# *Water Framework Directive Development of Classification Tools for Ecological Assessment:*

## *Intertidal Seagrass (Marine Angiosperms)*

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# 1. Summary

This report details the development of the Water Framework Directive (WFD) Intertidal Seagrass tool. The report consists of a general background to the WFD, normative definitions and reference conditions.

The provenance of the tool is fully discussed paying particular consideration to the sensitive nature of seagrass beds to human disturbance. Where seagrass beds are present, undesirable disturbance can result in habitat degradation, leading to loss of species and/or bed extent. Seagrass presence is generally regarded as indicative of a healthy environment. However, absence of seagrass is not necessarily indicative of adverse anthropogenic influence. Therefore a decline in the quality of seagrass beds based on taxonomic composition and abundance, recorded as shoot density for subtidal beds or percentage cover for intertidal beds, and bed spatial extent, has been used to correspond to a change in environmental conditions.

Historic seagrass data are rare, with a lack of comparability between locations. Therefore each area of seagrass is assessed on its deviation from reference conditions based on its own specific historic records. Due to natural levels of variation it is also proposed that indices be, preferably, calculated on a rolling mean where sufficient data exist.

The development of the database, reference conditions and threshold setting are discussed, together with the need to take account of geographic variability and the response to pressures.

Pressures with the potential to affect seagrass beds were identified.

Consideration is also given to calculating the final EQR using worked examples, and of calculating the confidence of classification.

# 2. Background to the WFD

The European Water Framework Directive 2000/60/EC governs the protection, improvement and sustainable use of inland surface waters, transitional waters (TW), coastal waters (CW) and groundwaters. The directive, which came into effect on 22nd December 2000, updates previous water legislation and establishes a new integrated water management system based on river basin planning. The key aims of the Water Framework Directive (WFD) are outlined below:

- To prevent further deterioration and protect and enhance the status of aquatic ecosystems and associated wetlands;
- To promote sustainable use of water; and provide sufficient supply of good quality surface water and groundwater.
- To reduce pollution of waters from priority substances

- To prevent deterioration in the status and to progressively reduce pollution of groundwater; and
- To contribute to mitigating the effects of floods and droughts.

The main purpose under WFD guidelines is to develop robust ecological quality objectives (EQOs) and methods for the assessment of impacts of anthropogenic (human-induced) pressures in TWs and CWs. These should look beyond the drivers of change, and link physical and chemical conditions with a measurable biological response in the community.

The Water Framework Directive requires that defined areas of waters (i.e. water bodies) “achieve good ecological and good chemical status” by 2015 unless there are grounds for derogation. Annex V of the Water Framework Directive 2000/60/EC specifies the quality elements and normative definitions on which the classification of ecological and chemical status is based. Normative definitions outline what aspects of the biological quality elements (BQEs) should be assessed, and form the main driver behind the developmentof assessment tools.

The Directive’s requirements include ecological status and chemical status classification schemes for surface water bodies which will differ for rivers, lakes, transitional waters and coastal waters. Heavily modified and artificial water bodies will be assessed in relation to their ecological potential and chemical status classification schemes. The quality elements addressed in Annex V of the Directive for assessing ecological status and ecological potential are:

- biological quality elements;
- general physico-chemical quality elements;
- Environmental Quality Standards (EQSs) for synthetic and non-synthetic pollutants; and
- hydromorphological quality elements.

The specific biological requirements for transitional waters are the composition and abundance of:

- phytoplankton,
- macroalgae
- angiosperms
- benthic invertebrate fauna
- fish

For coastal waters the biological requirements are composition and abundance of:

- phytoplankton,

- aquatic flora (macroalgae and angiosperms)
- benthic invertebrate fauna.

For the ecological status and ecological potential classification schemes, the Directive provides normative definitions of the degree of human disturbance to each relevant quality element that is consistent with each of the ecological status/potential classes. These definitions have been expanded and used in the development of classification tools and appropriate numeric class boundaries for each BQE. The results of applying these classification tools are used to determine the status of each water body or group of water bodies.

The UK was required by 2006 to identify waterbodies at risk of not meeting WFD objectives. This risk assessment exercise was supported by the establishment of national monitoring frameworks and classification schemes. All WFD national monitoring tools are subject to a Europe wide Intercalibration process, in order to ensure all member states assess and classify their waters in a manner consistent with each other and with the Directive.

The WFD is implemented within the UK by the relevant competent authorities, namely the various environment agencies.

This report outlines the development of the UK intertidal seagrass (angiosperm) classification tool, within transitional and coastal waters, to support assessment of the biological quality elements.

# 3. UK Process of WFD Development

## 3.1 UK TAG

The WFD UKTAG is the United Kingdom Technical Advisory Group supporting the implementation of the European Community (EC) Water Framework Directive (Directive 2000/60/EC). It is a partnership of experts from the UK conservation and environment agencies and the Department of Environment and Local Government for the Republic of Ireland. Its main function is to provide coordinated advice on technical aspects of the implementation of the Water Framework Directive (WFD). This includes a coordinated approached to the identification and characterisation of water bodies based on their physical attributes,, and the assessment of the risk of such water bodies failing to achieve the WFD's environmental objectives. It works alongside various experts, and government and stakeholder groups, to develop common approaches to WFD implementation. It also has oversight of the UK's efforts on methods intercalibration within the European Intercalibration framework. WFD requirements have legally binding timetables for completion, enabling a framework for general WFD implementation. This includes tool development and monitoring, which commenced in December 2006, and the setting of environmental objectives under the WFD within the UK and Europe.

Overall the UK TAG group initially provided guidance on;

- Development of **typology of surface waters** (describing water bodies into common types) and the establishment of **type specific reference conditions** for the classification of UK waters;
- The definition and subsequent analysis of **pressures and impacts** for the assignment of water bodies to risk categories;
- The development of **classification tools and methods** that will support monitoring of ecological status.
- Development of an overall **monitoring framework** that supports meeting the different requirements of the Directive and future Programmes of Measures. This includes **operational and surveillance monitoring** designed to assess changes from base-line status of UK water bodies, as well as **compliance monitoring**.
- Production of initial **reports for the European Commission** on characterisation and pressures and impacts analysis.
- Assistance with the **European intercalibration process** that will support defining the thresholds between the five status classes of water bodies under the WFD (high, good, moderate, poor, bad).

The UKTAG initiated the development of classification tools, during the 2003/04 period, with lakes, rivers and marine task teams formed and tasked to: *'coordinate the adaptation and development of suitable surface water classification tools for the biological quality elements'* under the compliance of the European Common Implementation Strategy (CIS). Some of these elements are part historically of UK classification systems, whilst others pose new requirements to support assessment of ecological status. To help implement its work programme, UKTAG has established a number of specialist groups:

- Task teams and steering groups comprising experts from the environment and conservation agencies. These are focused on specific themes or actions (eg lakes, rivers, river basin planning etc). These groups may initiate new research programs.
- Drafting Groups - Small short-lived groups of experts charged with producing specific advice (e.g. drinking water guidance).

The Marine Task Team (MTT) leads the development of classification systems within Transitional and Coastal waters (TraC), providing further guidance to the relevant subgroups including the Marine Plants Task Team (MPTT).

## 3.2 MPTT

The Marine Plants Task Team consists of a number of representatives from various government agencies to provide expertise on the translation, development and implementation of the WFD. It met initially every 4-6 months to discuss progress within the phytoplankton, macroalgae and marine angiosperm classification tools as directed by the MTT and UK TAG. The Marine Plant Task Team's role has been to translate the WFD legislative report into practical ecological and scientific classification methods for marine plants, and in so doing it has developed a number of classification tools to comply with the requirements of the WFD.

Within this group the UK and Republic of Ireland (RoI) representatives have had to ensure harmonisation of ecological classification systems to ensure a coherent approach by both member states. The tools have been, or are being, developed both 'in house' and by consultants, with funding from a number of sources including the environment agencies, SNIFFER, and the Irish North South (SHARE) project, which is INTERREG funded and managed jointly between authorities in Northern Ireland and the Republic of Ireland. The approach includes the:

- review and adaptation of existing methods for potential to support classification schemes under the WFD;
- development of new tools for elements not previously monitored in the UK and RoI;
- assessing which parameters have the best correlation for assessing pressures and impacts;
- development of reference conditions from which to base boundary criteria;
- trialling such tools in the assessment of ecological quality status; and
- the review, comparison and agreement of methods with other EU Member States to comply with intercalibration requirements

This document describes the process involved in the development stages of the angiosperm (intertidal seagrass) tool, considering both its theoretical and practical elements, subsequent implementation and inclusion in the European Intercalibration process. The tools are grounded in scientific knowledge and published and unpublished research, but wherever there is uncertainty or a scarcity of quantitative scientific evidence the precautionary principle has been invoked.

# 4. Normative definitions & Reference conditions

The criteria by which ecological status should be evaluated are detailed in the normative definitions in Annex V(1.2) of the Water Framework Directive. Normative definitions provide definitions of ecological quality and the values for the quality elements of ecological status for coastal and transitional waters. They describe the various aspects of marine angiosperms that must be used in the ecological status assessment of a water body. In WFD "marine angiosperms" comprises both seagrass and saltmarsh species. Indices ('tools' or 'metrics') have been developed to address these aspects of the normative definitions for each of the five status classes. The WFD normative definitions specify which aspects of each biological quality element must be assessed, and the plants tools have been developed accordingly. The WFD treats marine angiosperms and macroalgae as separate BQEs in TWs but groups them in CWs. These are therefore two sets of normative definitions, but these are very similar, and are shown in Tables 1a) and b).

Table 1 a) Normative definitions for marine angiosperms in Coastal Waters

| Class | Normative Definition |
|---|---|
| HIGH | All disturbance-sensitive angiosperm taxa associated with undisturbed conditions are present. The levels of angiosperm abundance are consistent with undisturbed conditions. |
| GOOD | Most disturbance-sensitive angiosperm taxa associated with undisturbed conditions are present. The levels of angiosperm abundance show slight signs of disturbance. |
| MODERATE | The composition of the angiosperm taxa differs moderately from the type-specific communities and is significantly more distorted than at good quality. There are moderate distortions in the abundance of angiosperm taxa. |
| POOR | Major alterations to the values of the biological quality elements for the surface water body type.<br><br>Relevant biological communities deviate substantially from those normally associated with the surface water body type under undisturbed conditions. |
| BAD | Severe alterations to the values of the biological quality elements for the surface water body type.<br><br>Large portions of the relevant biological communities normally associated with the surface water body type under undisturbed conditions are absent. |

Table 1b) Normative definitions for marine angiosperms in Transitional Waters

| Class | Normative Definition |
|---|---|

| | |
|---|---|
| HIGH | The taxonomic composition corresponds totally or nearly totally to undisturbed conditions. There are no detectable changes in angiosperm abundance due to anthropogenic activities. |
| GOOD | There are slight changes in the composition of angiosperm taxa compared to the type-specific communities. Angiosperm abundance shows slight signs of disturbance. |
| MODERATE | The composition of the angiosperm taxa differs moderately from the type-specific communities and is significantly more distorted than at good quality. There are moderate distortions in the abundance of angiosperm taxa. |
| POOR | Major alterations to the values of the biological quality elements for the surface water body type.<br><br>Relevant biological communities deviate substantially from those normally associated with the surface water body type under undisturbed conditions. |
| BAD | Severe alterations to the values of the biological quality elements for the surface water body type.<br><br>Large portions of the relevant biological communities normally associated with the surface water body type under undisturbed conditions are absent. |

## 4.1 Expanded Normative Definitions

These Normative definitions have been expanded by the MPTT (Dublin 2004) (see Table 2) to provide examples of how they apply directly to the abundance of seagrass within Transitional and Coastal waters including their structural and functional relevance. These descriptions form the basis for the development of the seagrass tool currently being used for WFD ecological assessment and apply to sedimentary shores. As there was no evidence for seagrass responding differently to pressures in coastal or transitional waters, MPTT decided to treat them together and so the expanded normative definitions apply to both CWs and TWs.

Table 2: Description of the characteristics of seagrass at the High, Good and Moderate WFD status classes in accordance with the normative definitions (WFD Annex V) and expanded normative definitions (detailed national interpretation).

| | | |
|---|---|---|
| Interpretation of structural & functional relevance | There are only 5 UK seagrass species[1]; *Zostera marina*, *Z. angustifolia* (known as littoral *Z. marina* in continental Europe) and *Z. noltei plus 2 species of Ruppia*. *Z. noltei* (littoral) and *Z. marina* (sublittoral) occur commonly as mono-specific stands in UK waters.<br><br>Where present, beds should be healthy, with no loss of bed extent or density (shoot density/percentage cover). This defines the Good/Moderate boundary. Note: natural variability may be up to 30% (Krause-Jensen *et al*., 2003).<br><br>Where data sets allow, a 5-year rolling mean for shoot density[2] should be used to reduce noise and identify longer term trends. A 30% reduction in density when using a 5-year rolling mean will mask underlying trends. Therefore 15% is considered as tolerable evidence of natural variation and decreases in extent of > 15% should be viewed suspiciously. | |
| High | *The angiosperm taxonomic composition corresponds totally with undisturbed conditions. There are no detectable changes in angiosperm abundance due to anthropogenic activities* | No loss of seagrass species.<br><br>Abundance as bed extent: no loss in area of seagrass bed – at maximum potential and stable (within natural variability).<br><br>Abundance as density: no loss of density/% cover – increasing or at highest value previously recorded (within natural variability). |
| Good | *There are slight changes in the composition of angiosperm taxa compared with the type-specific communities. Angiosperm abundance shows slight signs of disturbance* | No loss of seagrass species.<br><br>Abundance as bed extent: < 30% deviation from highest recorded; i.e. within natural variability, but bed at less than maximum potential extent for local physical regime or compared with bed's historic extent.<br><br>Abundance as density/% cover: no loss – < 30% (or <15% if using 5-year mean) deviation from highest previously recorded; i.e. within natural variability.<br><br>Changes that occur at this stage are gradual and reversible in the short term. |
| Moderate | *The composition of angiosperm taxa differs moderately from type-specific conditions and is significantly more distorted than at good quality. There are moderate distortions in the abundance of angiosperm taxa* | Loss of 1 seagrass species, but at least 1 species still remaining in the water body.<br><br>Abundance as bed extent: >30% deviation from highest recorded; i.e. greater than natural variability. Disturbance evident as moderate loss of area covered compared with previous highest recorded extent.<br><br>Abundance as density/% cover: >30% (or >15% if using 5-year mean) deviation from highest value previously recorded; i.e. beyond natural variability.<br><br>The changes that occur at this stage are still gradual and reversible in the medium-term; e.g. within a |

| | | |
|---|---|---|
| | | reporting cycle (5 year rolling mean). |

*Note 1: For WFD purposes* Z.angustifolia *is considered as a separate taxon, although considered by many to be a variant of* Z.marina. Ruppia *is only considered at genus level, due to the practical difficulties of species level identification. The maximum number of taxa per WB therefore is 4.*

*Note 2: Shoot density is applicable in subtidal beds; density is measured as "percentage cover" in intertidal beds.*

## 4.2 Reference Conditions

Reference conditions represent, as far as possible, undisturbed conditions for the BQE, and class boundaries are set in relation to these. The Water Framework Directive states type-specific biological reference conditions may be spatially based, based on modelling, or derived using a combination of these methods. For spatially based type-specific biological reference conditions, Member States are developing a reference network for each surface water body type. Predictive models or hindcasting methods should use historical, palaeological and other available data. Where it is not possible to use these methods, expert judgement may be used to establish such conditions.

