequently offers an insight into the source of that disturbance.

The internodal length is also an early indicator of meadow decline. This method has been useful in assessing the early impact of fish farming activities on *P. oceanica* meadows growing up to 400 meters away from the cages (Marbà *et al.* 2006), as well as the identification of the deleterious effects of heatwaves on seagrass growth (Mayot *et al.* 2005). Collection of vertical shoots is easy, but it is a destructive sampling method (although 10-15 long shoots per station are enough) that requires laborious lab work (peeling rhizomes and taking precise measurements of internodal lengths).

Most programs combine seagrass monitoring with the monitoring of environmental parameters, which are usually measured more frequently. Table 3 summarizes the most relevant environmental parameters, their recommended time frame, and offers some links to protocol references. Submersible temperature data loggers are now widespread and relatively cheap. Installed within the canopy, they can record daily meadow temperatures. This is particularly useful in identifying the duration and intensity of heat-waves which are known to strongly affect *P. oceanica* meadows. Light data loggers also exist but they require the frequent cleaning of fouling films which can be impractical.

Total organic and phosphorus sedimentation rates are great predictors of meadow dynamics. They are easy to measure using benthic sediment traps. Water transparency, the most integrative and robust indicator of water quality, can be easily monitored from boats, lowering a secchi disc (very cheap device), or underwater with more sophisticated light meters. An easy and integrative indicator of sediment red-ox conditions is the depth of the oxic front. It can be measured by inserting a thin silver bar in the meadow sediment. The silver becomes black when reduced. Thus, the length of intact bar corresponds to the depth of the oxic front. Finally, the abundance of indicator species (e.g. *Pinna nobilis, Paracenthrotus lividus*) is a good measure of meadow health (see the habitat features section).

Table 3. Some environmental parameters relevant to the stability of P. oceanica meadows

Parameter	Method	Frequency	Link / reference
Temperature	Temp. data logger	1 – 12 hours	e.g. www.onsetcomp.com
PAR irradiance	Light data logger	1 min-1 hour	e.g.www.alec-electronics.co.jp
Total, organic and nutrient input rates	Benthic sediment traps	Monthly-seasonal	- Gacia *et al*, 1999 - www.medpan.org
Water transparency	Secchi disc	Monthly-seasonal	e.g. www.globe.gov/tctg/sectionpdf.jsp?sectionId=149
O_2 and nutrients		Monthly-seasonal	
Sediment redox front	Silver bar	Seasonal-annual	Frederiksen 2005
Indicator species	Censuses, sampling/resampling	Seasonal-annual	www.medpan.org/_upload/996.pdf

Remediation of meadow sediments loaded with organic matter

Sediments loaded with organic matter shift to reduced anoxic conditions that persist well after organic inputs have stopped. When sediments are rich in iron, the toxic hydrogen sulphide produced in these conditions reacts with this metal, precipitating as pyrite which is innocuous to the meadow community. *P. oceanica* meadows are very sensitive to labile organic matter and to hydrogen sulphide because they typically grow on carbonate sediments which are iron-poor and thus have a low buffering capacity against this toxic substance.

A pilot remedial experiment was conducted in Es Port Bay (Cabrera National Park, Balearic Islands, Spain) on a meadow impacted by excessive organic load. The experiment consisted of injecting iron chelates (Fe-EDDHA) into the carbonate meadow sediment over a limited area ($9m^2$). After 2 years of semestrial injections which produced pulses of iron concentration in the first 30 cm of sediment at 0.8 mol iron m^{-2}, the hydrogen sulphide had dissolved in the sediment and plant tissues were reduced. Meanwhile the seagrass showed greater shoot recruitment and recruit survival than control plants, leading to a reversal of the seagrass's decline at the end of the experiment (Marbà *et al.* 2007).

The plant tissues were also richer in iron, indicating that plant growth in the carbonate sediment could also be iron-limited. The experience has not been assayed yet at the scale of an entire meadow as iron injection in its present technical state would be too laborious to introduce into large areas. Two possible trials have been proposed: iron shavings could be seeded to the seabed from boats. In the case of fish-cage organic loading, a possible palliative measure could be to enrich fish feed with iron. Iron excess would be excreted in fish pellets so that the problem would also bring with it a partial remedy. However, such strategies have still to be tested for their effectiveness and for the absence of any secondary negative effects.

Special requirements driven by relevant species

The large bivalve *Pinna nobilis* is a strictly protected species (Annex IV of Habitats Directive) with very slow growth rates. The collection of this species is forbidden and control measures and information campaigns have to be further developed. However, the most important measure to conserve the species is the maintenance of healthy seagrass meadows with which it is strongly associated.

