

# eDeposit Ireland

## Management considerations for subtidal *Zostera marina* beds in Ireland

Authors	Dale, A.L.;McAllen, R.;Whelan, P.
Publisher	National Parks and Wildlife Service
Rights	Y
Download date	2026-03-14 19:37:27
Link to Item	<a href="https://hdl.handle.net/20.500.14765/69408">https://hdl.handle.net/20.500.14765/69408</a>

# Management considerations for subtidal *Zostera marina* beds in Ireland



Irish Wildlife Manuals No. 28



AN ROINN COMHSHAOIL, OIÐHREACHTA AGUS RIALTAIS ÁITIÚIL

DEPARTMENT OF THE ENVIRONMENT, HERITAGE  
AND LOCAL GOVERNMENT



# Management considerations for subtidal *Zostera marina* beds in Ireland

Dale, A. L., McAllen, R. & Whelan, P.

Department of Zoology, Ecology & Plant Science  
University College Cork

**Citation:**

Dale, A. L., McAllen, R. & Whelan, P. (2007) Management considerations for subtidal *Zostera marina* beds in Ireland. *Irish Wildlife Manuals*, No. 28. National Parks and Wildlife Service, Department of Environment, Heritage and Local Government, Dublin, Ireland.

**Cover photo:** *Zostera marina* in Barloge Creek, Lough Hyne Marine Nature Reserve, Co. Cork. (©Amy Dale)

Irish Wildlife Manuals Series Editors: N. Kingston & F. Marnell  
© National Parks and Wildlife Service 2007

ISSN 1393-6670

## ACKNOWLEDGEMENTS

This report was commissioned by the National Parks and Wildlife Service and is based on studies carried out for a Ph.D studentship funded by the Glucksman Fund (University College Cork). The authors are grateful to M. Jessopp and R. Thomas (University College Cork) for fieldwork assistance and M. Holmes (Natural History Museum, Dublin) for help with crustacean identification. The authors would also like to thank D. O'Donnell and P. Graham from the National Parks and Wildlife Service (Department of Environment, Heritage and Local Government) for granting Lough Hyne research permits and E. Kelly for editorial assistance.

## CONTENTS

	Page
EXECUTIVE SUMMARY	6
1. INTRODUCTION	7
2. MATERIALS & METHODS	10
Environmental Parameters	10
Physical Structure of Plant Communities	11
Associated Faunal Diversity	14
3. RESULTS	15
Environmental Parameters	15
Physical Structure of Plant Communities	18
Associated Faunal Diversity	23
4. DISCUSSION	25
Management Implications	30
5. CONCLUSIONS	32
6. REFERENCES	33
7. APPENDICES	36
Infaunal Species List	36
Epifaunal Species List	39

## EXECUTIVE SUMMARY

Seagrasses are a worldwide group of marine flowering plants that act as an important site for maintaining faunal marine biodiversity. They do this by acting as a foraging and refuge habitat, providing a nursery and spawning ground for many species and aiding sediment stabilisation (and hence coastal protection) due to their compact root-rhizome matrix. However, limited research has been undertaken on seagrasses in Ireland to date. *Zostera* is the dominant seagrass genus around Ireland with two species generally recognised, the subtidal and intertidal *Zostera marina* and the intertidal *Zostera noltii*. The present study investigated the dynamic nature of four subtidal *Z. marina* beds at two sites in south-west Ireland; at Barloge Creek in Co. Cork and at Ventry Bay in Co. Kerry. Studies examined environmental variation in the seagrass beds, the physical structure of the plant communities and their associated infaunal diversity over a two-year period.

Seawater temperature and salinity showed patterns typical of fully marine habitats, whilst there was a significant difference in grain size composition between the two sites. This subsequently influenced the organic carbon data within the sediments with Barloge Creek showing a higher content due to its higher mud/silt composition. *Zostera* shoot density, blade measurements and root-rhizome biomass showed large fluctuations between months, depths and sites with a broad seasonal pattern evident in unison with seasonal environmental change (e.g. temperature and light). In addition, infaunal community diversity also showed large fluctuations between months, depths and sites.

The inherent natural variability observed in the *Zostera* beds in this study has implications for the management of this habitat type and its potential use as an indicator of habitat quality. It is evidently clear that any seasonal patterns are difficult to detect without regular monitoring of the beds, and the need for extensive baseline data sets (two years minimum) before “spot-monitoring” occurs is vital. However, it should also be noted that numerical abundances have been used in this study, which has led to this conclusion of high natural variability in both the seagrass itself and the associated fauna. This may not be the most appropriate measure of the beds from a monitoring perspective, and the use of much broader DAFOR scales, for example, may be more suitable for long term monitoring of bed status and hence quality of the *Zostera* beds in Ireland.

## 1. INTRODUCTION

Seagrasses are a small group of angiosperms found in the shallow waters of all continents except Antarctica. They encompass 12 genera containing approximately 50 species although the actual number of species is continuously under debate depending on the level of discrimination in physical taxonomy, genetics and the species proximity to the marine environment (Spalding *et al.* 2003). Seagrasses reproduce sexually and asexually through detached rhizome fragments. Nine of the 12 genera are dioecious (separate female and male clones) which is relatively rare in angiosperms. Flowering is controlled by temperature with temperate genera, such as the monoecious Genus *Zostera*, generally flowering in late spring concurrently with increasing irradiance and water temperature (Hemminga and Duarte 2000).

*Zostera* (commonly known as eelgrass), although considered to be temperate, is very widespread and so also occurs in tropical climates (Hemminga and Duarte 2000) (Fig. 1). Its taxonomy under the Angiosperm Phylogeny Group system (APG 1998) is as follows:

Clade - Monocots

Order – Alismatales

Family – Zosteraceae (Eelgrass family)

Genera – *Zostera* (Eelgrass)