The approach of establishing spatially based type-specific reference conditions from a reference network for each surface water body type and comparing seagrass beds against these is problematic. A limited number of potential reference sites in UK TraC (Transitional and Coastal) waters do exist, however it is not possible to establish type-specific reference conditions for all types. Seagrass distribution, abundance and ecological condition are highly variable and sensitive, and causes of deviation from proposed reference conditions are multiple, not always detectable and difficult to monitor in a quantifiable manner. Historical data of appropriate quality are very limited.

The approach of using predictive models has been considered. Ideally it would be possible to identify locations with suitable environmental parameters for seagrass and therefore to predict presence. This would enable targeting of monitoring directly to extant seagrass beds and potential sites, and absence of seagrass communities from such identified locations would be indicative of poor classification status. However, such accurate prediction has proved to be elusive, despite best modelling efforts in recent years (e.g. Fonseca *et al*., 2002: Krause-Jensen *et al*., 2003). Krause-Jensen *et al*. (2003) were able to evaluate the importance of light, wave exposure, slope, salinity and depth in regulating sublittoral eelgrass cover in Danish coastal waters. The role of each regulating factor in relation to eelgrass cover at different depth intervals was determined with statistical significance. Even so, the predictive power of the models was limited, and for management purposes they cannot adequately predict eelgrass cover. In some localities eelgrass was absent or exhibited very low cover in shallow waters despite sufficient light. Such discrepancies suggest other factors are acting on seagrass distribution; for example, grazing by wildfowl, fish or invertebrates, sediment conditions, epiphytes and freeliving macroalgae, extreme low tides and extreme dynamic events (Krause-Jensen *et al*., 2003). The past history of extreme events may play a significant role for the current presence and cover of seagrasses in most systems and any field survey will include sites representing many different developmental stages of the vegetation ranging from bare sediment, initial colonisation, to fully developed meadows (Krause-Jensen *et al*., 2003).

Establishment of reference conditions based on historic data and expert judgement is possible in some localities. In Wales, the Countryside Council for Wales (CCW) (Kay, 1998) conducted a comprehensive review of the knowledge of seagrass beds around the Welsh coast. The review pools verbal, written and numeric information from a wide variety of sources and provides summaries of knowledge of individual beds. Such historic data may be suitable for the establishment of reference conditions for individual seagrass beds, provided they are considered accurate and quantifiable. The National Biodiversity Network (NBN) also provides a source of seagrass data, which are collated from many of the UK's wildlife conservation organisations, the government and countryside agencies, environment agencies, local records centres and also many voluntary groups. This can provide information on the geographic distribution of seagrass beds and species presence throughout the UK. However, there has not been a national seagrass monitoring programme in the UK and, for many sites, monitoring on a local scale has employed one of a variety of methods, resulting in data that cannot be compared across sites. Where no historic data exist baseline surveys must be conducted and expert judgement relied upon to identify reference conditions for the seagrass bed. The objective is for a seagrass bed's taxonomic composition and abundance to be stable at the maximum potential for the site.

There is a lack of evidence to show that seagrasses respond differently to pressures in CWs and TWs, and a lack of data to develop type-specific reference conditions. CWs and TWs are therefore treated together here in the development of reference conditions and class boundaries. Deriving these for saline lagoons would be yet more challenging, and so this WB type is not covered by the tool.

## 4.3 Ecological Quality Status

Once reference conditions are established, the departure from these can be measured. The degree of deviation sets boundaries for each of the WFD ecological status classes. These boundaries need to be described, and criteria established which reflect the normative definitions.

Annex V 1.4.1 of the Directive states *"the results of the (classification) system shall be expressed as ecological quality ratios for the purposes of classification of ecological status. These ratios shall represent the relationship between the values of the biological parameters observed for a given body of surface water and the values for these parameters in the reference conditions applicable to that body. The ratio shall be expressed as a numerical value between zero and one, with high ecological* *status represented by values close to one and bad ecological status by values close to zero."*

Figure 1 illustrates this concept.

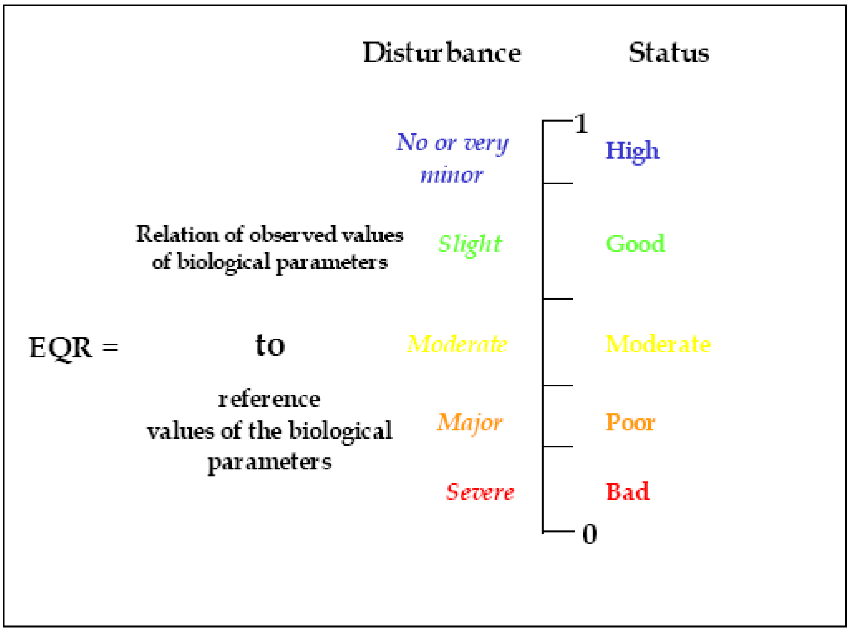



Figure 1: Suggested Ecological Quality Ratio; (From COAST Guidance, Vincent *et al*., 2002).

EQRs are derived by comparing monitoring results with the reference conditions . The values of the EQR then set for each ecological status class must ensure that the water body meets the normative definition for that status class given in Annex V. As such the reference conditions form the anchor for the whole ecological assessment. Ecological status classes will be defined by their deviation from reference.

### 4.3.1 Classification

The outcome of any one assessment tool will be combined with the assessments of other WFD quality elements to inform the overall classification of a water body.

# 5. Seagrass Monitoring Tool

## 5.1 Introduction

Seagrasses are the only truly marine angiosperms. Eelgrass (*Zostera* species) is the only true seagrass occurring in the UK. Although *Ruppia* species are not strictly considered as part of the traditional seagrass arrangement (Kuo & den Hartog, 2001) workers often group *Ruppia* species with *Zostera* species, considering them all as seagrasses, as they occupy a similar niche. For the purposes of WFD assessment both genera are monitored. .Seagrasses can be used as monitoring objects because they are sensitive to anthropogenically induced disturbance (Short & Wyllie-Echeverria, 1996). All UK seagrass species are included in the UK Biodiversity Action Plan, 1994, and are considered nationally scarce. Annex V of the Water Framework Directive ((WFD) Directive 2000/60/EC) states that angiosperms are a *biological quality element* to be used in defining ecological status of a transitional or coastal water body. Reference conditions for transitional (TW) and coastal waters (CW) are defined:

**Coastal Waters**

*All disturbance-sensitive angiosperm taxa associated with undisturbed conditions are present*. *The levels of cover and angiosperm abundance are consistent with undisturbed conditions.*

### Transitional Waters

*The angiosperm taxonomic composition corresponds totally or nearly totally with undisturbed conditions. There are no detectable changes in angiosperm abundance due to anthropogenic activities.*

These descriptors set out the attributes to be used in angiosperm monitoring and the standards required within a water body for it to be considered pristine, i.e. at reference condition. They can be summarised as taxonomic composition (including presence of disturbance-sensitive taxa) and abundance (determined by seagrass bed density and spatial extent) in both CWs and TWs.

The normative definition requires translation into a workable metric system that can be utilised in the assessment of seagrass beds for establishing the ecological quality status of water bodies. The first step in this process was the collation of various sources of literature, accurate data and expert knowledge from which to establish class boundaries and help set initial reference conditions. This initial study provided the information on the main characteristics used in seagrass monitoring which were later combined and translated onto a numerical scale or index of ecological quality status from 0.00 to 1.00. Detailed monitoring procedures were produced to ensure consistent collection and interpretation of data. Each competent monitoring authority will hold its own standard operating procedures based on these. The metric system was applied to existing historical data to assess the accuracy of the tool and apply preliminary water body classification status to those areas in which seagrass are present. The following sections outline the stages in the development of the angiosperm tool.

## 5.2 Database

As stated previously, historic seagrass data are rare, and different monitoring methods mean that data cannot be compared across sites. There is, however, a plethora of reports and papers on individual seagrass beds, or groups of beds, published in peer-reviewed journals and grey literature. There are too many such sources of data to provide a comprehensive list here, but there follow a few examples of the more useful reports that help provide historic data.

The Countryside Council for Wales (CCW) (Kay, 1998), documented the collation of several sources of information and data from a number of localities. English Nature published a similar report (Hocking & Tompsett, 2002) detailing the location of eelgrass (*Zostera* spp.) beds in Cornwall and the Isles of Scilly, including all historic data these authors were able to find. Such historic data have been used in the establishment of reference conditions and can act as baseline data for many seagrass beds. Davison (1997) and Davison & Hughes (1998) provided a comprehensive overview of *Zostera* biotopes, including site descriptions for extant beds at that time. An overall distribution of *Zostera* species in mainland UK is provided in Figure 2 based on those given in Stewart *et al*. (1994) and these are reproduced below.

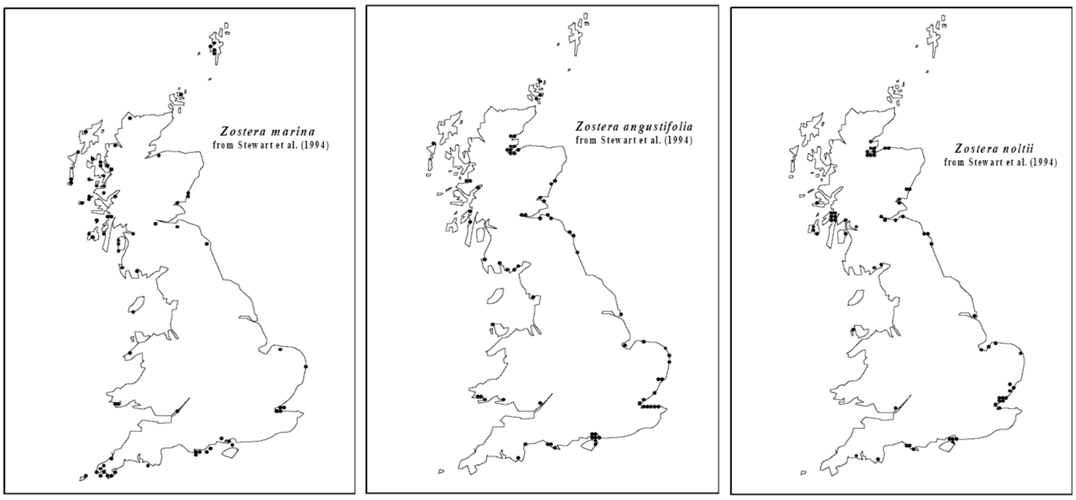



**Figure 2:** Distribution of Zostera species in mainland UK (Davison & Hughes, 1998 and Stewart *et al*., 1994)

The Marine Nature Conservation Review of Great Britain (the MNCR) commenced in 1987 with the main objectives of identifying sites and species of nature conservation importance in Great Britain in particular, descriptions of their characteristics, distribution and extent. Subsequent to the Environmental Protection Act 1990, the MNCR was undertaken by the JNCC on behalf of the Countryside Council for Wales (CCW), English Nature (EN) and Scottish Natural Heritage (SNH). Seagrass species appear in the database and records are useful for presence/absence of seagrass around the UK as Figure 3 shows. Some records date back to the 1970s and as such provide good historic data. *Note*: The map shows both subtidal and intertidal seagrass beds.

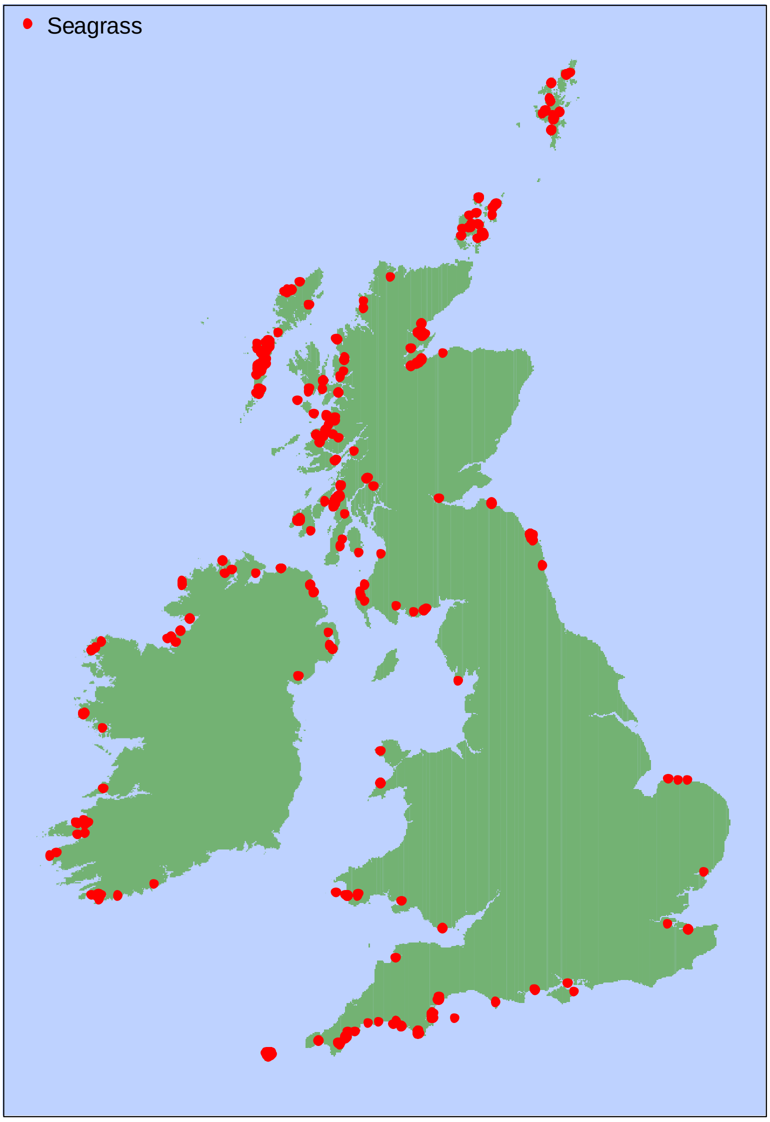



**Figure 3:** Seagrass bed locations from the JNCC's MNCR database

These historic data form a vital component in the process of developing an assessment tool. Not only do they provide detailed information on the location of beds, but also some time series data and natural levels of seagrass that can be used as references from which to establish class boundaries. The literature further highlights those characteristics of seagrass that are most appropriate for continued studies, and these were investigated further during the development of the tool.