Cost estimates and potential sources of EU financing

Relevant parameters for estimating costs have already been described for most of the measures mentioned in this model, and examples from practical experiments carried out at some sites provide further indications of costs (e.g. seagrass-friendly moorings installation). In general, conservation and monitoring actions in the marine environment require particular efforts and specialised equipment (e.g. boats). Some activities also require a great deal of diving and trained staff time.

The European Commission has proposed an ambitious strategy to protect the marine environment across Europe more effectively. The EU Marine Strategy will constitute the environmental pillar of the future EU maritime policy (http://ec.europa.eu/environment/water/marine/index_en.htm). It aims to achieve good environmental status of the EU's marine waters by 2021.

Conservation management actions on *P. oceanica* meadows have been financed by the EU's LIFE-Nature funds (e.g. LIFE *Posidonia* in the Balearic Islands, Spain). The new LIFE+ instrument offers similar possibilities for the future. Projects funded under LIFE+ 'Nature and Biodiversity' should specifically contribute to the implementation of Community policy and legislation on nature and biodiversity, such as the Habitats and Birds Directives. They should also support the further development and implementation of the Natura 2000 network in the marine environment in particular.

The following measures may be financed under LIFE+ 'Nature and Biodiversity':

- Site and species management and site planning, including the improvement of the ecological coherence of the Natura 2000 network;
- Monitoring of conservation status, including setting up procedures and structures for such monitoring;
- Development and implementation of species and habitat conservation action plans;
- Extension of the Natura 2000 network in marine areas.

Other actions in marine areas (e.g. surveillance and monitoring in marine reserves) were financed in the past under the Financial Instrument for Fisheries Guidance (FIFG). In the new EU financial period (2007-2013), the European Fisheries Fund (EFF) will provide aid for actions that could contribute to the conservation of *Posidonia* beds for instance by:

- Financing equipment and strategies for reducing the impact of fishing on ecosystems and the sea bottom (art. 25);
- Payment of premiums for fishermen and owners of fishing vessels involved in small-scale sustainable coastal fishing in order to improve management and control access to certain fishing areas (art.26);
- Diversification of activities, to promote multiple jobs for fishermen (art. 27);
- Productive investments in aquaculture; implementation of aquaculture methods that substantially reduce their negative impact or enhance positive effects on the environment (art. 29);
- Sustainable aquaculture compatible with specific environmental constraints resulting from the designation of NATURA 2000 areas in accordance with Council Directive 92/43/EEC (art. 30);
- Improved management and control of access to fishing areas, in particular through the drawing up of local management plans approved by the competent national authorities (art. 37).

The EFF may also support measures of common interest that are intended to protect and develop aquatic fauna and flora while enhancing the aquatic environment, such as: the construction or installation of static or movable facilities intended to protect and develop aquatic fauna and flora (eg artificial reefs), and the protection and enhancement of the environment in the framework of NATURA 2000 where its areas directly concern fishing activities, excluding operational costs (art. 38).

The EU civil protection financial instrument may also provide funds for actions in the field of Marine Pollution.

Another possible source of funding is the INTERREG IVC Programme (http://www.interreg4c.net/). This EU programme provides funding for all regions of Europe plus Switzerland and Norway (regional and local public authorities) to exchange and transfer knowledge and good practice. Two main priorities are targeted: 'Innovation and Knowledge economy' and 'Environment and Risk prevention'. The programme has a budget of 321 million euros, financed from the European Regional Development fund (ERDF), for the period 2007-2013.

For further information on EU financial possibilities and synergies between funding programs during this period, a Guidance Handbook on the funding of the Natura 2000 network has been produced (Torkler 2007) and translated to all the EU official languages. A web tool (based on that handbook) to easily determine the possible funding for Natura 2000 sites is available in: http://financing-natura2000.moccu.com/pub/index.html. There is also a more general handbook on the EU financing of

Environmental projects (Lang *et al.* 2005), which is available in 5 languages at: http://ec.europa.eu/environment/funding/intro_en.htm

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Projects

LIFE00/NAT/E/7303, Protection of Posidonia seagrass at LICs of the Balearic Islands.
http://lifeposidonia.caib.es/user/home.htm

LIFE92ENV/F/0066, LIFE92ENV/E/0067 and LIFE92ENV/IT/0068, Proliferation of the tropical algae *Caulerpa taxifolia* in the Mediterranean.

LIFE95ENV/F/782, Control of the *Caulerpa taxifolia* extension in the Mediterranean Sea.