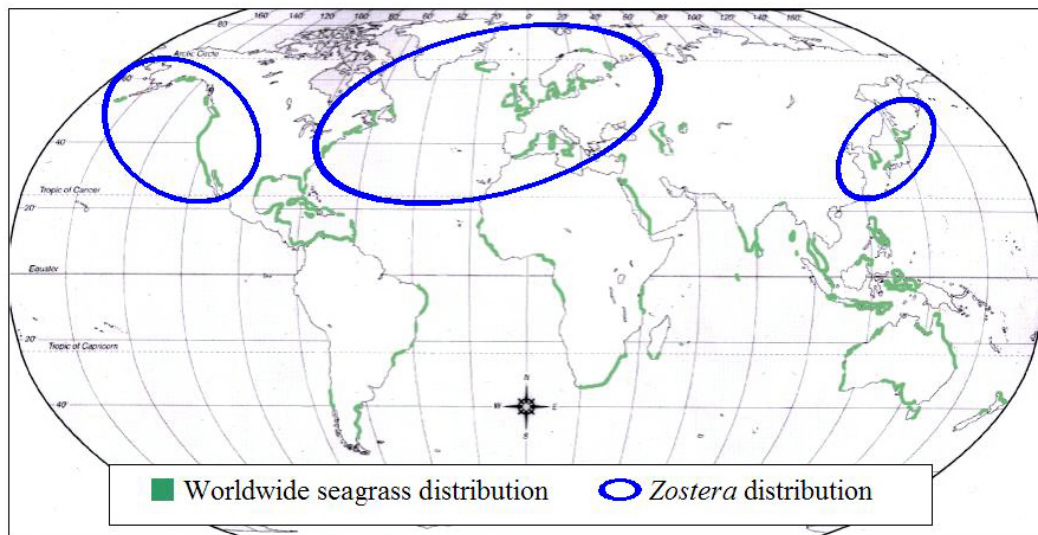


Figure 1. Worldwide distribution of seagrass (modified from Rasmussen 1977 and Spalding *et al.* 2003).

There are two internationally recognized species of seagrass in Ireland, *Zostera marina* (Common Eelgrass) and *Z. noltii* (Dwarf Eelgrass; syn. *Z. noltei*). *Z. marina* forms dense beds, with trailing leaves up to 1 m long, in sheltered bays and lagoons from the lower shore to about 4 m depth (however it can occur much deeper), typically on sand and sandy mud (occasionally with a mixture of gravel) (Fig.

2a). *Z. noltii* forms dense beds, with leaves up to 20 cm long, typically in the intertidal region (although it can occur in the very shallow subtidal), on mud/sand mixtures of varying consistency (OSPAR Commission 2005) (Fig. 2b). *Z. angustifolia* has traditionally been regarded as a third species by British and Irish authorities as it differs from *Z. marina* in morphology, reproductive strategy and habitat use (Robinson 2003). However, these characteristics are known to vary depending on habitat and season, and results from DNA sequencing supports the hypothesis that *Z. marina* and *Z. angustifolia* are variants of a single species (Davison and Hughes 1998).

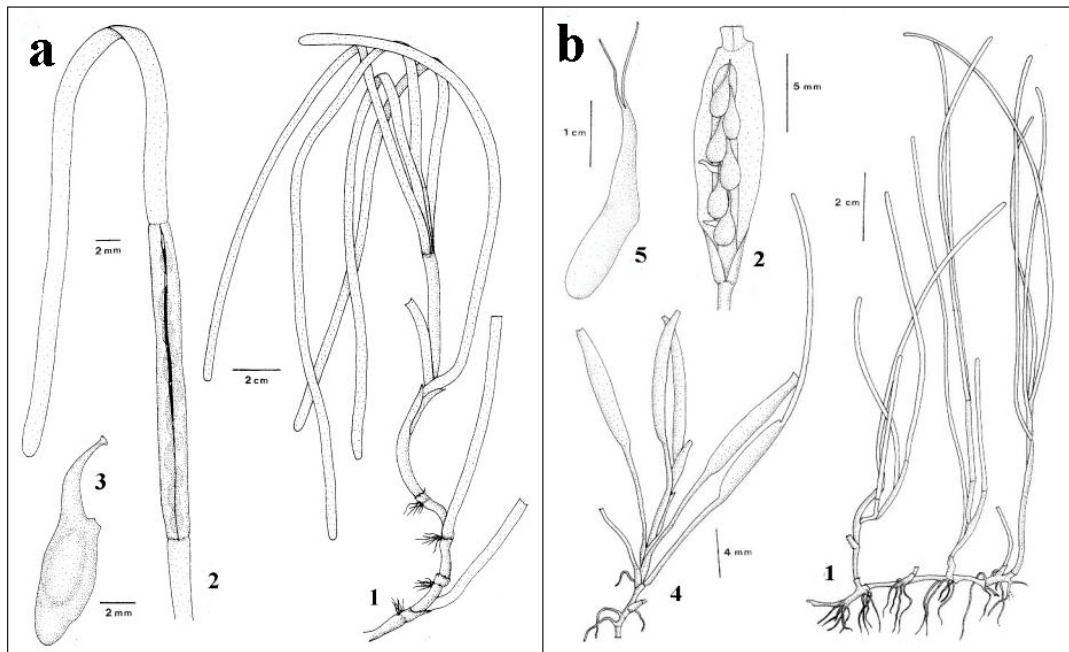


Figure 2. Morphological comparison of *Zostera marina* (a) and *Zostera noltii* (b). Full plant (1), spadix (2), fruit (3), fertile plant with spadices (4), and female flower (5) (from Phillips and Meñez 1988).

Seagrasses have four main ecological requirements; (1) sufficient immersion in seawater to avoid osmotic stress, (2) a sheltered environment to protect their root system, (3) sandy to muddy nutrient rich substrata to stimulate plant growth, and (4) sufficient illumination to maintain growth. Habitats hosting seagrass communities are known to provide important ecological services through their role in maintaining fisheries and biodiversity by:

- providing a foraging and refuge habitat,
- acting as a nursery and spawning site for many species, including commercially important fishery species,
- providing a source of nutrients for mobile non-resident fauna such as the internationally important Light-bellied Brent Geese (*Branta bernicla hrota*) and Wigeon (*Anas penelope*) (Kikuchi 1980; Robinson 2003) as well as many sessile detritivores, and
- their compact root matrix facilitates coastal protection, sediment stabilisation, and nutrient cycling (Hemminga and Duarte 2000).

In the early 1930s there was a severe natural decline in *Zostera marina* vegetation along the North Atlantic coasts. This decline was shown to be due to an infection (“wasting disease”) that resulted in the destruction of leaf and rhizome tissue and eventually led to death (Rasmussen 1977). Early hypotheses that the disease was caused by fungus and slime mould remain unproven. By the early 1950s many areas affected by the 1930s destruction had recovered although there were several localities to which *Z. marina* did not return. The main causes of decline in recent decades are anthropogenic. Land reclamation, coastal development, boating and fishing activity, sewage discharge and agricultural run-off often result in physical disturbance and siltation as well as increased water turbidity and nutrient loading (Spalding *et al.* 2003). Most seagrass species are not robust and can be easily dislodged as their root-rhizome matrix is usually situated within the top 20 cm of the sediment (Fonseca 1992). Extended seagrass beds with good penetration to deep waters are characteristic of coastal waters with minimal anthropogenic impact (Borum *et al.* 2004).