## 5.3 Determination of tool components

As seagrasses are disturbance sensitive (Short & Wyllie-Echeverria, 1996) their presence, health and abundance are likely to indicate a water body's classification as being at good or higher status; provided there is no evidence of degradation or loss of species from localities where previously found. Importantly, despite much recent research effort, the ideal environmental parameters for supporting seagrass are not entirely understood, so that absence of seagrass from areas apparently suited to its growth is not always explicable (Krause-Jensen *et al*., 2003). An absence of seagrass from an apparently suitable environment, therefore, does not necessarily suggest a catastrophic loss of species has occurred, unless a historic bed was previously recorded and is no longer present.

Based around this, the main approach in the establishment of a robust tool using seagrass communities to describe ecological status, was to determine which parameters were most appropriate. The primary difficulty lay in the interpretation and implementation of the definitions of ecological status and three main factors needed to be considered;

1. The species of seagrass in the UK (3 *Zostera* & *Ruppia*) tend to occur in mono-specific or 2-species stands;
2. All UK seagrass species are disturbance sensitive; and
3. Total biomass as a measure of abundance would require undesirable destructive sampling.

Therefore three indices have been developed that apply to littoral seagrass beds in both TWs and CWs to meet the monitoring requirements:

- Taxonomic composition (presence of disturbance sensitive taxa)
- Abundance, expressed intertidally as percentage cover of substratum by seagrass leaves and subtidally as seagrass shoot density (no. shoots per $m^2$)
- Abundance, measured by seagrass bed spatial extent

*Note*: Shoot density is more appropriate for subtidal seagrass beds, where shoots are upright and easier to count. Percentage cover has been adopted for intertidal seagrass beds.

Each of these indices was investigated for their response to anthropogenic and natural change and their applicability in assessing ecological quality status.

### 5.3.1 Taxonomic Composition

Seagrasses in the northern temperate oceans tend to form broad, mono-specific stands (Davison & Hughes, 1998), often patchy in nature, typified by meadows of *Zostera* spp. in the Atlantic coastal regions (Short *et al*., 2001). The three species of true seagrass found in the UK are *Z. noltei*, *Z. marina* and *Z. angustifolia* (Davison & Hughes, 1998). *Zostera angustifolia* may be regarded as a littoral ecotypic or phenotypic form of *Z. marina*, however, in most UK literature the two species are made distinct and this convention is followed for WFD. UK seagrass beds tend to be more modest in extent than in other European countries. In UK waters *Z. marina* is predominantly a sublittoral species found in shallow, fully marine conditions on relatively coarse sediment (Davison & Hughes, 1998). *Zostera angustifolia* and *Z. noltei* are found in the littoral (intertidal) zone. *Zangustifolia* generally occurs between the mid- and low-tide mark, preferring poorly-draining muddy sediments, particularly pools and creeks that are unlikely to dry out entirely during exposure. *Z.noltei* occurs higher on the shore to the high-tide mark, on mud and sand, and being more tolerant of desiccation will inhabit exposed areas that may entirely dry out over a tidal cycle (Davison & Hughes, 1998).

Although *Ruppia* species (widgeon grass) are not strictly considered as part of the traditional seagrass arrangement (Kuo & den Hartog, 2001) workers often group *Ruppia* species with *Zostera* species, considering them all seagrasses. For the purposes of WFD assessment both genera are monitored. *Ruppia* spp. are poikilosaline aquatic plants, which may occur together with *Zostera* seagrass, as their environmental preference is very similar; i.e. temporarily to permanently flooded mesohaline-hyperhaline estuarine wetlands (Kantrud, 1991), brackish waters of lagoonal habitats, lochs and estuaries. As with most species of *Zostera*, *Ruppia* populations generally inhabit warm, relatively unpolluted, and well lit waters <2.0 m deep where current, fetches and wave action are minimal (Kuo & den Hartog, 2001). *Ruppia* spp. can tolerate significant water level fluctuations, including periodic exposure in tidal areas (Kantrud, 1991). Around the UK *Ruppia* beds have a scattered distribution, dependant on available habitat. There are concentrations in the Cromarty Firth, Scotland, The Fleet, England, and lagoonal habitats of western Scotland, Orkney and Shetland, but they appear to be absent from Ireland.

*Ruppia* species are difficult to identify to species level, and so for WFD purposes it was decided to record *Ruppia* to genus level only. This gives a possible maximum of 4 taxa in any UK water body.

### 5.3.2 Abundance Determined by Bed Extent and% cover (Shoot Density)

Normative definitions in the WFD state that there should be no detectable changes in angiosperm abundance due to anthropogenic activities and they should be consistent with undisturbed conditions. Abundance is being addressed as total bed extent and shoot density subtidally or % cover intertidally. Total biomass as a measure of abundance requires destructive sampling; this was considered not to be in the spirit of the WFD or Habitats Directive (1992) and was disregarded for further investigation and development as a metric. In contrast, mapping abundance (as the spatial extent of seagrass beds) and recording density are less destructive and were proposed as more appropriate indicators for continued investigation.

Loss of seagrass abundance occurs in many coastal environments (Short & Wyllie-Echeverria, 1996), often from natural causes such as wasting disease (more applicable to subtidal beds) or high energy storms. However, undesirable disturbance has also been caused by anthropogenic activity leading to hydro-morphological changes; such activities include fishing, vessel mooring, coastal defence engineering, industrial development and waste dumping (Kemp *et al*., 1983 and Short & Burdick, 1996). The consequence of such activities may manifest as a reduction in seagrass abundance and results can be catastrophic.

Seagrasses are sensitive to nutrient enrichment and in some temperate estuaries (in the northern hemisphere) areas of eelgrass (*Zostera* spp.) habitat have been shown to decrease logarithmically, and % loss of habitat to increase logarithmically, as nitrogen loading rates increased (Hauxwell *et al*., 2003). This has sometimes been recorded even at relatively low loading rates. However, seagrass can recover if conditions improve. Seagrass beds of *Zostera* and *Ruppia* species, in Corsica, exhibited a 12% decrease in abundance between 1990 and 1994 as a result of salinity and temperature shocks and elevated levels of silting: there was a 16% increase between 1994 and 1997 following a return to background levels of discharge (Agostini *et al*., 2002). These observations imply that regressions are not irreversible and show that seagrass meadows can recover if environmental conditions revert to a 'pre-disturbance' state.

Nutrient enrichment may also lead to excessive growth of opportunistic epiphytic algal species such as *Enteromorpha (=Ulva* tubular forms*)*, *Ulva* (laminar forms), *Chaetomorpha* and *Ectocarpus/Pylaiella* on seagrass beds. The effect of macroalgal mats is dependent on their density and persistence (with considerable geographic and temporal variation), potentially compromising the health and viability of seagrass if overlying and smothering. Descriptive field studies have found that such algae appear to inhibit or eliminate eelgrass (Kemp *et al*., 1983; Dennison *et al*., 1993) and excessive growth can cause serious deterioration or even the eradication of seagrass. For example, a seagrass bed of *Z. noltei* and narrow-leaved littoral *Z. marina*, i.e. *Z. angustifolia*, approximately 10 ha in size on the intertidal flats of Langstone Harbour, UK was monitored annually from 1986. In September 1991 this seagrass bed appeared to be largely destroyed by a thick blanket of the green alga, *Enteromorpha radiata*. Most still living *Zostera* plants were in a severely deteriorated condition. By August 1992 *Zostera* was completely absent from the growing area (den Hartog, 1994). Abundance of opportunist macroalgae is considered a significant problem and has been tackled as a separate entity negating the need for an additional macroalgae index as part of the seagrass tool. *Note: The genus* Enteromorpha *is now recognised as part of the genus* Ulva *(Hayden* et al *2003), but is retained as a separate entity for WFD macroalgal blooming work owing to its different morphology.*

Abundance and taxonomic composition of seagrass beds both show evidence of change as a result of direct and indirect influences. The exact response and level of change needed to be examined more carefully in order to correlate this with changes in the environment, be they natural or anthropogenic, and furthermore developed into a metric.

## 5.4 Development of Metrics

It is not considered possible for a single metric to be used in isolation to understand seagrass ecology or to derive a classification for a water body. In water bodies where seagrass are, or historically were, present all of the proposed metrics should be used in assessing the ecological status for this biological quality element. Results for these are then combined within this multi-metric tool.

### *5.4.1 Taxonomic Composition*

As previously noted, UK seagrasses comprise three species of *Zostera*, with possible co-occurrence of *Ruppia* spp. The taxonomy of *Ruppia* spp. is difficult and under revision as *Ruppia maritima* may be confused with *Ruppia cirrhosa* (syn. *spiralis*) (Preston, 1995). Consequently identification to genus level only is recommended, resulting in a relatively low level of identification expertise required of field workers to implement this index.

As the actual number of seagrass species is low, total richness is inappropriate as an accurate measure. Therefore the final index for this particular element of seagrass was based around the number of species present, as detailed from historical records or baseline surveys, remaining consistent. Any loss of species is considered to be as a consequence of changing environmental conditions and would result in a deviation from reference conditions.

Table 3 presents the proposed scoring scheme for this index based on the five disturbance descriptors representing each ecological status class. Scores in the range 0-1 align with the Directive's requirements for an EQR. The taxonomic composition metric has scores associated with loss of species from reference conditions representing the midpoints of the class ranges. Although five classes have been defined, there are limitations on the applicability of classes for some water bodies, due to the starting number of seagrass taxa. For example, some water bodies may have *Ruppia* spp. and all three *Zostera* spp. present, though occurring in different beds. In such cases the water body would have a maximum number of taxa of 4. Another water body may naturally have only mono-specific beds. If there is no loss of seagrass taxa over time, each WB will score 0.9 for this metric. While intuitively this should be 1.0, we cannot have 100% statistical certainty that all taxa have been found. It is considered that there is a greater probability of false negative results (missing taxa which are present) than of false positives (identifying more taxa than are actually present). Assuming a probability of 10% for a false negative means that the maximum EQR can only be 0.9. Further explanation of this is given in Section 7.3 Confidence of classification.

The score of 0.7 applies only in water bodies with reference conditions of three or four seagrass taxa present, where 1/3 or 1/4 of species, respectively, is now absent. The metric score of 0.5 is only applicable where a water body has reference conditions of two or four species naturally co-existing, but only 1/2 of these are now extant. The score of 0.3 is applicable to water bodies with reference conditions of three or four species, and in either case only one species remains (a species loss of 2/3 or 3/4 respectively).

Where no seagrass taxa remain the water body would be scored as 0.1 for this metric, regardless of the starting number of taxa. Once again, we cannot be absolutely certain that we have not overlooked some seagrass, and so statistically the minimum EQR must be 0.1 rather than zero (see Section 7.3).

Total loss of a mono-specific bed could therefore downgrade a water body from a score of 0.9 to 0.1 in one step. In such cases the metric is insensitive to intermediate classes (0.3–0.7).

**Table 3:** Metric description and EQRs for taxonomic composition

| *Quality Status* | *Disturbance* | *Change in taxonomic composition from reference conditions* | *Metric EQR* |
|---|---|---|---|
| High | No detectable change | All reference taxa present | 0.9 |
| Good | Slight signs of disturbance | Loss of 25% to 33% of reference number of taxa | 0.7 |
| Moderate | Moderate distortions | Loss of 50% of reference number of taxa | 0.5 |
| Poor | Major distortions | Loss of 66% to 75% of reference number of taxa | 0.3 |
| Bad | Severe distortions | Loss of all taxa | 0.1 |

### *5.4.2 Abundance - Bed Extent*

The proposed scoring scheme for changes in bed extent is based on the five disturbance descriptors with consideration given to the normative definitions, reference conditions and boundary condition descriptors. The objective is for a seagrass bed's spatial extent to reach, and be in equilibrium at, its maximum potential physical extent given the local climate, substratum and hydrodynamic regime, tolerating natural variability. The expectation is that the bed will decrease in size in response to pressures.

Annual natural variability may be high so, wherever possible, assessment should be based on trends in bed extent. Ideally trend lines would be neutral if the bed is in equilibrium at its predicted maximum potential, or positive if the bed's abundance is lower than its predicted potential, but is in a recovery phase. A negative trend is a signal of deterioration and more detailed investigation may be necessary to halt further decline.

As noted above, a combination of present knowledge and expert judgement has been used to set the boundary criteria for the UK's abundance metrics (Table 4), whilst employing the precautionary approach. A loss of >30% from reference conditions should be viewed as questionable. As with density, if a >30% loss in seagrass bed extent is recorded, investigative monitoring should determine if the loss is attributable to an extreme natural or anthropogenic event, and a final classification reached accordingly. The class boundaries have been set as described for percentage cover.

**Table 4:** Metric system developed for the classification of changes in spatial extent of angiosperm beds

| Quality Status | Disturbance | Exemplar metric scores for % loss of area from reference conditions | Metric EQR |
|---|---|---|---|
| High | No detectable change | >0% – 10% area loss | 1.0 – 0.8 |
| Good | Slight signs of disturbance | >10% – 30% area loss | 0.8 – 0.6 |
| Moderate | Moderate distortions | >30% – 50% area loss | 0.6 – 0.4 |
| Poor | Major distortions | >50% – 70% area loss | 0.4 – 0.2 |
| Bad | Severe distortions | >70% – 100% area loss | 0.2 – 0.0 |

As with the assessment of shoot density (% cover) the metric for bed extent also works on a sliding scale to enable an accurate EQR value to be calculated for this particular parameter.

**Calculation of the ecological quality ratio**

The ecological quality ratio (EQR) for the parameter, % loss of area, should be calculated using the following equation whereby "value" signifies the observed value:

$$\textbf{EQR = upper EQR parameter range -} \left\{ \left[ \frac{\textbf{value – lower class range}}{\textbf{class width}} \right] \textbf{x EQR band width} \right\}$$

***Example*** using a value of 39% for % loss of shoot density, consult Table 4: 39 lies between 31 and 50 and with an EQR between 0.4-0.6, therefore:

Score = 0.6 - ((39 – 31)/19) x 0.2

Score = 0.6 – 0.084 = 0.516

Where no historic data exist, the first year of monitoring may provide a reference value for the bed extent. Where several years of data exist, these may be averaged. However, in practice, the first year of monitoring may not identify all beds locally, and this should be considered when establishing the reference value for bed extent. The second year may present greater extent, so expert knowledge should be applied as to which year represents the baseline.

