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How are seagrass distribution and abundance monitored?

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

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

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

- Predictability in its response to human pressure

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

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

Indicators of seagrass distribution

Presence/absence and area distribution

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

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

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

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

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

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

Mapping of seagrasses by remote sensing relies on the fact that information regarding bottom features shows up as variations in the radiance directed towards the sensor. The signal received by the sensor is, however, made up of contributions from the atmosphere, the water column and the sea bottom, so the bottom signal is not always distinct. Figure 8.1 illustrates this complexity. In clear shallow waters with seagrasses occurring on a light, sandy bottom,

the contours of the meadows can easily be distinguished in remotely sensed images such as aerial photos (Figure 8.2). In general, however, a wide dynamic range of colours in the image is necessary to distinguish the entire spectrum of objects from the lightest (shallow sand bottom, sun glint) to the darkest (e.g. seagrasses, mussel beds and silty bottom in deeper water).

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

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

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

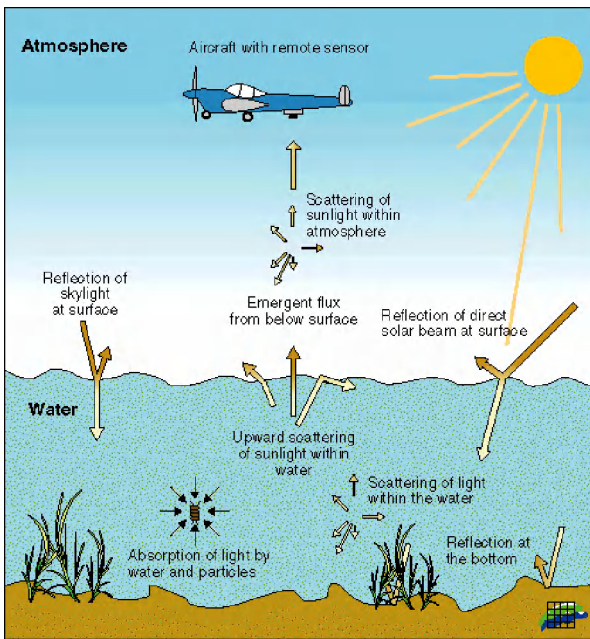


Figure 8.1. Remote sensing of bottom features. From <http://www.dmu.dk/rescoman/Project/Backgrounds/challenges.htm>.

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

Method evaluation: The choice of method for mapping seagrass beds depends on the objectives of the monitoring. When the objective is



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

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

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

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

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

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

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

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

If time series of seagrass area distribution are to be used to forecast seagrass status in future water quality scenarios, relations between water quality and area distribution are needed. A relation between the maximum potential area distribution and water quality may be established based on empirical relations between water quality and depth limits combined with information on the bottom area available for seagrass

colonisation at different water depths in the area in question. Historical information on seagrass area distribution under different water quality regimes in the area may also provide useful input for forecasting future distribution.

Colonisation depths

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

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

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

Advantages	Disadvantages
<ul style="list-style-type: none"> • High spatial resolution • Spatial resolution (as determined by the scale) can be selected based on project objectives • Flexible acquisition – imagery can be planned captured at the most optimal time of day and under the best environmental conditions • Low technology information extraction – seagrass maps can be made from aerial prints or diapositives with little technical hard- or software resources, but in most cases aerial photos should be digitised and rectified before mapping. • Stereometry can greatly enhance mapping. 	<ul style="list-style-type: none"> • Cost – The fine spatial resolution provided by the photographs comes at the cost of obtaining a large number of frames • It is produced in an analogue format and must be scanned if any computer enhancement, image processing or rectifying is anticipated • Distortion – The nature of the camera lens and position, roll, yaw and tilt of the plane introduces some distortion into the imagery. A problem if not corrected by rectifying. • Lack of light can make interpretation difficult in deep and turbid waters • Highly variable sun-glint reflection from all directions in image. • Clouds.

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

Advantages	Disadvantages/potential errors
<ul style="list-style-type: none"> • Enables differentiation between objects whose colour would appear identical to a photo-interpreter • High spatial resolution • Digital from acquisition • Large coverage, easy to georectify 	<ul style="list-style-type: none"> • Photographic distortion • Photo-interpretation • The spectral output of seagrass beds may vary over very short distances due to: <ul style="list-style-type: none"> - growth of epiphytes - "health" of the grasses - water depth - optical properties of overlying water • Low radiometric resolution • Clouds are a big problem

to be documented, however.

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

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

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

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

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

Indicators of seagrass abundance

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

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

of year – for example in August/September where

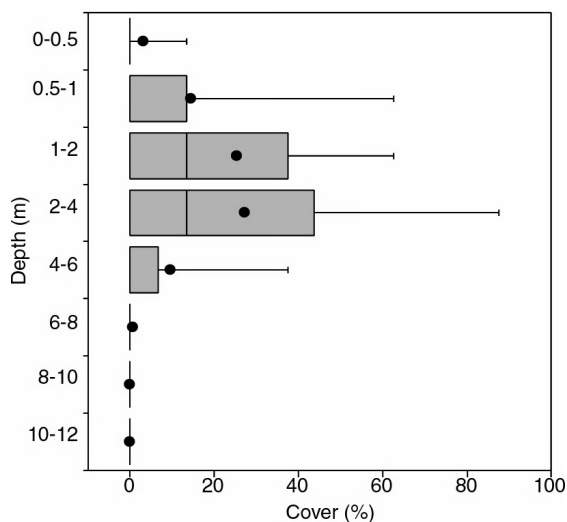


Figure 8.3. Eelgrass cover as a function of water depth. Circles show averages, vertical lines medians, boxes 25-75% percentiles and whiskers 10-90% percentiles of cover observations. From Krause-Jensen et al. 2003. Permission to reproduce the figure has been granted by the Estuarine Research Federation.

biomass usually attains its annual maximum.