*Zostera* beds are recognised as a characteristic component of five Annex I habitats in the EU Habitats Directive (92/43/EEC). In addition, in 2004, OSPAR produced descriptions of habitats on the Initial List of OSPAR Threatened and/or Declining Species and Habitats to act as a working definition for the collection of information on distribution and abundance (OSPAR Commission 2005). This outlined 14 habitat types considered to be causes for concern, and included *Zostera* seagrass beds.

There has been little research into *Zostera* habitats in Ireland to date. Robinson (2003) reviewed the status and distribution of intertidal *Zostera* beds in Ireland, while some studies in Northern Ireland concentrated on the distribution of intertidal *Zostera* occurring in Strangford Lough (Anon. 1995). Detailed research of the fully subtidal species *Z. marina* in the Republic of Ireland has been limited to a comprehensive study of its distribution for the whole of Ireland, which includes density biomass in Co. Cork and Co. Kerry and variation with depth in Co. Kerry (Whelan 1986), and studies of *Z. marina*-associated algal communities (e.g. Cullinane *et al.* 1985; Whelan and Cullinane 1985). To date there have been no comprehensive studies on subtidal faunal communities.

The present study was undertaken to investigate natural dynamic trends in *Zostera marina* beds and their associated fauna at two sites in Ireland. Specifically, the study undertook to:

1. provide an environmental description of two *Z. marina* communities in the south-west of Ireland,
2. compare and contrast the spatial and temporal variations in the physical structure of the plant communities, and
3. investigate features of the associated faunal diversity.

The implications of these results for the conservation and management of seagrass communities in Ireland was explored.

## 2. MATERIALS & METHODS

Sampling of *Zostera marina* beds was conducted in 2004 and 2005 in Barloge Creek, Co. Cork and in Ventry Bay, Co. Kerry in the south-west of Ireland (Fig. 3). Lough Hyne is a fully marine, south-facing, lough connected to the open sea by a narrow, sheltered channel called Barloge Creek. The adjacent coastline is exposed, rocky shore. Sampling at Barloge Creek focussed on a bed approximately 200 m long and 50 m wide, with a maximum depth of 4 m (51°29.985'N, 9°17.742'W). Ventry Bay is a south-facing, sheltered embayment approximately 2.5 x 1.5 km with a wide opening into Dingle Bay. The adjacent coastline consists of exposed rocky shore with some sandy inlets. *Z. marina* is extensively distributed throughout the bay. Sampling at this location focussed on a shallow site 4 m deep (52°06.725'N, 010°21.576'W), an intermediate site 7 m deep (52°06.843'N, 010°21.503'W) and a deep site 10 m deep (52°07.013'N, 010°21.421'W). All sampling was conducted using SCUBA techniques.

### Environmental Parameters

Bottom water temperature at substrate level was recorded using a Suunto Mosquito dive computer at each Ventry Bay site after completion of sample collection (i.e. every four months between April 2004 and December 2005). Bottom water temperature at Barloge Creek was recorded using an Onset Computer Corporation Optic Temperature Logger anchored 10 cm above the substrate and downloaded every three months between March 2004 and December 2005. Water samples were collected monthly for one year at Barloge Creek (except April 2005) and in December 2004, and April and August 2005 at each of the Ventry Bay sites, and used to measure water salinity immediately upon return to the surface.

Grain-size and organic carbon content of sediment was obtained by taking sediment samples to a depth of 120 mm using a 105 mm wide cylindrical core. Four replicate samples were taken at each sampling month. At Barloge Creek cores were taken in August, October and December of 2004 and in February, April, August and December of 2005 at randomly selected points along a permanent transect that ran north to south through the centre of the *Zostera* bed to remove any possible edge effects. At Ventry Bay cores were taken in August and December of 2004 and April, August and December of 2005 at random locations positioned >1 m apart and >1 m from the edge of the bed at each site (no samples were taken at the intermediate or deep sites at Ventry Bay in December 2005 due to poor weather conditions). From each core approximately 30 g of sediment was dried at 60 °C for 24 hours for organic content analysis, while the rest was kept at 4 °C for grain-size analysis. Percentage organic carbon content was calculated from the weight-change following combustion of 1 g of the sub-sample at 450 °C for 4 hours. The remainder of the core was rinsed in freshwater as soon as possible to

remove salt before drying at 60 °C for 48 hours, and sieved to establish gravel, sand and silt-clay fractions using the Wentworth scale (Buchanan and Kain 1971).