### 5.4.3 Abundance - Shoot Density/Percentage cover

Krause-Jensen et al. (2003) analysed the importance of light, wave exposure, substratum slope and salinity on the biomass, cover and shoot density of a large data set crossing geographic regions, at different depth intervals. The authors found variability to be high in shallow water where populations were disturbed by physical parameters. The proposal, therefore, is that density data are not compared across geographic regions, as naturally occurring local physical parameters may cause significant natural change. Rather, an individual bed's current spatial extent and density should be compared against historic data representing its healthiest previously recorded condition (reference condition).

The proposed scoring system for shoot density is based on loss in density compared with reference conditions, measured as % leaf cover. The calculated percentage change should be rounded to the nearest integer and assigned a metric score. The normative definitions, reference conditions and boundary condition descriptors previously discussed have been taken into consideration. The objective is for a seagrass bed's abundance to increase and be in equilibrium (tolerant of natural variability) at the maximum potential for the site, with the expectation that the bed will decrease in density if there is ecological deterioration in a water body. Where several years of annual data exist for a seagrass bed, abundance will be the previously recorded maximum density of the bed, or the mean of several years' data where such exist. If data exist to enable trend lines to be plotted these should be neutral if the bed is in equilibrium at its predicted maximum potential, or positive if the bed's abundance is lower than its predicted potential but is in a recovery phase. A negative trend is a signal of deterioration and more detailed investigation may be necessary to halt further decline. *Note: When using a specific year as baseline density data, this* *must be the year of the greatest bed extent, i.e. the density data linked to the greatest bed extent data, even if this is not the year of greatest density.*

Various literature and historical data were used to establish boundary values. A precautionary approach, in combination with present knowledge and expert judgement, has been used to set the boundary criteria for the abundance metrics. These are defined as percentage losses from reference conditions, rather than absolute values. The boundary between good and moderate is perhaps of greatest significance because a water body falling below good status may be subject to investigative monitoring and programmes of measures involving investment to improve the environment. A study by Krause-Jensen et al. (2003) on the effects of light, exposure and salinity produced models which explained up to 40% seagrass presence and cover on large spatial scales, and these authors suggest it is likely the remaining >60% variability results from a combination of natural causes and anthropogenic influences leading to undesirable disturbance. Based on this the good/moderate boundary value has been set at 30% less than reference conditions. This limit allows for natural variability but is sensitive enough to highlight variability caused by anthropogenic activity. Seagrass knowledge and research in other European Union member states supports this boundary value (de Jong, 2004; de Jong, pers. comm., 2006). Density losses in excess of this percentage may be indicative of undesirable disturbance. If a >30% loss in seagrass abundance is recorded, investigative monitoring should determine if the loss is attributable to an extreme natural event (e.g. weather or low annual light levels), or an extreme anthropogenic event; a final classification can be assigned accordingly. Where sufficient data allow, trends in abundance or a rolling mean may be calculated, which provide evidence of general loss or recovery in the bed's condition.

Based on expert judgement (Alex Portig, Dick de Jong, Paul Brazier and Jo Foden) and historical data from the British Isles, it was thought that the High/Good boundary should be set at <10% loss of density. However if a bed is expanding or becoming more dense than its reference condition it will record 0% loss and will naturally be in 'High' status. It was decided that a loss of 70% of bed density would possibly result in a change in hydrodynamics or altered sediment regime and with the additional causative factor contributing to the initial decrease the remaining bed is likely to struggle to survive. The poor/bad boundary was therefore set at 70% with the remaining class boundary between moderate and poor being chosen mathematically as the mid-point between 30% and 70% so has been set at 50% (Table 5).

Table 5: Metric system developed for the classification of annual changes in angiosperm shoot density/% cover.

| | ***Annual change*** | | |
|---|---|---|---|
| Quality Status | Disturbance | Exemplar metric scores for % loss of density from reference conditions | Metric EQR |

| | | | |
|---|---|---|---|
| High | No detectable change | 0% – 10% loss | 1.0 – 0.8 |
| Good | Slight signs of disturbance | >10% – 30 % loss | 0.8 – 0.6 |
| Moderate | Moderate distortions | >30% – 50% loss | 0.6 – 0.4 |
| Poor | Major distortions | >50% – 70% loss | 0.4 – 0.2 |
| Bad | Severe distortions | >70% – 100% loss | 0.2 – 0.0 |

As noted above, density will vary naturally between beds. It is more appropriate, therefore to monitor temporal fluctuations within a water body, than to compare across sites. For example, where seagrass exists in marginal areas, abundance may be low naturally. This does not necessarily signify low ecological status and is why abundance should be monitored for individual beds and compared, where possible, against long-term data.

Duarte and Kirkman (2001) found the time frame to determine real changes brought about by most human disturbance may take 5–10 years, unless disturbance is catastrophic such as habitat removal for coastal redevelopment. Consequently it is proposed that classification status for density takes account of trends, where enough data exist. If there is a very high degree of annual variability, calculation of rolling means will considerably reduce noise and underlying trends become more apparent; but the seagrass bed would need ideally to have been monitored routinely for in excess of ten years for a rolling mean to become a useful statistic. Trends or rolling means of five to six years duration can be designed to coincide with the WFD's reporting cycle. The rolling mean for an individual bed and the % loss or gain, as compared with reference conditions (the maximum recorded density), can be used to establish a scoring system. The rolling-mean value for each year is an average of that year and the previous four years' mean densities (or as many years' data as exist within the time period). However, there are few UK seagrass beds that have long-term monitoring data available. Where data sets of this length do not exist, trends in annual mean density must be ascertained from the available years of data. In practice, the first year of monitoring may not identify all beds locally, and this should be considered when calculating mean data.

Where there is a dataset long enough to use rolling means in scoring the water body, the boundary values between classes are half those of annual percentage changes (Table 6).

**Table 6: Metric system developed for the classification of changes in rolling mean of angiosperm shoot density/% cover.**

| | *5 year rolling mean change* |
|---|---|

| Quality Status | Disturbance | Exemplar metric scores for % loss of density from reference conditions | Metric EQR |
|---|---|---|---|
| High | No detectable change | 0% – 5% loss | 1.0 – 0.8 |
| Good | Slight signs of disturbance | >5% – 15% loss | 0.8 – 0.6 |
| Moderate | Moderate distortions | >15% – 25% loss | 0.6 – 0.4 |
| Poor | Major distortions | >25% – 35% loss | 0.4 – 0.2 |
| Bad | Severe distortions | >35% – 100% loss | 0.2 – 0.0 |

The metric also works on a sliding scale to enable an accurate EQR value to be calculated for this particular parameter using both values for annual changes and rolling means.

**Calculation of the ecological quality ratio**

The ecological quality ratio (EQR) for the parameter, % loss of shoot density, should be calculated using the following equation whereby "value" signifies the observed value:

$$\textbf{EQR = upper EQR parameter range} - \left\{ \left[ \frac{\textbf{value – lower class range}}{\textbf{class width}} \right] \textbf{x EQR band width} \right\}$$

***Example*** using a value of 17 for % loss of shoot density, consult Table 6: 17 lies between 15-25 and with an EQR between 0.4-0.6 therefore:

Score = 0.6 - (17 – 15)/10 x 0.2

Score = 0.6 – 0.04 = 0.56

### *5.4.4 Ecological Status: Combining the Metrics*

As taxonomic composition and abundance may be considered to be equally significant in a water body's overall ecological status, it was considered appropriate that the final classification for a WB should be determined as an average of all the metrics, rather than taking simply the lowest metric outcome. The potential consequence of the latter approach could be the classification of an entire water body as Poor status, based on the outcome of one metric. The overall class boundaries are evenly spaced and are shown in Table 7. All metrics should be used, again in order to provide an accurate and balanced assessment of the water body.

Table 7: Overall ecological status boundaries for the intertidal seagrass tool

| Status | EQR |
|---|---|
| High/Good | 0.80 |
| Good/Moderate | 0.60 |
| Moderate/Poor | 0.40 |
| Poor/Bad | 0.20 |

## 5.5 Application of the Tool

This tool is specifically geared towards the monitoring of sedimentary shores which may be both coastal and transitional. While the development of the metrics includes data from both intertidal and subtidal beds, the tool at present will relate only to intertidal beds, due mainly to the practical difficulties of obtaining subtidal seagrass data. The exact survey methods used may vary depending on the extent and accessibility of the bed, but for a survey to take place there must also be evidence of seagrass presence. Therefore, the methods developed for the seagrass tool have been based on a tiered procedure. These methods follow the requirements of the multi-metric tool.

### *5.5.1. Preliminary Assessment*

The first stage of this process requires a preliminary risk assessment in the form of a desk based data collation exercise. This aims to assess the potential or current pressures faced by a particular water body such as habitat loss or eutrophication,, and the designation of specific areas e.g as Nitrate Vulnerable Zones (NVZs; Nitrates Dir., 1991) or Urban Waste Water Treatment Directive (UWWTD, 1991) Sensitive Areas. This process will also highlight any drivers related to these risks such as the Nitrates Directive, OSPAR (OSPAR, 2003), the UWWTD and the Habitats Directive (Habitats Dir.1992, . Finally it aims to produce an historical baseline, where possible, using various forms of data such as previous surveys, aerial photos, water quality data, information on sediments, the general extent of the available habitat and the geographic distribution from the NBN. This enables a picture to be established of the general area of interest. If there is no available sedimentary intertidal area, or if light penetration is known to be limited, there is little chance of a seagrass bed establishing. Therefore it is advisable to gather information on the probable location and extent of seagrass beds to inform a preliminary site visit. This may include aerial photographs, satellite or other imagery, or a general visual assessment.

### *5.5.2 General Sampling Considerations*

The influence of sampling scale and survey method on the prediction of coverage and ecological attributes of seagrass beds dictates that managers and scientists need to choose sampling designs carefully (Fonseca *et al*., 2002). For detecting seagrass bed spatial extent and large-scale features, sampling over a large area (~100s metres) appears to be the most appropriate strategy. Aerial photographic surveys may be appropriate for intertidal beds to help determine bed size, though an appropriate level of ground-truthing would be necessary. Video transects would be suitable for subtidal beds. Conversely, ecological attributes of the seagrass bed such as percentage cover are best characterised by sampling at a finer scale, e.g. < 50 m, using a gridded system of transects (Fonseca *et al*., 2002). The distance between transect lines is dependent on the overall size of the bed and the total number of sample points is dependent on variation within the bed, the resource available (e.g. number of field workers) and the period of time the site is exposed at low water. Successful littoral density surveys have been carried out in UK seagrass beds based on such a system (e.g. Boyes *et al*., 2005). Alternatively a random, stratified sampling approach may be taken.

Surveys should be conducted over a set period under the same conditions and standardised for each repeat survey in order that natural seasonal cycles in seagrass presence/abundance are considered. It is important that comparisons between years are based on sampling performed at the same time of year whenever biomass usually attains its annual maximum (Olesen & Sand-Jensen, 1994). It is recommended for the purposes of the WFD that annual monitoring should take place during the bloom period for seagrass. This is likely to fall in August or early September for most parts of the UK: this may vary geographically or with climatic conditions, but is defined here as June to September inclusive.

Naturally occurring local events need to be considered in the sampling protocol. In some instances wildfowl exploit littoral seagrass beds, e.g. Fenham Flats in Lindisfarne Bay, Burry estuary, Swansea (Kay, 1998) and Strangford Lough (Portig *et al*., 1994). Grazing by Brent geese (*Branta bernicla*) occurs in Strangford during the early part of the winter and biomass is reduced considerably. Such local occurrences must be taken into account when deciding appropriate dates so that the peak bloom period is captured, without other events having had an opportunity to reduce the biomass. Any permissions for protected nature conservation sites should be considered before surveys take place.

Ideally sampling would take place annually, at least until some idea of natural inter-annual variability is obtained, as this can be signficant (Duarte, 1989; Duarte & Kirkman, 2001). Measurements are best carried out for at least three years. Experience has shown that second year results may show greater bed number and extent than first year survey results; this is due to the increased skill on the part of surveyors. Baseline data should be determined from the year of greatest bed extent from the initial set of surveys (where not using historical data). Baseline % cover data would be those linked to the year of greatest bed extent.

### 5.5.3 Abundance - Bed Extent

Initial mapping surveys may provide baseline information for monitoring programmes. Geographic Information System (GIS) base-maps provide a quick, precise drafting and mapping tool and the best data presentation, analysis, interpretation and storage systems (McKenzie *et al*., 2001). Seagrass resources can be mapped using a range of approaches from *in situ* observation to remote sensing. The choice of technique is scale and site dependent and a range of approaches may be used. All seagrass mapping should be ground-truthed to evaluate image signatures of the remotely sensed data, to examine areas where the imagery does not provide information and to produce reference information and accuracy assessment.

Aerial photography is the most common remote sensing method for seagrass mapping studies (McKenzie *et al*., 2001) and offers the means for monitoring over time (Agostini *et al*., 2002). Recent and historical photographs have been used to study long-term changes in seagrass bed spatial extent with great success (e.g. Kendrick *et al*., 2002; Agostini *et al*., 2002). Boundary maps of seagrass beds may also be generated from *in situ* surveys using Geographic Positioning Systems.

The resolution of remotely sensed satellite imagery is generally too limited for detecting patterns in intertidal seagrass bed density, and it is often a prohibitively expensive technique for mapping bed extent (McKenzie *et al*., 2001). Aerial photography is a preferable remote sensing method and is often employed successfully for surveys of large seagrass beds, in combination with thorough ground-truthing (e.g. Frederiksen, *et al*., 2004). Ground-truthing is essential because areas need to be examined where the imagery is incomplete and features with similar signals need to be distinguished; e.g. macroalgae can be mistaken for seagrass (McKenzie *et al*., 2001). In the UK, by contrast, typical seagrass beds are small (100s metres to a few $km^2$) and ground surveying entire seagrass beds is often feasible during a single tidal cycle.

### 5.5.4 Abundance - Shoot Density (% cover)

Seagrass bed density is a measure of the percentage cover (intertidal) or number of shoots of seagrass in an area (subtidal). As noted above, this is best characterised by fine-scale sampling using a gridded system of transects (Fonseca *et al*., 2002). There are significant practical differences in sampling a littoral or sublittoral bed to be taken into account when sampling seagrass density; i.e. accessing littoral beds at low tide if the substratum is firm enough

Estimates of % cover can be a reliable alternative to counting the number of shoots (Kirkman, 1978). Photographic standard ranks of shoot density may be used to estimate cover and this method has been found to reduce the differences in estimates between observers (Kirkman, 1978). Leaf cover does not always provide an adequate comparison between species because populations of small seagrass species tend to be denser than those of large ones (Duarte & Kalff, 1987). Consequently a photographic standard rank procedure is recommended to aid with abundance estimations to reduce discrepancy among surveyors (Kirkman, 1978). This is established by producing a series of quadrat photographs from minimum to maximum peak cover with a consensus reached on the % cover from within each quadrat which can be set as references. Standard ranks for % cover and photographs are best determined for the time of peak seasonal biomass, to ensure that subsequent maximum biomass values used to establish the ranks fall within the range of measured biomass values used to establish the regression. A set of standard ranks can be used in different locations, provided species mix and biomass ranges are similar. Different ranks may be needed for different species where stands are mono-specific. It is recommended that a photograph is taken of each quadrat: these can be used later for quality control purposes.