Cover

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

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

Method description: As cover is depth dependent, any measure of cover must be related to water depth. The study area can be either coarsely defined as a corridor through which the diver swims, or be more precisely defined as quadrates of a given size. Percent cover of seagrasses is usually estimated visually by a diver as the

fraction of the bottom area covered by seagrass. Both shoot density and shoot length affect this estimate and, consequently, meadows consisting of dense, short shoots may have the same cover as meadows of sparser but longer shoots. The cover can be estimated directly in percent or assessed according to a cover scale. When stones constitute part of the bottom substratum it is important to define whether seagrass cover is assessed relative to the total bottom area or relative to the sandy and silty substratum where seagrasses can grow.

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

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

Indicative potential: Cover estimates are coarse but well suited for surveys at the landscape level and they scale well to environmental factors such as light, wave exposure and littoral slope that operate at scales of 10-10³ meters.

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

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

Biomass

Seagrass biomass is the weight (measured as fresh weight, dry weight or ash-free dry weight) of seagrasses per m² and thereby provides a measure of seagrass abundance along depth gradients. The measure refers to either the total biomass or the aboveground biomass of the seagrasses.

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

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

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

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

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

between biomass and production, the method can be used to estimate total eelgrass production in a given area.

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

Shoot density

Shoot density is the number of seagrass shoots per m² and thereby provides a measure of seagrass abundance along depth gradients.

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

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

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

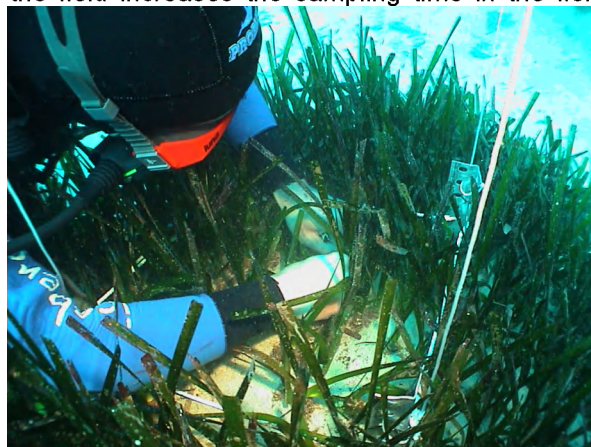


Figure 8.4. Underwater sampling of *Posidonia oceanica* using quadrats. Photo: Jaume Ferrer.

Box 8.1. Sampling strategy for eelgrass abundance - an example

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

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

Sampling was conducted at the time of annual biomass maximum in August-September each year. The programme included assessment of the total eelgrass distribution area using aerial photography and image analysis. Moreover, aboveground biomass and shoot density were determined in six sub-samples of 1/16 m² at each site and percent cover was visually assessed at each site.

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

but requires no laboratory work. As shoot density of eelgrass in shallow water may amount to about 2500 shoots per m², counting dense stands in the field is only feasible if small sub-areas are used.

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

Sampling strategy

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

sites are distributed within separate strata in the sampling area, e.g. in inner, central and outer parts of the estuary and that sampling results are calculated as means for each stratum rather than being calculated as means for the entire estuary. Similarly, it may be an advantage to conduct seagrass sampling within depth strata because many seagrass parameters change markedly with water depth.

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

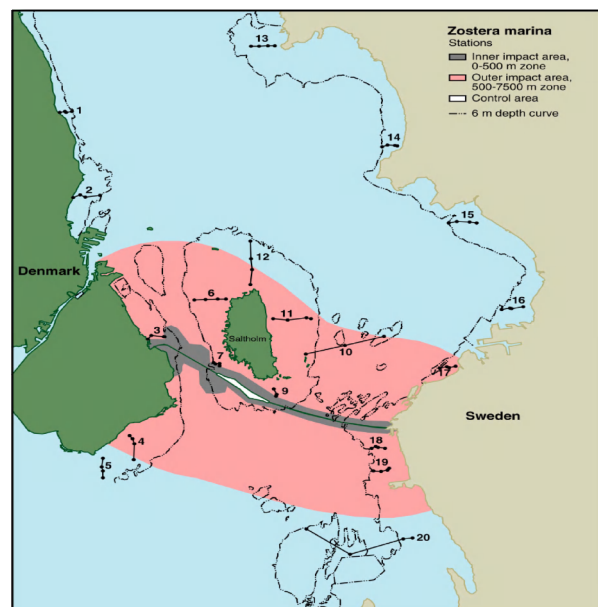


Figure 8.5. Sampling design for eelgrass monitoring under the "Authorities' control and monitoring programme for the fixed link across Øresund". Depth gradients and sampling sites are shown with black lines and dots. Redrawn from Krause-Jensen et al. 2001.

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

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

Conclusions

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

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