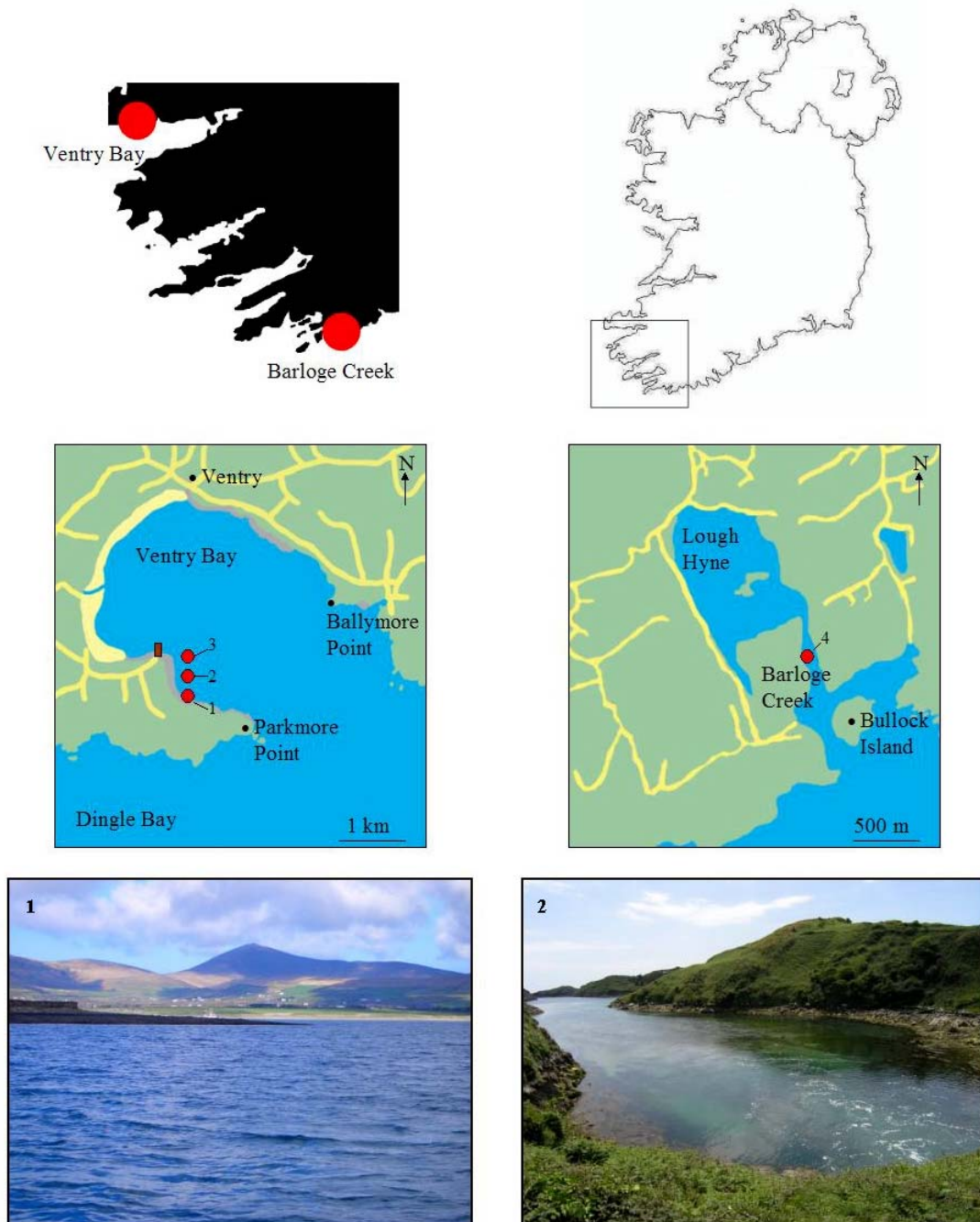


Figure 3. Location of study sites. Sampling areas: shallow (1), intermediate (2), and deep (3) sites at Ventry Bay, Co. Kerry (photograph 1), and the Barloge Creek site (4), Co. Cork (photograph 2).

### **Physical Structure of Plant Communities**

Shoot density was determined by counting the number of shoots in a quadrat (0.5 m<sup>2</sup>) and then multiplying the data to give density per m<sup>2</sup>. Shoot counts at Barloge Creek were carried out every month between April 2004 and August 2005, and once in December 2005. No counts were conducted

in May 2004 due to poor weather conditions. Twelve replicate counts per sampling month were taken, all within 1 m of a permanent transect that ran north to south through the centre of the *Zostera* bed, again to negate any possible edge effects. Shoot counts at Ventry Bay were carried out every 4 months for 2 years between April 2004 and December 2005, with 4 replicate counts per site per sampling month. No counts were taken at the intermediate or deep sites at Ventry Bay in December 2005 due to poor weather conditions. Quadrats were placed randomly in the vicinity of sediment cores, ensuring they were located at >1 m from the edge of the bed. Four replicate shoots per sampling month were collected at a location close to sediment cores at each site. Shoots were collected with rhizome intact by digging below the sediment surface to expose the rhizome mat, and placed in individually labelled bags. Upon return to the laboratory, each shoot was cut from its attached rhizome (start of the rhizome was indicated by the presence of a small protrusion from which roots protrude) (Fig. 4a). Blades were separated and the total length of the longest blade in each shoot noted (Fig. 4b). Rhizome biomass data were collected by removing root-rhizome material from all sediment cores (cores primarily collected for sediment grain-size and organic carbon content or infaunal composition) and drying at 60 °C for 24 hours. All data were analysed with one-way analysis of variance (ANOVA).

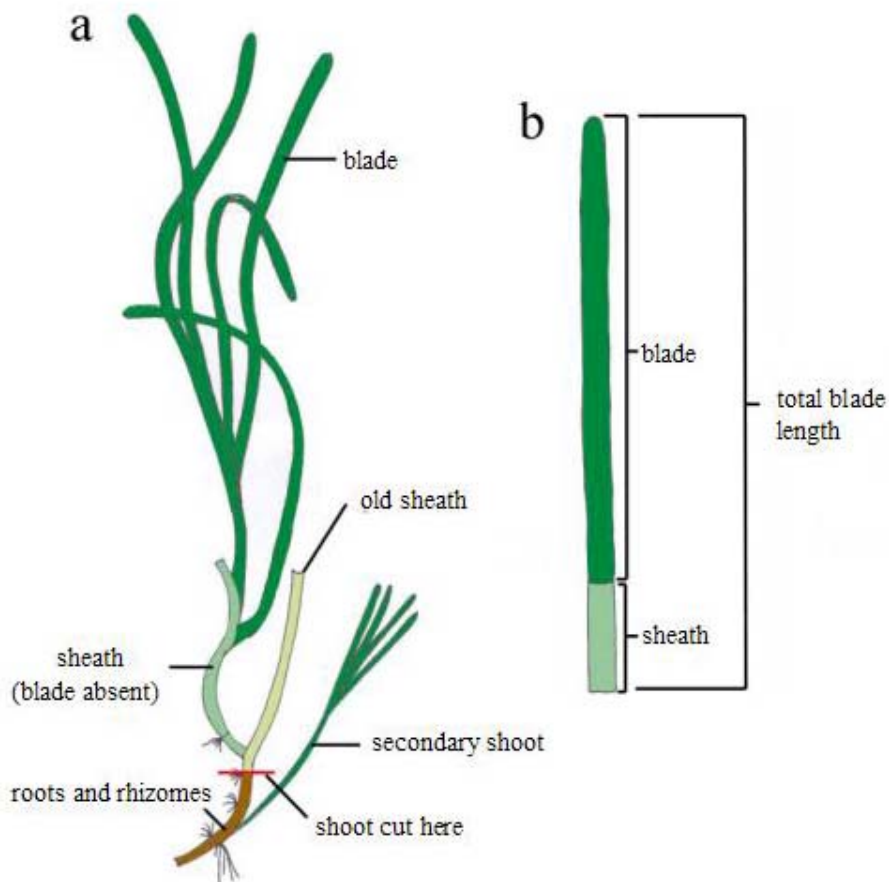


Figure 4. Diagrammatic representation of *Zostera marina*. (a) main shoot with associated secondary shoot and root-rhizome matrix (modified from Phillips and Meñez 1988), and (b) single blade showing points of measurement for total length of blade.