### *5.5.5 Taxonomic Composition*

As previously noted, the taxonomic component of the metric requires identification of three species of seagrass, *Zostera* (*Z. noltei*, *Z. marina* and *Z. angustifolia)* and *Ruppia* to genus level. A relatively low level of identification expertise is required of field workers to implement this index.

## 5.6 Data collection

To ensure consistent collation and interpretation of data collected within the field, data should ideally be recorded and stored in a similar format. Figure 4 shows an example of a hard copy field record form. Alternatively GPSs with data dictionaries may be used, with information recorded directly in the field.

Consistent data storage methods will allow accurate calculation of the final ecological quality status for the seagrass multi-metric tool.

Consideration should be given to training of surveyors and their initial and continuing competence to carry out assessments to a required level of accuracy.

**Figure 4:** Example recording sheet - manual method.

| Surveyor/s & organisation: | | |
|---|---|---|
| Seagrass bed name: | Physiography: | WFD typology: |
| Seagrass meadow description: | | |
| Site: | Start time (GMT): | Finish time (GMT): |
| Date: | No. quads sampled: | Tidal state: |
| Quadrat size: | $H_2O$ Temp: | Height/depth: |
| Salinity: | GPS Easting: | GPS Northing: |

| **Transect No:** | |
|---|---|
| GPS Start<br>Easting:<br>Northing: | GPS Finish<br>Easting:<br>Northing: |
| Distance between quadrats: | |

| **Quad No.** | **% Cover** | | | | | **Photo #** | **GPS way point** | **Notes** |
|---|---|---|---|---|---|---|---|---|
| | Sp: | Sp: | Sp: | Other (specify) | Bare | | | |
| | | | | | | | | |
| | | | | | | | | |

## 5.7 Summary of Classification Process

The full classification process can be more clearly understood following the flow chart below (Figure 5).

| *Work Area* | *Considerations* |
|---|---|
| Monitoring Design | • Are seagrass beds present in the water body/have they been present historically?<br>• Are there historical records for setting reference conditions ?<br>• Can trends be assessed? |
| ↓ | |
| Sample collection | • Use of standardised methods<br>• Identify seagrass taxa (*Zostera* species and *Ruppia*) present in water body<br>• Identify bed extent (all beds in water body at >5% cover of seagrass, optional for <5% boundary)<br>• Identify percentage cover through quadrats representing all beds; take photos of at least 10% of quadrats.<br>• Record any negative impacts/signs of disturbance.<br>• Collate historic information |
| ↓ | |
| Sample analysis / Data treatment | • Quality assurance of percentage cover in photographed quadrats<br>• QA and orthorectification of any remote imagery used in mapping of beds and density<br>• Calculation of individual bed extents (ha)<br>• Calculate change in bed extent<br>• Identify taxa and calculate taxa change<br>• From quadrats identify change in shoot density |
| ↓ | |
| EQR calculation | Calculate EQR<br><br>*Final Equidistant index score = Upper Equidistant Class range value – ((Face Value - Upper Face value range) * (Equidistant class range / Face Value Range))* |
| ↓ | |
| Water body classification | • Derive WB average EQR<br>• Assign Class Status (use defined boundaries)<br>• Calculate Confidence of Class (“Risk of Misclassification”) |

**Figure 5:** Flow chart summarising the main stages involved in undertaking and assessment using the intertidal seagrass tool

## 5.8 Worked Examples

For confidence in the final classification, assessments of seagrass beds should be conducted using *all* of the metrics; taxonomic composition, and abundance determined by (a) bed extent and (b) density. Presented here are examples of intermediate scores for each of the individual metrics, followed by examples of the overall ecological status for water bodies, calculated by combining their results.

Although this tool is designed for intertidal seagrass beds, initial work included data from subtidal also.

### *5.8.1 Taxonomic Composition*

Of the three proposed seagrass assessment metrics, taxonomic composition is likely to be the simplest assessment to make of a seagrass bed. This metric was tested against several seagrass beds in the UK, comparing their historic and current taxonomic compositions with data sourced from literature. Table 8 summarises the results of testing the metric against a variety of littoral and sublittoral UK seagrass beds, their scores and references to data sources.

Table 8: Taxonomic composition metric tested against a variety of UK coastal (CW) and transitional (TW) water bodies with littoral or sublittoral seagrass beds.

| Water body | Seagrass site | Species historically recorded | Species most recently recorded | Composition change | Index score | Reference |
|---|---|---|---|---|---|---|
| Scilly Isles CW | Five beds in the Isles of Scilly | *Z. marina* | *Z. marina* | No loss | 0.9 | Cook, 2005 |
| Helford TW | East of Passage Cove | *Z. marina* | *Z. marina* | No loss | 0.9 | Hocking & Tompsett, 2002 |
| | Helford Creek | Z. angustifolia<br>*Z. noltei* | *Z. noltei* | Half no. of species lost | 0.5 | Spooner & Holme, 1986<br>Covey & Hocking, 1987<br>Hocking & Tompsett, 2002 |
| South Pembroke-shire CW | North Haven, Skomer Island | *Z. marina* | *Z. marina* | No loss | 0.9 | Lock, 2003<br>Burton *et al*., 2005 |
| Strangford Lough North CW | Newtownards to Rough Island | *Z. angustifolia*<br>*Z. noltei* | *Z. angustifolia*<br>*Z. noltei* | No loss | 0.9 | Portig *et al*., 1994 |
| Milford Haven CW | Sandy Haven Pill | *Z. angustifolia* | *Z. angustifolia* | No loss | 0.9 | Davis, 1961 in Kay, 1998 |
| Langstone Harbour CW | Hayling Island littoral flats | *Z. angustifolia*<br>*Z. noltei* | None | Total loss | 0.1 | Den Hartog, 1994 |

### *5.8.2 Abundance - Bed Extent*

Examples of scoring the seagrass bed extent metric are presented. The first two are examples for intertidal beds, while the latter is for subtidal.

#### 5.8.2.1 Milford Haven Intertidal Seagrass Beds

At Angle Bay, in Milford Haven, the largest of 3 *Zostera noltei beds* was surveyed along a transect for bed extent and density. The bed is approximately 300m wide (parallel with the waterline) and 200m deep (perpendicular to the waterline). Seagrass bed A was surveyed along one straight-line transect only, across the widest part of the bed, approximately parallel with the waterline. Digital photographs of $1m^2$ quadrats were taken at 10m intervals. The perimeter of the bed was mapped once using a hand-held GPS.

Seagrass bed B, also in Angle Bay, was surveyed more thoroughly:

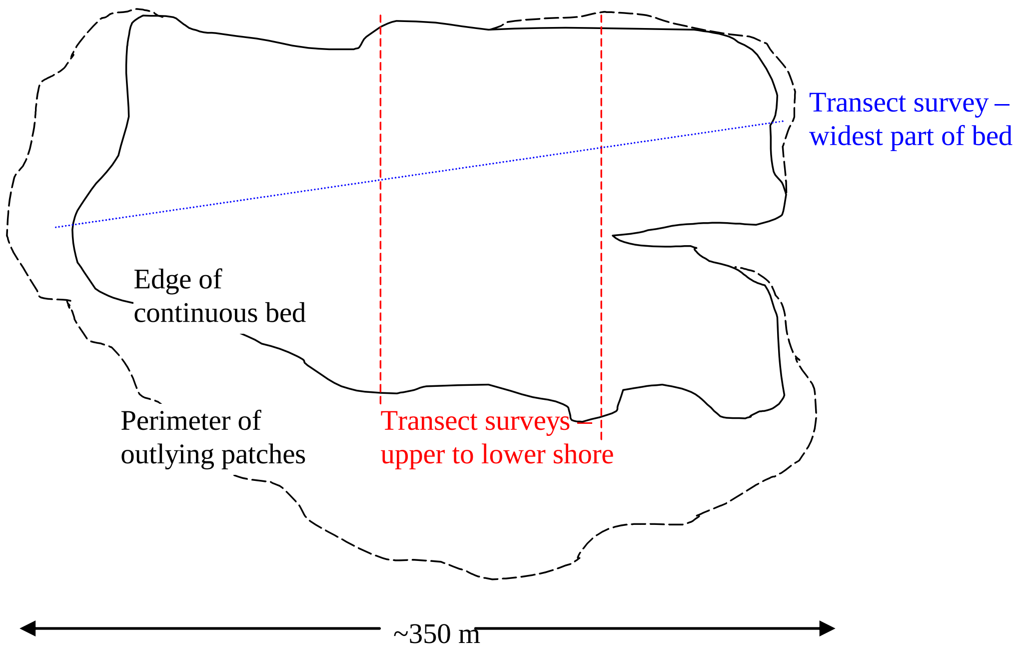



Figure 6: Angle Bay inter-tidal seagrass bed B.

Three transects were surveyed; one across what was considered to be the widest part of the bed, and 2 from the upper to the lower shore, perpendicular to the waterline (Figure 6). $1m^2$ quadrats were laid so the left hand side of the quadrat lay against the tape with the bottom left-hand corner at the 10m-interval point in the tape. The quadrats were digitally photographed for subsequent lab analysis of % cover.

The perimeter of the continuous bed was mapped using a GPS and also the perimeter of the area where seagrass was intermittent and patchy. Notes were taken where there was evidence of anthropogenic impact; for example bait-digging holes, vehicular wheel tracks, propeller scour, anchor chain dragging or scour, etc.

Notes were taken of sediment type (including soft/firm as this affects access by fieldworkers). Also records were made of opportunistic algal cover (e.g. *Ulva*, *Enteromorpha*, *Chaetomorpha*, *Ectocarpus*, etc ) with regard to location, size of patch and thickness of cover.

Bed extent was recorded and results were as follows:

| Bed | Area | Anthropogenic pressure |
|---|---|---|
| Angle Bay bed A | 538959.55 $m^2$ | Bait digging |
| Angle Bay bed B (entire bed) | 30497.19 $m^2$ | |
| Angle Bay bed B (continuous bed) | 24169.00 $m^2$ | |

No data were available of historic bed extent at these sites for comparison with 2004 surveys. However, anecdotal evidence was available from a CCW representative who confirmed the two beds have been stable in extent. They may be close to their maximum potential extent, though this is difficult to confirm, as there are sections of the bay that appear to be no different, but do not support *Zostera.* There was no or minimal evidence of direct anthropogenic impact.

Provisional Classification for bed extent is GOOD

### 5.8.2.2 Pembrokeshire Intertidal Beds

The CCW review of the knowledge of seagrass beds around the coast of Wales (Kay, 1998) can be used in the classification of the current status of some seagrass beds, with regard to their historic extent. An example is Sandy Haven Pill, Milford Haven, first discovered in 1958 and described as forming a narrow belt 400 yards long (Davis, 1961). By 1995 CCW files describe only two remaining patches each of 1 x 0.5 m with *Spartina* sp. 1-4 metres seaward of these (Kay, 1998). Although the precise width of the seagrass bed in 1958-1961 is not described, it is apparent that bed extent has decreased significantly. If the bed in 1958-1961 was as narrow as 0.5 m, the total area would have been 200 $m^2$ (which would be the baseline, or reference condition, for this bed), whereas the spatial extent of the two remaining patches in 1995 was only 1 $m^2$. Referring to Tables 4 and 5, Sandy Haven Pill seagrass beds would score 0.0 for this metric.

### 5.8.2.3 Pembrokeshire Subtidal Beds

Sites where *Zostera* has been recorded, but has since disappeared, include; Dale (George, 1958 in Kay, 1998), Pwllcrochan (Davis, 1971 in Kay, 1998), Garron Pill (Knights, 1979 in Kay, 1998) and Landshipping (Davis, 1962 in Kay, 1998). Complete loss of a seagrass bed would score all of these areas at 0.1 for this metric.

#### 5.8.2.4 North Haven, Skomer Marine Nature Reserve, Pembrokeshire

The SMNR management plan (Newman *et al*., 2000) establishes favourable conditions as a bed extent of 6700 $m^2$ with a lower level of acceptable change of 5500 $m^2$ (Lock, 2003; Burton *et al*., 2005). The population boundary mapping occurs every two years.

*Zostera marina* bed area has been measured for five years between 1982 and 2004 as shown in Figure 7. The area calculations have been made either from the abundance and distribution maps or from the GPS maps (Lock, 2003; Burton *et al*., 2005). These reports state the accuracy of the abundance and distribution areas, as mapped by divers, is high, but the maps derived from GPS are less accurate. No error statistics are presented in Figure 7, as raw data are unavailable. It is possible that the bed is at its potential maximum size due to restrictions in expansion by unsuitable substratum to the south and deep water to the north.

Skomer Island's seagrass bed appears to be both relatively stable in area and at or near its maximum physical extent. Since 1991 *Z. marina* in the MNR is protected, with restrictions on anchoring and fishing. The site would be scored at 1.0 for this metric, using the proposed system for determining and scoring extent (see Table 2 and 3).

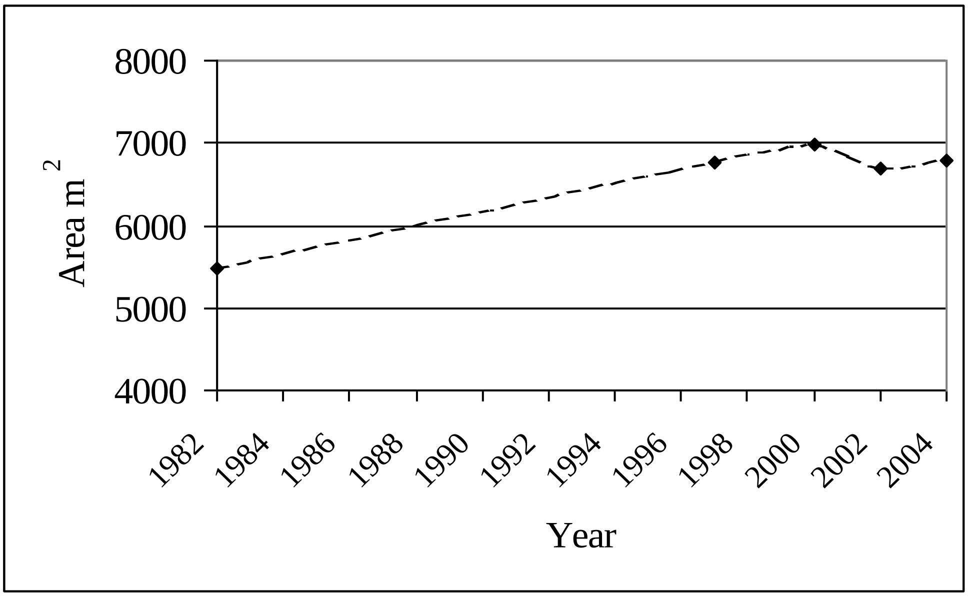



**Figure 7:** North Haven *Zostera marina* bed area; 1982, 1997 and 2002 areas mapped by divers with high accuracy, 2000 and 2004 areas derived from boat GPS are less accurate (Burton *et al*., 2005) (raw data unavailable for calculation of error statistics).