During preliminary observations of the Barloge Creek *Zostera* bed, it was noted that there was a small independent patch of seagrass at the northern end of Barloge Creek, separated from the main bed by bare sediment. It was hypothesised that the bare sediment between the two areas of seagrass was a result of seasonal dieback of the canopy and that the two beds would rejoin during periods of peak growth. In order to measure the expansion of the seagrass bed into an adjacent area of bare sediment, a permanent 30 m transect was anchored to the seabed running SE to NW (and hence running parallel to each of the seagrass bed boundaries and through the centre of the bare area of sediment). The distance between the permanent transect and seagrass was measured at 2 m intervals along the transect by running a tape measure out at a perpendicular direction from the transect to the seagrass bed (Fig. 5). The distance along the tape measure when bare substrate gave way to seagrass was recorded. Tape measures were extended for 20 m so that small patches of seagrass and bare patches in the beds could also be recorded. This procedure was repeated on each side of the transect. Surveys were carried out monthly between June 2004 and May 2005 and in August 2005.

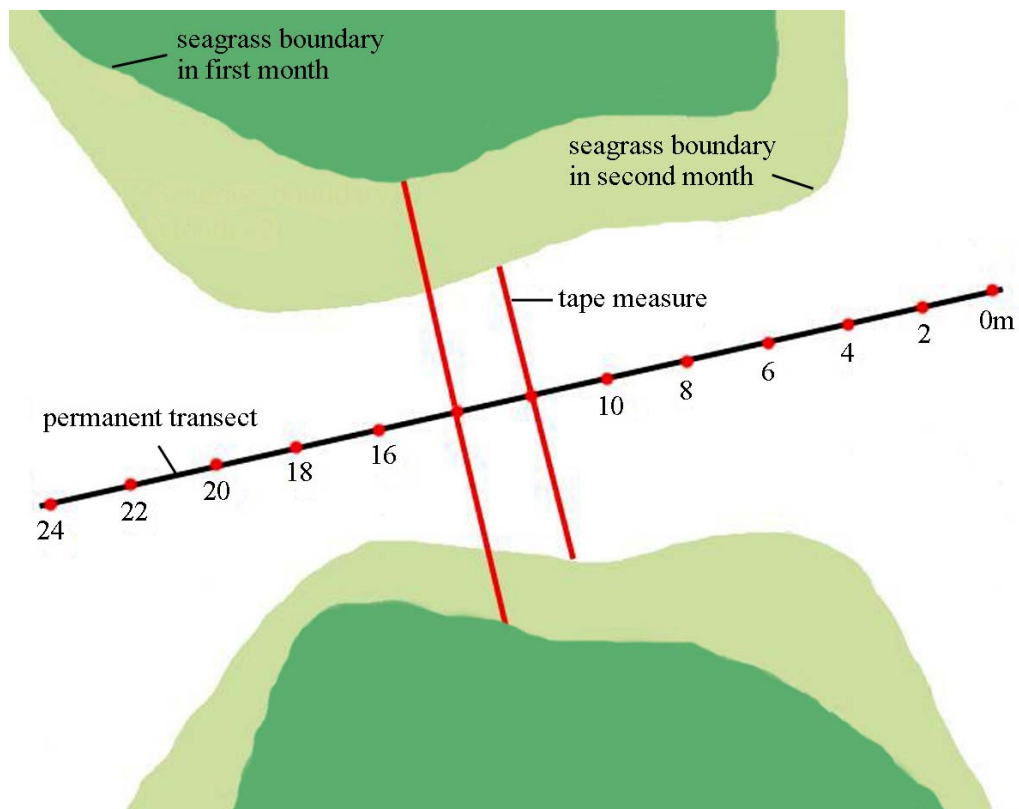


Figure 5. Procedure for mapping *Zostera marina* boundary extension at Barloge Creek.

**Associated Faunal Diversity**

Replicate infaunal samples were collected in April, August and December of 2004 and 2005 at all sites using plastic cylindrical corers with a width of 105 mm to a depth of 120 mm. Cores containing the sample were placed in a plastic bag, tagged and sealed before returning to the surface. Samples were sieved in seawater through a 0.5 mm sieve and preserved in 70 % alcohol immediately. Rose bengal dye was added to all samples to dye all organisms pink and so aid the sorting process. In the laboratory, organisms were removed from the sediments by hand and identified to species level where possible. All specimens were stored in 70 % alcohol after identification. Gastropod and annelid specimens were not included as their small size (gastropods) and damage during the sieving process (annelids) made it impossible to determine the abundance of live individuals. Total species abundance (N), species richness (S) and species diversity, using the Shannon-Weaver Diversity Index ( $H'$ ), were calculated using Primer-E software (Plymouth Marine Laboratory 2006) and analysed using one-way ANOVAs in the statistical package SPSS for Windows (2003).

### 3. RESULTS

#### Environmental parameters

The highest temperatures recorded were in August 2005 at all sites (Barloge Creek site 16.27 °C; Ventry Bay shallow site 16.65 °C; Ventry Bay intermediate site 16.25 °C; and Ventry Bay deep site 15.95 °C). The lowest temperatures recorded were in March/April 2004 (Barloge Creek site 8.23 °C; Ventry Bay shallow site 9.80 °C; Ventry Bay intermediate site 9.80 °C; and Ventry Bay deep site 9.60 °C). Water temperature was comparable between Barloge Creek and Ventry Bay in those months where sampling was undertaken at both sites. Water temperature was only found to decline with depth in April and August 2004 and August 2005 sampling periods (Fig. 6).

Salinity at Barloge Creek and Ventry Bay measured 34.6 - 35.3 psu and 33.7 – 35.2 psu, respectively. Lowest salinities were recorded in the winter months (December/January sampling) and highest salinities were recorded in summer (August sampling), but all fluctuations were within the range expected for a fully marine system.

Grain-size composition of sediments showed little variation over the course of a year at all sites. However, sediments at Barloge Creek and Ventry Bay were shown to have very different grain-size compositions. The sediment at Ventry Bay was mainly sand (sediment composition 89.04 – 94.74 % sand) with a low silt percentage, and composition varied little between depths. Conversely, the sediment at Barloge Creek had larger variation in grain-size (sediment composition 30.41 % gravel, 36.72 % sand and 28.22 % silt) (Table 1). Sediment organic carbon content data showed considerable variation between sites. Barloge Creek sediment had a higher organic carbon content than the Ventry Bay sediments at all times of the year. Variability was greatest in the Ventry Bay shallow site (range recorded 1.38 – 3.45 %) and narrowest in the Ventry Bay deep site (range recorded 1.47 - 2.19 %). Minimum percentage organic carbon contents were recorded in August 2005 at Barloge Creek (3.29 %) and August 2004 at Ventry Bay (1.38 %, 1.38 % and 1.47 % for the shallow, intermediate and deep sites respectively). Maximum percentage organic carbon contents were recorded in April 2005 for Barloge Creek and Ventry Bay deep site (4.71 % and 2.19 % respectively), and in December 2004 for Ventry Bay shallow and intermediate sites (3.45 % and 2.93 % respectively) (Fig. 7).

Table 1. Summary of sediment grain-size composition at Barloge Creek and Ventry Bay.

Site	Mean percentage of sediment sample				
	> 16 mm	> 8 mm	> 2 mm (gravel)	> 63 µm (sand)	< 63 µm (silt)
Barloge Creek (4 m)	10.37	5.69	14.35	36.72	28.22
Ventry Bay (4 m)	0	0.38	0.55	89.04	9.37
Ventry Bay (7 m)	0.07	0.19	0.34	94.10	5.35
Ventry Bay (10 m)	0.08	0.16	0.65	94.74	4.31

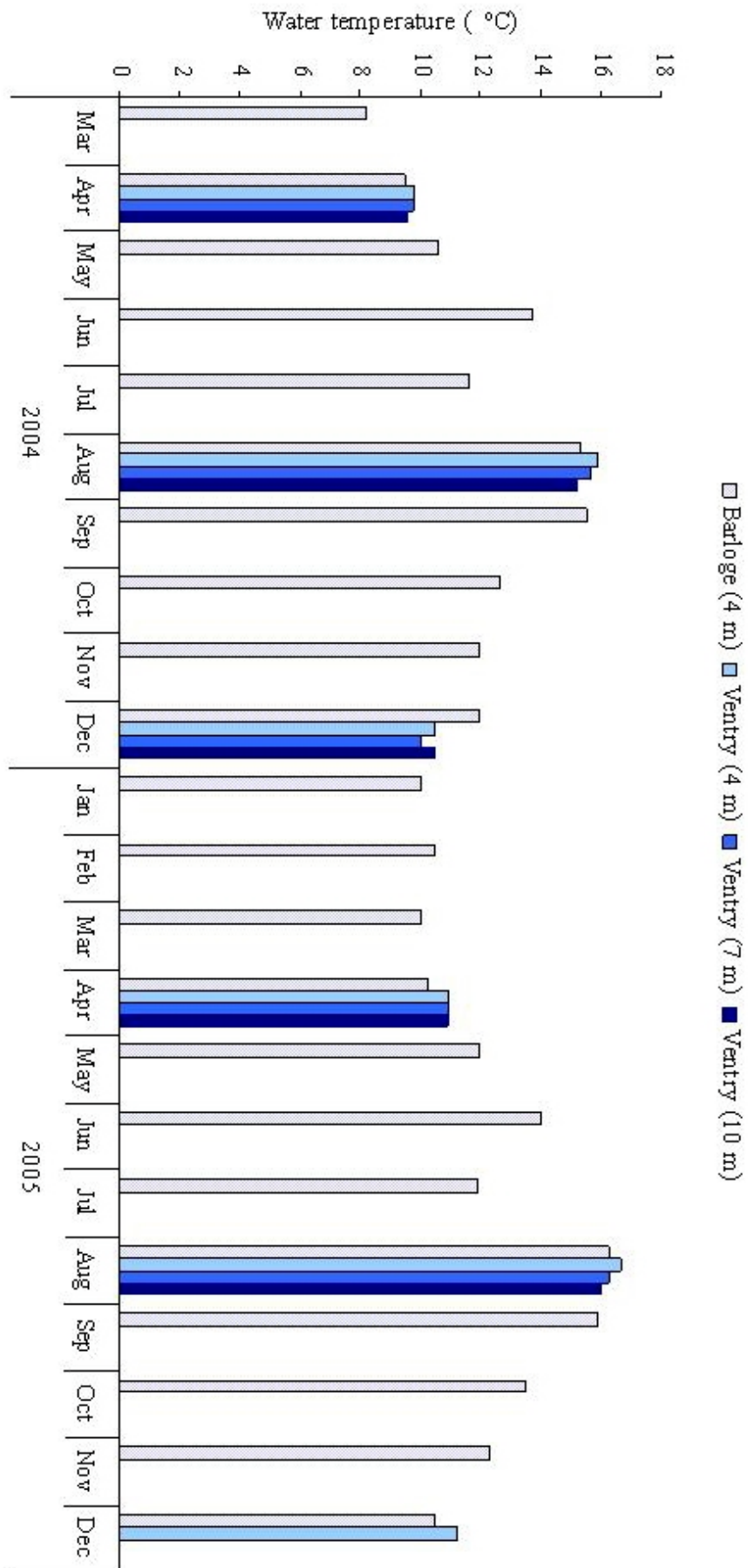


Figure 6. Monthly bottom water temperatures at Barloge Creek and Ventry Bay.