### 5.8.3 Abundance - Shoot Density

Three methods of surveying shoot density/percentage cover and three methods for examining the annual mean density data are presented, with examples.

### 5.8.3.1 Milford Haven Intertidal Beds

As previously described in section 5.8.2.1 two intertidal beds in Angle Bay, Millford Haven were surveyed. Shoot density was recorded as % leaf cover in $1m^2$ quadrats along 3 transects (see Figure 6). Assessment of percentage cover of seagrass was complicated by the presence of opportunistic macroalgae. Abundance, as percentage cover, of opportunistic macroalgae on seagrass was also recorded. Results are presented in Table 9.

**Table 9: Percentage cover results for Angle Bay**

| | % Cover | | | | | |
|---|---|---|---|---|---|---|
| | Transect 1 | | Transect 2 | | Transect 3 | |
| | Seagrass | Algae | Seagrass | Algae | Seagrass | Algae |
| **Mean values (%)** | 27.4 | 5.7 | 6.9 | 0.3 | 33.9 | 0.6 |

Overall percentage cover would be calculated using patch size. As there are no temporal data for this tool it is possible to calculate neither a rolling mean of % cover, nor annual % loss or gain. However, the survey provides useful baseline data for future studies. Mean opportunistic macroalgal cover was <10% on all transects across the bed. There is no recorded loss of seagrass species.

### 5.8.3.2 North Haven, Skomer, Subtidal Beds

The subtidal *Z. marina* bed in North Haven, Skomer Island has been surveyed regularly (Lock, 2003; Burton *et al*., 2005). Part of the SMNR management plan (Newman *et al*., 2000) aims to maintain the population of *Z. marina* in North Haven in favourable condition whereby shoot density does not fall below the 1997 survey mean of 36 shoots $m^{-2}$.

Divers established transects at 10 m intervals and completed seagrass shoot counts in quadrats at 5 m intervals along the transect lines. Count data were converted to mean values per square metre and mapped using ‘vertical mapper’ software at intervals of 10 shoots $m^{-2}$ to show the distribution and density of *Z. marina*. In 1997 the mean density was 36.2 shoots $m^{-2}$ and this increased to 54 shoots $m^{-2}$ in 2002. The percentage frequency of 50 shoots $m^{-2}$ or greater was 38% in 1997 increasing to 56% in 2002; in 2002 8% of quadrat counts were recorded as 100 shoots $m^{-2}$ or greater, whilst in 1997 this was less than 1% (Lock, 2003; Burton *et al*., 2005). Such an increase in density above previously highest recorded levels would score the site as 1.0 for this index, using the proposed system for seagrass bed spatial extent (see Table 5) and would reset the maximum (reference) against which future assessments are made

### 5.8.3.3 Isles of Scilly Subtidal Seagrass Beds

An annual diving expedition recorded a variety of parameters in several Isles of Scilly seagrass beds, for 10 years (Cook, 2005). The five main beds are: Old Grimsby Harbour, Tresco; Higher Town Bay, St. Martin's; Broad Ledge, Tresco; West Broad Ledge, St Martin's, and; Little Arthur, Eastern Isles. Percentage leaf cover and shoot density (shoots $m^{-2}$) are two of the expedition's monitoring parameters. Density is determined by counting all the seagrass shoots in a 25 x 25 cm quadrat, the position determined by randomly generated bearings and distances from a central datum point in each bed. Figure 8 (a-e) illustrates the mean annual shoot density of these five *Zostera* beds, with standard error bars shown for the most recent five years (raw data prior to 2001 were unavailable for calculation of error statistics). Shoot density data presented in this form show considerable fluctuation and longer term underlying trends of losses or gains in density are not always easy to identify. Two further methods for examining the data to help determine such trends are presented in Figure 9 (a & b); the rolling 5-year mean densities for each bed and the annual mean seagrass density for the whole Isles of Scilly as one water body, respectively, with standard error bars shown. For individual beds underlying trends are more easily recognised using a rolling mean; e.g. in Figure 8 (a) mean annual shoot density $m^{-2}$ in Old Grimsby Harbour shows an increase of 40 shoots to a density of 117 shoots $m^{-2}$ between 2002 and 2003 and then a decline over the next two years. Whereas the 5-year rolling mean (Figure 9 (a)) remains unchanged between 2002 and 2003 and then increases in 2004 and again in 2005, reflecting the overall increase since 2000. This technique is suitable for water bodies with only one or two small seagrass beds, i.e. approximately of the size that can be reasonably surveyed during one tidal cycle.

Presentation of these Isles of Scilly data as 5-year rolling means is essentially illustrative. The seagrass beds in the Isles of Scilly are all in close proximity, are subtidal, consist of the same taxon and all fall into one water body so are affected by the same general hydrodynamic and environmental regime. Therefore, it is recommended the data are combined and annual mean changes in density are reported as a whole for the 'Scilly Isles' CW, thereby reducing the noise of natural variability (Figure 8 (b)). When all seagrass beds are considered as a whole in this manner, the pattern of variability in any one bed will be somewhat attenuated and it is possible to establish the overall status of seagrass in the Isles of Scilly in any one year, without recourse to a 5-year rolling mean.

When the 5-year rolling mean method is employed, beds are scored by comparing current shoot density with the previously highest recorded shoot density for the seagrass bed in question. To illustrate this, 2005 density data for the five Isles of Scilly seagrass beds have been compared with previous records and index scores assigned accordingly (Table 10). An overall score for the whole 'Scilly Isles' water body has also been assigned by determining the difference in mean density of all beds in 2005 from the previously highest recorded. *Zostera* beds have been scored with reference to the scheme in Table 4, note the difference in percentage ranges for annual means as opposed to 5-year rolling means in Table 5 and 6.

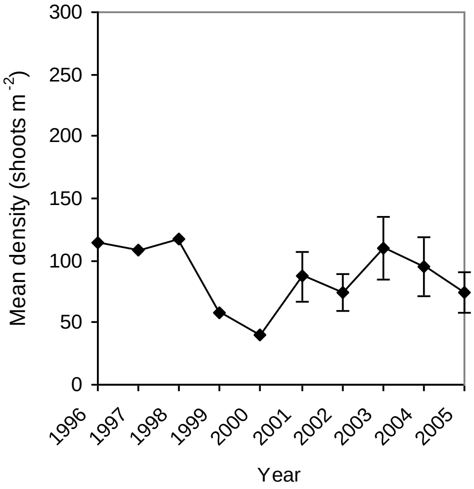



Figure 8 a

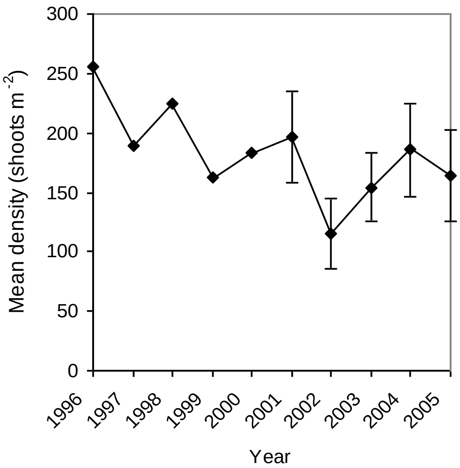



Figure 8 b

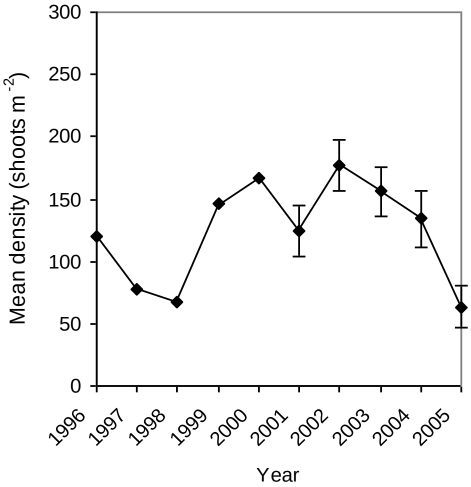



Figure 8 c

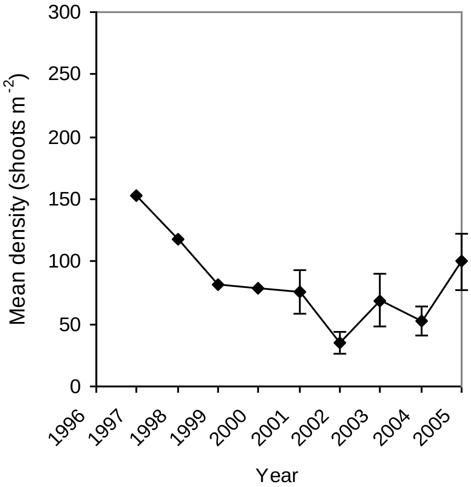



Figure 8 d

Figure 8 e

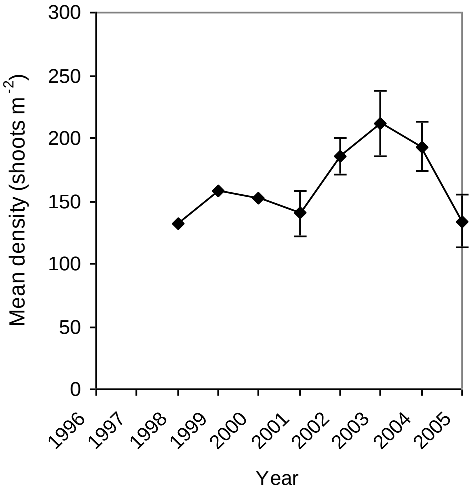
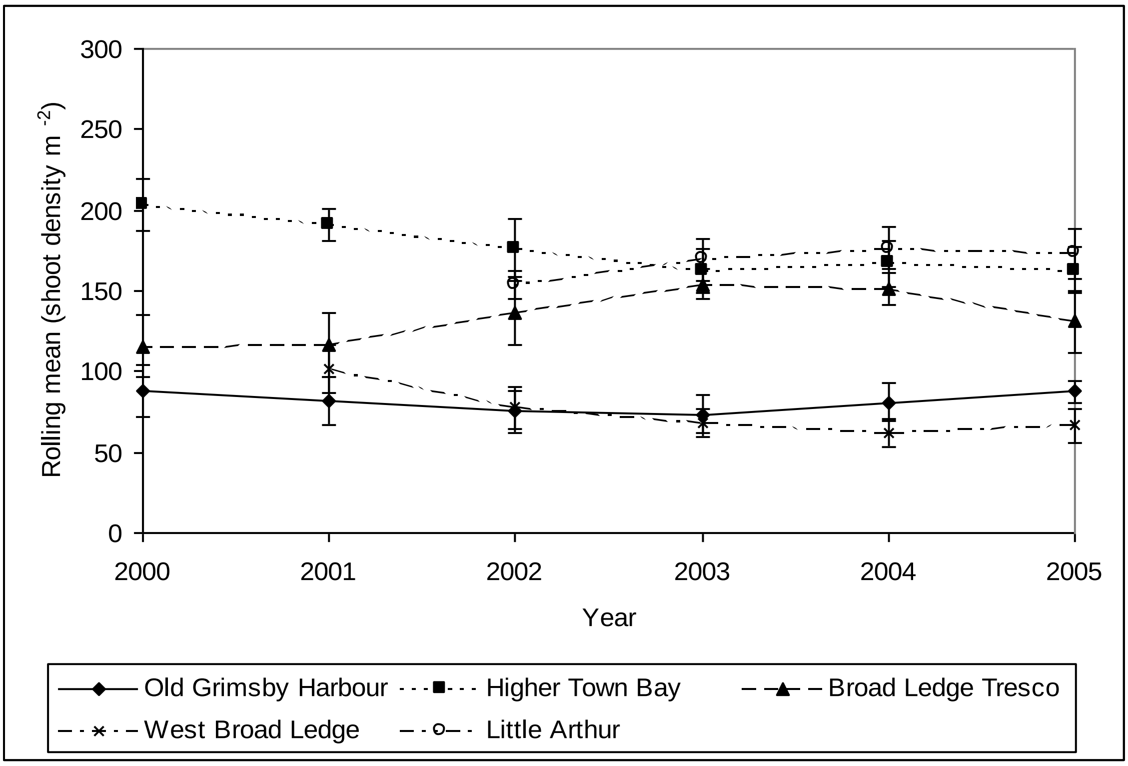



**Figure 8 a-e**: Annual mean shoot density in five Isles of Scilly sublittoral seagrass beds with standard error bars; (a) Old Grimsby Harbour, (b) Higher Town Bay, (c) Broad Ledge Tresco, (d) West Broad Ledge, and (e) Little Arthur (data from Cook, 2005). Note data only available for 9 and 8 years for West Broad Ledge and Little Arthur, respectively.



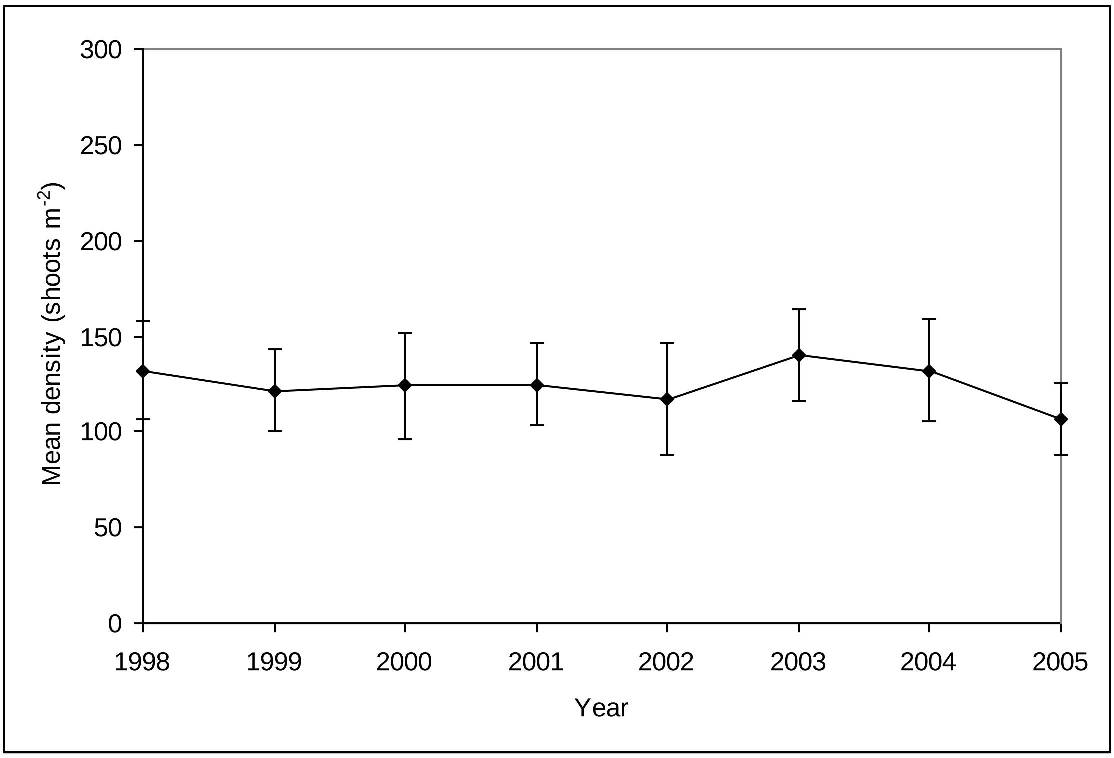




**Figure 9a & b:** Shoot density in Isles of Scilly sublittoral seagrass beds; (a) rolling 5-year mean of shoot density of five beds, and (b) annual mean shoot density for all seagrass beds in the Isles of Scilly water body (data from Cook, 2005). Note different time periods on *x*-axes

**Table 10:** Abundance determined by shoot density index, tested against individual Isles of Scilly seagrass beds and as an overall mean for the water body. Density (shoots $m^{-2}$) reported to 3 significant figures.

| **Site** | Highest recorded 5 yr rolling-mean density (shoots $m^{-2}$) and year | 2005 5 yr rolling-mean density (shoots $m^{-2}$) | 2005 % difference from previous highest density | Abundance (density) score |
|---|---|---|---|---|
| Old Grimsby Harbour, Tresco | 87.9 (2005) | 87.9 | 0 | 1.00 |
| Higher Town Bay, St. Martin's | 203 (2000) | 163 | -19.8 | 0.50 |
| Broad Ledge, Tresco | 154 (2003) | 131 | -14.9 | 0.75 |
| West Broad Ledge, St. Martin's | 101 (2001) | 66.3 | -34 | 0.25 |
| Little Arthur, Eastern Isles | 177 (2004) | 173 | -2 | 0.75 |
| | Highest recorded annual mean density (shoots $m^{-2}$) and year | 2005 annual mean density (shoots $m^{-2}$) | 2005 % difference from previous highest density | Abundance (density) score |
| Whole 'Scilly Isles' CW | 140 (2003) | 107 | -24 | 0.75 |

### 5.8.4 Overall Ecological Status; Combining the Metrics

To assign a water body's overall ecological status for seagrass all three metrics must be used and the final classification for this biological quality element uses the previously described scoring systems. Examples of UK seagrass beds that have been assessed using the indices are presented in Table 11. This shows the importance of using all metrics to give a truer estimation of class and increase confidence in the assessment.