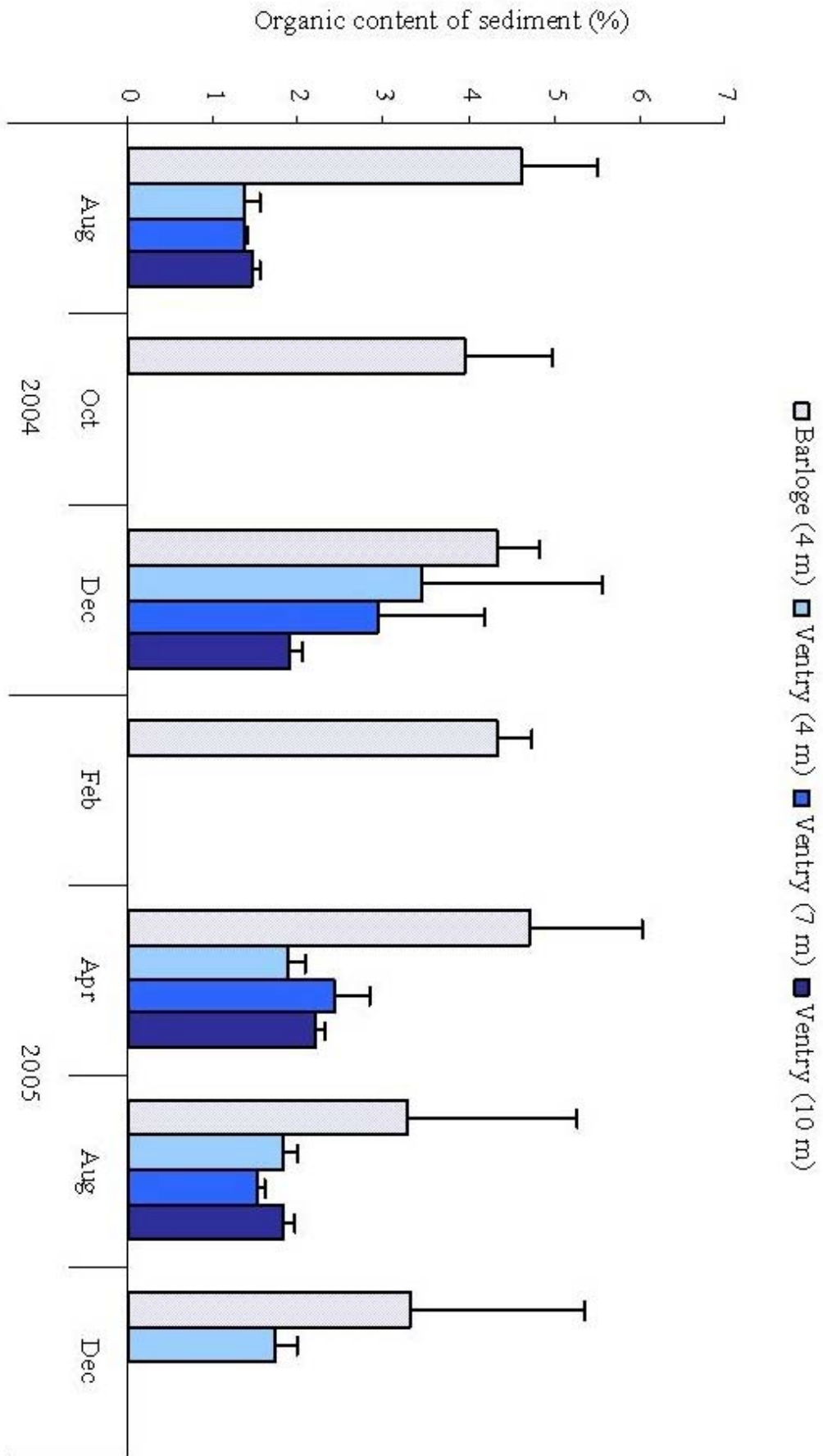


Figure 7. Seasonal variation ( $\pm$  s.d.) in organic carbon content of sediment at Barloge Creek and Ventry Bay.

### **Physical Structure of Plant Communities**

Monthly data showed large fluctuations in *Zostera marina* shoot density, both within and between years. Maximum density at Barloge Creek (168.33 shoots/m<sup>2</sup>) was recorded in July 2005 and minimum density (72.33 shoots/m<sup>2</sup>) was recorded in January 2005 (Fig. 8). ANOVA showed shoot density at Barloge Creek to be significantly different between seasons ( $P < 0.001$ ,  $F = 24.244$ ,  $df = 3$ ). Furthermore, shoot density was significantly higher ( $P < 0.01$ ) in summer (June-August) 2005 than in summer 2004. Shoot density at Ventry Bay was shown to decline with depth at all times of the year, over both years. ANOVAs showed there to be a significant difference in shoot density between each depth ( $P < 0.001$ ,  $F = 47.170$ ,  $df = 2$ ). There was no significant difference in shoot density between seasons at the shallow and intermediate sites. However, shoot density recorded at the deep site in August and December 2004 was significantly lower ( $P < 0.01$ ,  $F = 6.363$ ,  $df = 4$ ) from all other sampling months. The time of highest recorded density changed with year at the shallow and intermediate sites with only the deep site reaching peak density in spring (April sampling) during both years. The highest shoot densities recorded at each site over the duration of the study were 252.8 shoots/m<sup>2</sup> at the Ventry Bay shallow site (August 2004), 184 shoots/m<sup>2</sup> at the Ventry Bay intermediate site (August 2005), and 147 shoots/m<sup>2</sup> at the Ventry Bay deep site (April 2005) (Fig. 8).

Measurements to monitor the expansion of *Zostera* into an area of bare sediment in Barloge Creek showed a boundary growth of 4 – 20 m between June and July 2004 and a further  $\leq 3$  m between July and August 2004 (Fig. 9). Measurements in subsequent months did not show any further extension or retraction of the bed. The 0 – 8 m section of transect was laid along an incorrect bearing in June 2004 so measurements along this section could not be taken at this time. This section of transect was re-laid in time for the July and subsequent month measurements.

Monthly data from Barloge Creek showed large intra-annual fluctuations in *Zostera marina* maximum blade length per shoot. ANOVA showed there to be no significant difference between summer and autumn (June – November sampling) or between winter and spring (December – May sampling) but maximum blade length in summer/autumn was significantly greater than winter/spring ( $P < 0.001$ ,  $F = 31.320$ ,  $df = 3$ ) (Fig. 10). Data from Ventry Bay showed large intra-annual fluctuations in maximum blade length per shoot at the shallow, intermediate and deep sites. The longest blade of the sampling period was significantly shorter in the deep site than in the shallow and intermediate sites ( $P < 0.001$ ,  $F = 10.714$ ,  $df = 2$ ). Inter-annual variation was only apparent at the intermediate ( $P < 0.05$ ) and deep ( $P < 0.001$ ) Ventry Bay sites where the longest blades recorded were significantly shorter in August 2004 than in August 2005. Maximum blade length was longest in the summer (August sampling) of each year at all depths.