**Table 11:** Overall ecological status of three exemplar UK seagrass beds, calculated from mean scores of assessment metrics.

| Assessment Index | **Metric scores & Ecological status** | | |
|---|---|---|---|
| | Site A | Site B | Site C |
| Taxonomic composition | 0.90 | 0.90 | 0.90 |
| Seagrass bed density | 1.00 | 0.75 | - |

| | | | |
|---|---|---|---|
| Seagrass bed extent | 1.00 | - | - |
| Mean score | 0.97 | 0.83 | 0.90 |
| Final classification | **High** (with high confidence) | **High** (with low confidence) | **High** (with very low confidence) |

The concept of using seagrass as an ecological indicator is relatively new and to date there has been no national monitoring programme of littoral or sublittoral seagrass in the UK. These two issues have complicated the establishment of reference conditions, ranges and boundaries for each ecological status. There is a wide variety of naturally occurring physical and hydro-morphological conditions in UK TWs and CWs and a seagrass bed's taxonomic composition and abundance are a product of individual combinations of local conditions. For this reason reference conditions cannot be type-specific across national water body typologies. Where accurate and quantifiable historic data from trustworthy records exist, reference conditions may be established. If only recent survey data are available (e.g. < 5 years old) the previously recorded healthiest condition of a seagrass bed becomes its reference condition (based on largest bed extent). If the bed is found to be expanding and improving in quality, this may indicate recovery and positive scores will be recorded.

These metrics are likely to require continuous refinement to ensure the final boundary values accurately interpret the data collected and the final assigned quality status truly represents the state of the water body. The WFD permits revision of biological quality elements' classification schemes each reporting cycle (six years). Data collation will continue to aid this process and will inform any future refinements.

Where more than one angiosperm BQE (i.e. seagrass and saltmarsh) occurs in a WB, the overall assessment for the biological quality element of angiosperms will be the lower of the two.

# 6. Response to Pressures

The WFD requires the characteristics used in the assessment of water bodies to show evidence of response to changes in the natural environment through both direct and indirect pressures such as;

- abstraction & flow regulation,
- morphological pressures & alterations more specifically habitat modification,
- point source discharges or general pollution,
- increased nutrients leading to eutrophication, and
- presence of alien taxa.

The primary pressures thought to cause a shift in the balance of angiosperm communities are hydromorphological change, excess sediment deposition, physical impact (e.g. bait digging, fishing, anchoring), habitat loss, increased nutrient concentrations, and to a slightly lesser degree animal grazing

In general as pressures increase on seagrass beds there is an overall loss of ecological quality seen as:

- Decreasing extent of  bed and density of plants
- Loss of biomass
- Loss of sensitive seagrass-dependant species

## 6.1 Habitat Loss

This may be through hydromorphological change, excess deposition of sediments or physical removal of habitats including such processes as “coastal squeeze” due to flood defence structures or rising sea levels.
All of these may lead to loss of habitat particularly on the external perimeters of seagrass beds. Angiosperms are very selective with their growth conditions, so removal or degradation of suitable habitats can have a long term effect on seagrass beds, whereby they often show no indication of subsequent recovery.

## 6.2 Excess Suspended Solids

Increased suspended particulate matter can lead to severe smothering and light limitation. This may also induce a transition in community structure with an increase in grazing activity and subsequent loss of specific taxa. Increased sedimentation may lead to an increase in the abundance of filter feeding and general grazing activity, proving detrimental to the underlying vegetation.

## 6.3 Increased Nutrient status

Marine plants are a key component of the ecology of shallow coastal and transitional water environments.  In healthy shallow coastal waters with a balanced nutrient regime the dominant primary producers are perennial benthic macrophytes such as seagrass or long-lived seaweeds, with seasonal opportunistic macroalgae or phytoplankton playing a lesser role in biomass and production (Schramm & Nienhuis, 1996).

Increased nutrient inputs from both direct and indirect sources such as sewage outfalls and land run-off contribute to potential eutrophication problems and increased suspended sediment levels. These may exacerbate the growth of opportunist species with consequential smothering of seagrass beds ,whereby the entire benthic vegetation can become shaded and vegetation cover, biomass and depth distribution decline (Duarte 1991, Nielsen et al 2002b, Valiela et al. 1997). Smothering and anoxia under thick, persistent opportunistic macroalgal mats will cause seagrass shoots to thin and bleach. In the worst case scenarios seagrass beds will finally disappear.  Moreover, a reduced cover of benthic vegetation may cause increased resuspension of bottom material due to less stabilisation of the sediment which further reduces water clarity. Nutrient load therebfore may markedly influence both nutrient concentration, algal growth and water clarity. The changes in benthic vegetation due to eutrophication are a series of direct and indirect affects that feedback and self-accelerate, and are which are difficult to control once initiated (Schramm & Nienhuis, 1996). Additional responses to increased nutrients lay in undesirable shallow anoxic level as well as excess suspended particulate matter resulting from increased nutrients and runoff leading to light limitation, smothering and depth restriction of seagrass growth (Duarte 1995).

## 6.4 Example of pressure response

Unfortunately there is a lack of historical and long time-series data for any given pressure. The whole BQE response range is not covered for any given pressure. There are also no datasets across the whole range of different pressures. Some hydromorphological pressures can be acute, spasmodic & irregular (e.g. storms, bait digging, anchor chains) making it difficult to show the relationship against pressure gradient. It is also likely that several pressures may co-exist, making single pressure impact assessment impossible. However there follows an extreme example of deliberate and thorough clearing of a seagrass bed and its subsequent recovery.

**Morecambe Bay**: loss of seagrass density due to anthropogenic impacts

Construction of 2 pipelines required the clearance of parts of the *Zostera* bed at Westfield, Morecambe Bay. A vegetational survey of the area to be affected was undertaken in 1992 prior to engineering works. Recovery of the cleared sites has been followed by annual surveillance studies (Tittley *et al*., 1998). Plotting these data (Figure 10) provides a visual illustration of base-line abundance in 1992, depletion between 1993 and 1996/1997, and indication of recovery in the 2/3 most recently surveyed years.

The surveys along 7 transects show natural variability in the density (% leaf cover) of seagrass during the baseline survey, 1992. Anthropogenic impact was high because clearance was deliberate, and recovery at naturally variable rates is evident in all transects.

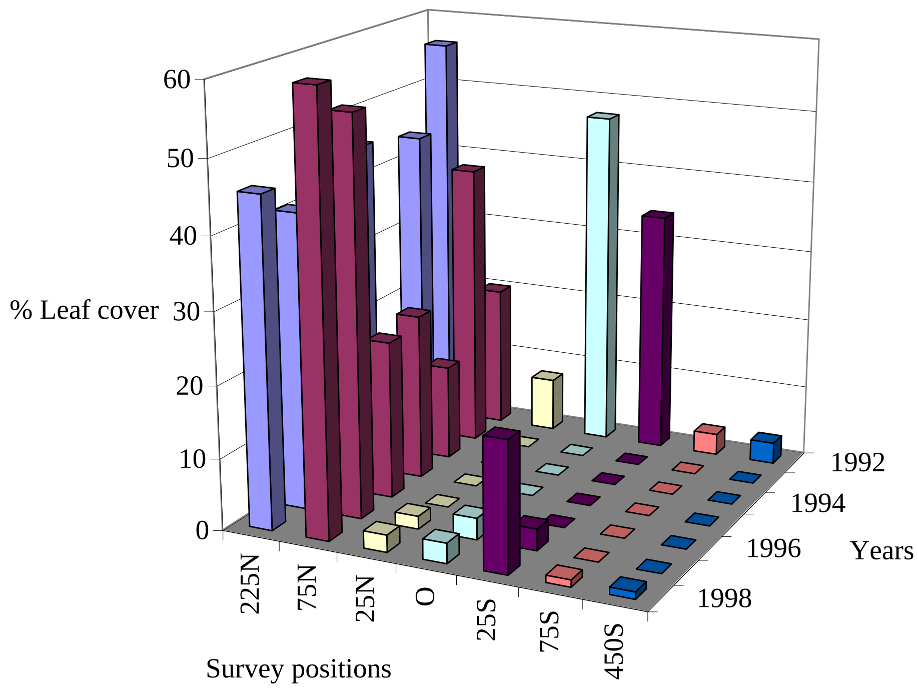


**Figure 10:** Mean % leaf cover of *Zostera* at 7 sites along a transect; Westfield, North Morecambe

# 7. Confidence in the assessment

The outcomes of the tools being developed will govern the action, if any, to be taken on a particular area, i.e. whether formal programmes of measures are required to improve the situation. Sampling strategy and frequency, data collection and interpretation should provide high levels of confidence,as must the tool's ability to classify accurately the ecological status of a water body.

## 7.1 Confidence in data - Sampling strategy and frequency

The monitoring of seagrass beds takes place over the full extent of beds within a waterbody, consistent with health and safety considerations. Ideally sampling should take place during the time of seagrass peak growth, which is likely to fall between June and September (inclusive) for most parts of the UK. There may be some temporal variation geographically or annually depending on climatic conditions. In subsequent years surveys should take place at approximately the same date (Foden and Brazier, 2007). Historical data may be used to establish the time of peak growth. Confidence in sampling can be increased by improving the frequency, however, monitoring such extensive areas is not always possible with limited resources. The application of a rolling mean can help with confidence although this requires annual sampling. The frequency of sampling should, ideally, be annual due to the levels of natural variability, but resources may not permit this. Longer data sets allow the examination of trends, which may be informative, and will increase confidence in assessments. Confidence decreases with low sampling frequency over a WFD reporting cycle; this could be summarised as: high confidence = 6 consecutive years of data, medium confidence = 3 years data within a cycle and low confidence = 2 years data within a cycle.

Transects or a random stratified sampling strategy may be used. The number of quadrats necessary will be proportional to the size and variability of seagrass beds, and resources. Patchier beds may require a larger number of quadrats to encompass the variation in % cover.

## 7.2 Confidence in Data – Quality control

The confidence here lies with the field surveyors' ability to collect all the required data to specified levels of accuracy. This is something to be dealt with both internally within each organisation, and externally as part of a proficiency testing scheme. Organisations will use their own audit programmes, and may also belong to an accreditation scheme. External proficiency testing is encouraged, e.g. the U.K. National Marine Biological Analytical Quality Control (NMBAQC) runs ring tests requiring estimations of % cover. These processes also help to identify areas where additional training is necessary.

## 7.3 Confidence of Classification

Providing an estimate of the statistical uncertainty of water body assessments is a statutory requirement of the WFD (Annex V, 1.3). In an ideal world of comprehensive monitoring data containing no errors, water bodies would always be assigned to their true class with 100% confidence. However, estimates of the truth based on monitoring are subject to error because monitoring is not done everywhere and all the time, and because monitoring systems, equipment and people are less than perfect. Understanding and managing the risk of misclassification as a result of uncertainties in the results of monitoring is important on two counts; first, because of the potential to fail to act in cases where a water body has been wrongly classified as being of better status than it is, and secondly because of the risk of wasting resources on water bodies that have been wrongly classified as worse than they are.

Like other biological quality elements, it is not always possible to survey seagrass communities across a whole water body continuously throughout the whole reporting period. Additionally there will always be some sampling error, which will lead to some uncertainty in the estimate of the EQR. This uncertainty can be quantified as the expected difference between the observed EQR and the true underlying EQR, which can then be used to calculate the probability of the water body being in each of the five status classes. From this it is possible to determine the most probable class and to estimate the risk of mis-classification.

An approach to assessing the precision of the results, Seagrass Assessment Incorporating Likelihood of Risk (SAILOR), is being developed by WRC (Davey, in draft).

SAILOR works in a similar way to the other CoC tools, however special consideration has to be given to taxonomic composition. Uncertainty in the EQR for this metric could arise from error in assessing which species are present; a species may either go undetected (a false negative) or mis-identification may lead to the mistaken belief that a species is present when it is not (a false positive). It is thought that for seagrass the taxonomic composition metric is more likely to be under-estimated than over-estimated.

For a water body in which the reference condition is three species, if the probability of a false positive is assumed to be 0% for each species and the probability of a false negative is assumed 10% for each species then:

> if the sampling identifies two species, then there is 90% confidence that status is Good (i.e. 33% species loss) and 10% confidence that the third species was accidentally missed and therefore that status is High.
>
> if the sampling identifies only one species, then there is 81% confidence that status is Poor (i.e. 66% species loss), 18% confidence that one species was accidentally missed and therefore that status is Good, and 1% confidence that two species were missed and that status is High.

It is assumed that the reference condition is known without error.

There is no way to reliably estimate a standard error for the metric EQR as it can take just one of five possible EQR values. An approximate standard error can be estimated, however, by calculating a weighted mean and standard deviation using the confidence of class results. Continuing the above example, if the confidence of class assessment gives 81% confidence of Poor (EQR = 0.3), 18% confidence of Good (EQR = 0.7) and 1% confidence of High (EQR = 0.9), then the weighted EQR result is:

Metric EQR = (0.81 * 0.3) + (0.18 * 0.7) + (0.01 * 0.9) = 0.378

and the associated standard error is:

$SE = SQRT\ (0.81 * (0.3 - 0.378)^2 + 0.18 * (0.7 - 0.378)^2 + 0.01 * (0.9 - 0.378)^2\ ) = 0.162$

# 8. European Intercalibration

The main aims of the European intercalibration exercise are to establish class boundary values for high-good and good-moderate status, which must in turn be consistent with the normative definitions for those class boundaries, and to ensure the various Members States (MS) are making equivalent assessments. The former is achieved by monitoring the degree of deviation from reference conditions by use of monitoring tools currently developed by the member states and incorporating various parameters, methodologies and assessment measures. The intercalibration process entails discussions between member states to agree a common means of assessment incorporating all biological quality elements, all waterbody types and all pressures.

The intercalibration process deals with the development of reference conditions, and the setting of specific class boundaries for those metrics of the biological quality element angiosperm for which suitable assessment methods and comparable data are available within the NEA GIG areas. A number of quality assessment tools have been developed by different member states incorporating various aspects of angiosperm density, taxonomic composition and depth distribution and limits. Discussions have been held throughout Europe to review the feasibility of such tools and how they may be adapted to include the variable habitat types and environmental factors experienced across the member states.

Guidance on the intercalibration process is developing over time. Phase 1 is discussed in the following sections, but Phase 2 of Intercalibration is not as yet complete at time of writing.

## 8.1 IC Phase One - National Methods Under Intercalibration

The national methods that have been proposed for intercalibration include changes in taxonomic composition, shot density and bed extent. However, not all members states have equivalent data and there are still a number of disparities between datasets. Therefore it is important to note that:

- Member states have different seagrass species and different numbers of species. Member states are considering standardising their percentage loss descriptors.

- The Netherlands considers NEA waterbody types 1 and 26 separately.
- Spain might only be able to intercalibrate on taxonomic composition (from the three metrics proposed) because the Spanish area and density (% cover) metrics combine actual data (for seagrass, saltmarsh and macroalgae) not potential or historical data.
- German seagrass data are derived from aerial monitoring and ground truth investigations. At present no differentiated data are available concerning species composition and coverage/density (per species). Accordingly only the bed extent of intertidal seagrass is ready to be classified. Future monitoring programs will help to close this gap.

However, the Netherlands, UK and Ireland have been able to fully intercalibrate using the desired parameters and this work has already been published by Foden (2007), summarised details of which are documented below. In June 2007 Portugal was still considering intercalibration, but progress is yet to be made.

Seagrass abundance and taxonomic composition is only being fully intercalibrated between the Netherlands, Ireland, and UK. Germany currently only has bed extent data and is currently only intercalibrating on Metric 2 (Seagrass Abundance: acreage/bed extent), however when species data is available Germany should be able to participate in all the metrics. Spain have developed a separate metric which has not yet been intercalibrated, this will be completed Phase II of Intercalibration. This metric has been declared as not applicable by the other GIG Member States.

The reference conditions for seagrass for each of the chosen sub-metrics in the co-operating countries are defined as below in Table 12. The assumption is made that these occur in unimpacted areas with unpolluted water quality and no hydromorphological alterations to the shore or seabed. Dutch waterbodies are embanked and may be classed as heavily modified. Although the waterbodies are managed and protected by engineering works, habitats such as seagrass beds have established naturally within them. Potential Reference Conditions (P-REF) and Potential Good Ecological Status (P-GES) are the highest two classes heavily modified waterbodies can attain, and scientists in the Netherlands have set values for these by focusing on the current situation in the waterbodies concerned (de Jong, 2004).

**Table 13:** Summary of UK and NL seagrass metrics and boundary conditions for ecological status classes (Foden and Brazier, 2007; de Jong, 2004).

| *Metric* | *Country Parameter* | | *Ecological Status Class* | | | | |
|---|---|---|---|---|---|---|---|
| | | | High (UK)<br>P-REF (NL) | Good (UK)<br>P-GES (NL) | Moderate | Poor | Bad |
| Taxonomic composition | UK | Seagrass species | No loss of species | Loss of ¼ to 1/3 sp. | Loss of ½ sp. | Loss of 2/3 -3/4 sp. | Loss of all sp. |
| | NL | *Z. angustifolia* and *Z. noltei* | 2 spp. | 1spp. | - | - | - |
| Bed extent | UK | Area of seagrass bed in water body | 0 -10% below ref. conditions | 11 -30% below ref. conditions | 31 - 50% below ref. conditions | 51 - 70% below ref. conditions | >70% below ref. conditions |
| | NL | Wadden sea | 250 ha | 150 ha | <25% below P-GES | 25 – 50% below P-GES | >50% below P-GES |
| | | Oosterschelde | 1000 ha | 750 ha | | | |
| | | Ems-Dollard | 100 ha | 50 ha | | | |
| | | Westerschelde | 3 ha | 2 ha | | | |
| Bed density/ coverage | UK | Density of all species in water body | 0 -10% below ref. conditions | 11 -30% below ref. conditions | 31 - 50% below ref. conditions | 51 - 70% below ref. conditions | >70% below ref. conditions |
| | NL | *Z. angustifolia* | ≥ 30% | ≥ 20% | ≥ 10% | ≥ 5% | < 5% |
| | | *Z. noltei* | ≥ 60% | ≥ 40% | ≥ 30% | ≥ 20% | < 20% |

Classification status for density is determined by the underlying trend over a period of 5-6 years, where data exist, to coincide with the WFD's reporting cycle. The trend for an individual bed and the loss or gain, as compared with a maximum recorded density, can be used to identify whether the seagrass bed is in a state of degradation or recovery.

### *8.2.1 Boundary Criteria*

**Species Composition**

Most seagrass beds in the UK will comprise of 1 or 2 species. Consequently, the NL and UK metrics are similar. The main difference is that no distinction can be made between High/Ref and Good for the UK metric because, for example, there are sublittoral beds of *Z. marina* that are naturally mono-specific and are at High status. The NL's metric is not able to define conditions less than good ecological status (GES), whereas the UK metric has boundaries between Good and Moderate, and between Moderate and Poor/Bad. As the tool testing examples show (below) in most cases the outcomes of the NL metric and UK metric are generally the same.

**Seagrass Abundance: acreage/bed extent**

There are significant similarities between the NL and UK metric boundary conditions between each ecological status class for seagrass acreage/bed extent. With only four waterbodies the NL have been able to use modeling and expert judgement to set precise bed areas for REF and GES for each of those waterbodies. The average difference between REF and GES is ~30% which is broadly in line with the UK's more generalised boundary of a 30% decrease in bed extent between High/Ref and Good. The mean difference between the NL's Moderate and REF for all four waterbodies is ~50%, between Poor and REF is ~70-75% and between Bad and REF is >70%. All of these boundaries are broadly in common with the UK/IE and Germany metric's boundary conditions.

**Seagrass Abundance: coverage/density**

As with bed extent, there are significant similarities between the NL and UK metric boundary conditions between each ecological status class for seagrass coverage/density. With only four waterbodies the NL have been able to use modeling and expert judgement to set precise density ranges for *Z. noltei* and *Z. angustifolia*, for REF and GES. The difference between NL's REF and GES for both species is ~30% which is broadly in line with the UK boundary of a 30% difference between High/Ref and Good. For *Z. noltei* the difference between The NL's Moderate and REF is 50%, which corresponds with the 50% difference between Moderate and High for the UK metric. For *Z. angustifolia* there is a greater difference between The NL's Moderate and REF (⅔) than between Moderate and High for the UK metric. However, only the Ems-Dollard waterbody will be assessed against this criterion because the other 3 waterbodies either comprise solely of *Z. noltei* or *Z. noltei* is the dominant species present.

There is a difference of ⅔ between NL's Poor and REF and >⅔ between Bad and REF, for *Z. noltei*. These boundaries are broadly in common with the UK metric's boundary conditions of <70% loss of seagrass for Poor and >70% loss for Bad.

**Sub-metric to support trends in seagrass abundance**

Both NL and UK agree that the underlying trend in seagrass abundance should show a stable seagrass bed (at the maximum potential identified for that site/waterbody). If abundance is less than would be expected for High/Reference conditions then abundance should show a positive underlying trend, indicative of recovery. Conversely, a negative trend in seagrass abundance is undesirable, indicative of degradation, and would signal a potential deterioration in ecological class.

Furthermore, the Member States agree that the ideal period over which to consider the trend in abundance is ~6 year, designed to coincide with the WFD reporting cycles.

### *8.2.2 Testing UK Waterbody Data against NL and UK Metrics*

Two areas with data have been used for the comparison; Strangford Lough, UK (Portig, 2004) and Fleet Lagoon.

The outcomes for the NL and UK metrics are the same in all three instances (Table 13) and for both sample areas. The lack of raw data makes precise statements regarding seagrass acreage (bed extent) and coverage (density) difficult. The underlying trend in seagrass density is positive over the 4 survey years, confirming the judgement of the seagrass being in a 'recovery' phase.

**Table 13:** Current situation of seagrass in (a) Strangford Lough (Portig, 2004) and (b) Fleet lagoon (Bunker et al., 2004), and outcomes of testing UK and NL metrics

| *Area* | *Metric* | | |
|---|---|---|---|
| | Species composition | Area / bed extent | Coverage / bed extent |
| **Strangford lough** | | | |
| Results | *Z. angustifolia, Z. noltei, Z. marina* and *Ruppia* spp. | 924 ha | 53% mean density for all species |
| NL outcome | P-REF | Moderate | P-REF |
| UK outcome | High | Moderate | High |
| **Fleet Lagoon** | | | |
| Results | *Z. noltei, Z. marina* and *Ruppia* spp. | Some loss of recorded species. Limited data; assume 50-70% of maximum potential | Broadly unchanged. Statistical comparison limited |
| NL outcome | P-REF | Moderate | P-GES |
| UK outcome | High | Moderate | Good |

Historical data are scarce, but there has clearly been a marked decline in the distribution of seagrasses in Northern Ireland since 1930s. This has been coupled with a change in the dominant *Zostera* spp. present in the intertidal areas with *Z. marina* in its perennial form dominant in the 1930s being replaced by *Z. noltei* and *Z. angustifolia* by 1970. In addition there has been a general improvement in the status of *Zostera* spp. in the northern end of Strangford Lough during the last 10 years (Figure 11). The necessary data are lacking, however, to determine whether these changes are part of ongoing cyclical processes or longer term changes.

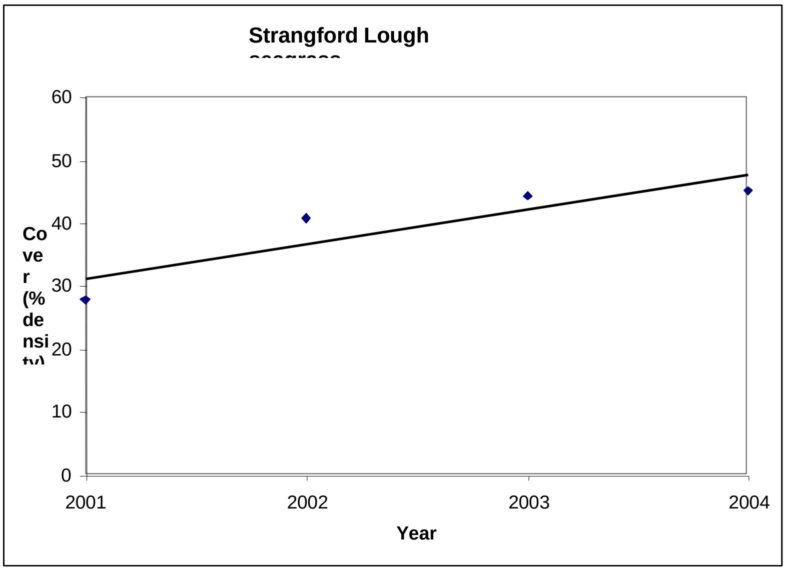



**Figure 11:** Temporal trends in Strangford Lough seagrass % density

### *8.2.3 Testing Dutch Waterbody Data against NL and UK Metrics*

A similar comparison has been achieved using data from 4 areas within the Netherlands; the Wadden sea, Oosterschelde, Ems-Dollard and Westerschelde. The results are found in Table 10 below.

All the outcomes for the NL and UK metrics are broadly comparable (Table 14). The only difference is for species composition where only one species present result in P-GES for the NL metric whereas since there is no loss of species the UK classification is High.

**Table 14:** P-REF and PGES for four Dutch water bodies (de Jong, 2004) and the outcomes of testing UK and NL metrics on data for; (a) Wadden Sea, (b) Oosterschelde, (c) Ems-Dollard and (d) Westerschelde.

| *Area* | *Metric* | | |
|---|---|---|---|
| | Species composition | Area / bed extent | Coverage / bed extent |
| **Wadden Sea** | | | |
| Results | 2 species | 47 ha | *Z. noltei* ≈ 40% |
| NL outcome | P-REF | Bad | P-GES |
| UK outcome | High | Bad | Good |
| **Oosterschelde** | | | |
| Results | 2 species | 94 ha | *Z. noltei* ≈ 62% |
| NL outcome | P-REF | Bad | P-REF |
| UK outcome | High | Bad | High |
| **Ems -Dollard** | | | |
| Results | 1 Species | 14 ha | *Z. angustifolia* ≈ 13% |
| NL outcome | P-GES | Bad | Moderate |
| UK outcome | High | Bad | Moderate |
| **Westerschelde** | | | |
| Results | 1 Species | 2 ha | *Z. noltei* ≈ 5 – 20% |
| NL outcome | P-GES | P-GES | Poor (possibly bad) |
| UK outcome | High | Good | Moderate to Poor |

In summary 18 metric tests were carried out on the metrics of NL and UK with an overall agreement for 15 metric tests, which is compliance of >83%. Where NL outcomes differ from UK metrics these differences are explicable. For Ems-Dollard the species composition metric outcomes were High for UK and P-GES for NL, reflecting national differences in the way reference conditions for species composition are set. The outcomes for species composition metric are also different for Westerschelde, but it is the lack of historic data to confirm Z. noltei as having been historically present that results in a class of High for theUKmetric. The third difference in outcome between NL and UK is for the density metric in Westerschelde. This difference is likely to be of minor significance for two reasons; raw data are not available to allow a more precise setting of class under the UK metric, and both nationalities' outcomes are less than Good, meaning a programme of investigative measures would be undertaken. All metric tests on other water bodies showed parity in their outcomes for UK and NL.

The temporal trends in abundance data are useful supporting metrics and can act as an early warning system where deterioration in ecological quality may be starting to happen. Examination of trend data is not a specific parameter to be assessed under the Directive but it is recommended by both MSs. Importantly where there are differences between the nationalities' metric outcomes, the classifications still fall on the same side of the Good-Moderate boundary. This boundary is significant because water bodies falling below Good status will be subject to further monitoring to identify the causes ((WFD) Directive 2000/60/EC) and remedial action must be taken to improve ecological quality. This may be a costly process. MSs individually developed national metrics for angiosperm ecological assessment, and yet there is strong agreement between nationalities' metrics.

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