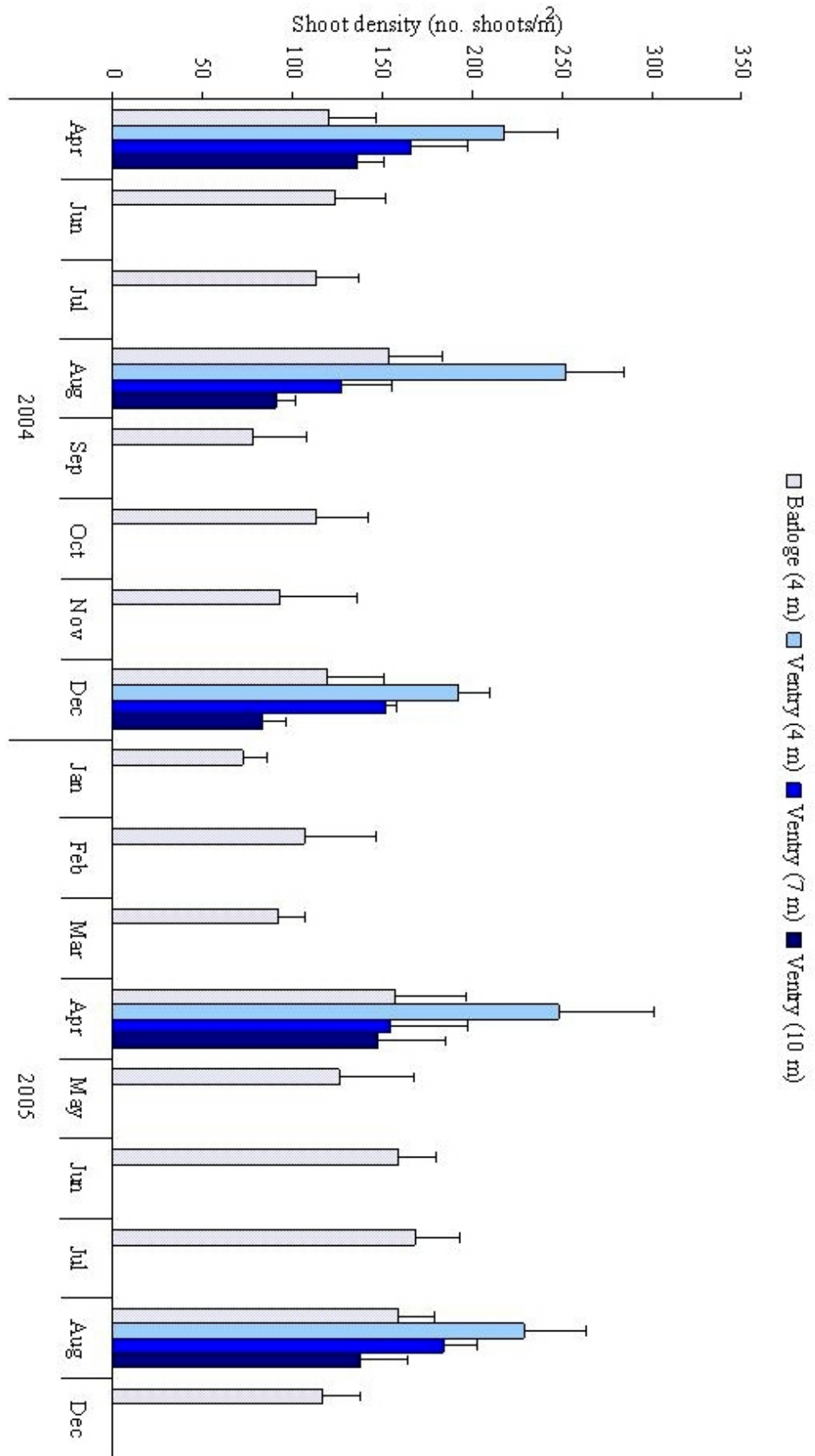


Figure 8. Monthly variation ( $\pm$  s.d.) in shoot density at Barloge Creek and Ventry Bay.

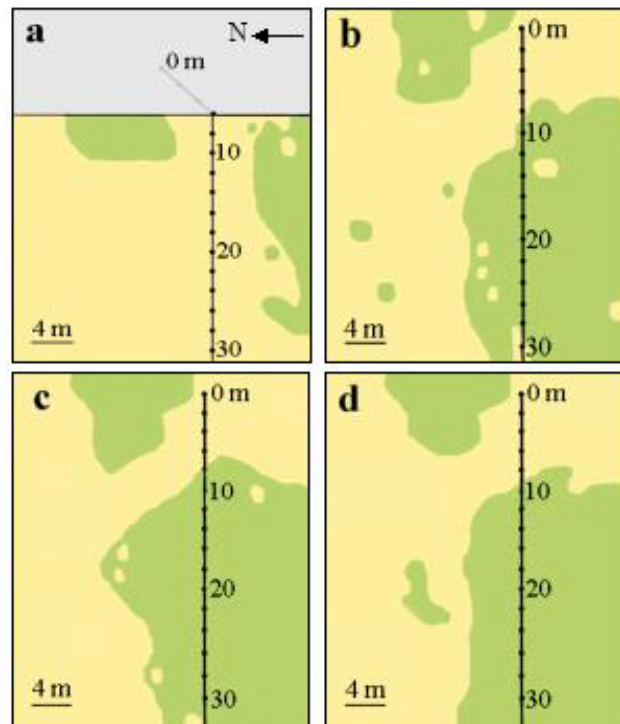


Figure 9. Diagrammatic representation of northwards expansion of *Zostera marina* in Barloge Creek. (a) June 2004, (b) July 2004, (c) August 2004, and (d) September 2004. Seagrass (green), bare substrate (yellow).

There was a large range of maximum blade lengths throughout the year at each site. Maximum blade lengths were 59 – 74% greater at the upper end of the range than lengths recorded at the lower end of the range (Table 2). Longest blade lengths were encountered in summer (August/September sampling) at all sites, while shortest blade lengths occurred in winter (December sampling) at the Ventry Bay intermediate and deep sites, and spring (March/April sampling) at the Ventry Bay shallow site and Barloge Creek (Fig. 10).

Table 2. Mean range of maximum blade length per shoot at Barloge Creek and Ventry Bay.

Site	Longest blade range (mm)	Percentage difference
Barloge Creek (4 m)	383.25 – 998.50	62 %
Ventry Bay (4 m)	400.25 – 969.25	59 %
Ventry Bay (7 m)	301.00 – 933.75	68 %
Ventry Bay (10 m)	155.00 – 577.50	74 %

No obvious seasonal pattern was revealed in the root-rhizome biomass data at any of the study sites. ANOVA showed the root-rhizome biomass at the Barloge Creek, Ventry Bay intermediate site and Ventry Bay deep site did not significantly differ throughout the year. However, samples collected from the Ventry Bay shallow site in April 2004 were shown to have a significantly lower biomass than those collected in December 2004, August 2005 and December 2005 ( $P < 0.001$ ,  $F = 5.372$ ,  $df = 5$ ). Root-rhizome biomass at the Ventry Bay shallow and deep sites were shown to significantly differ from all other sites ( $P < 0.001$ ,  $F = 51.567$ ,  $df = 3$ ). Biomass data for Barloge Creek and Ventry Bay intermediate site each significantly differed from Ventry Bay shallow and deep site data (Fig. 11).

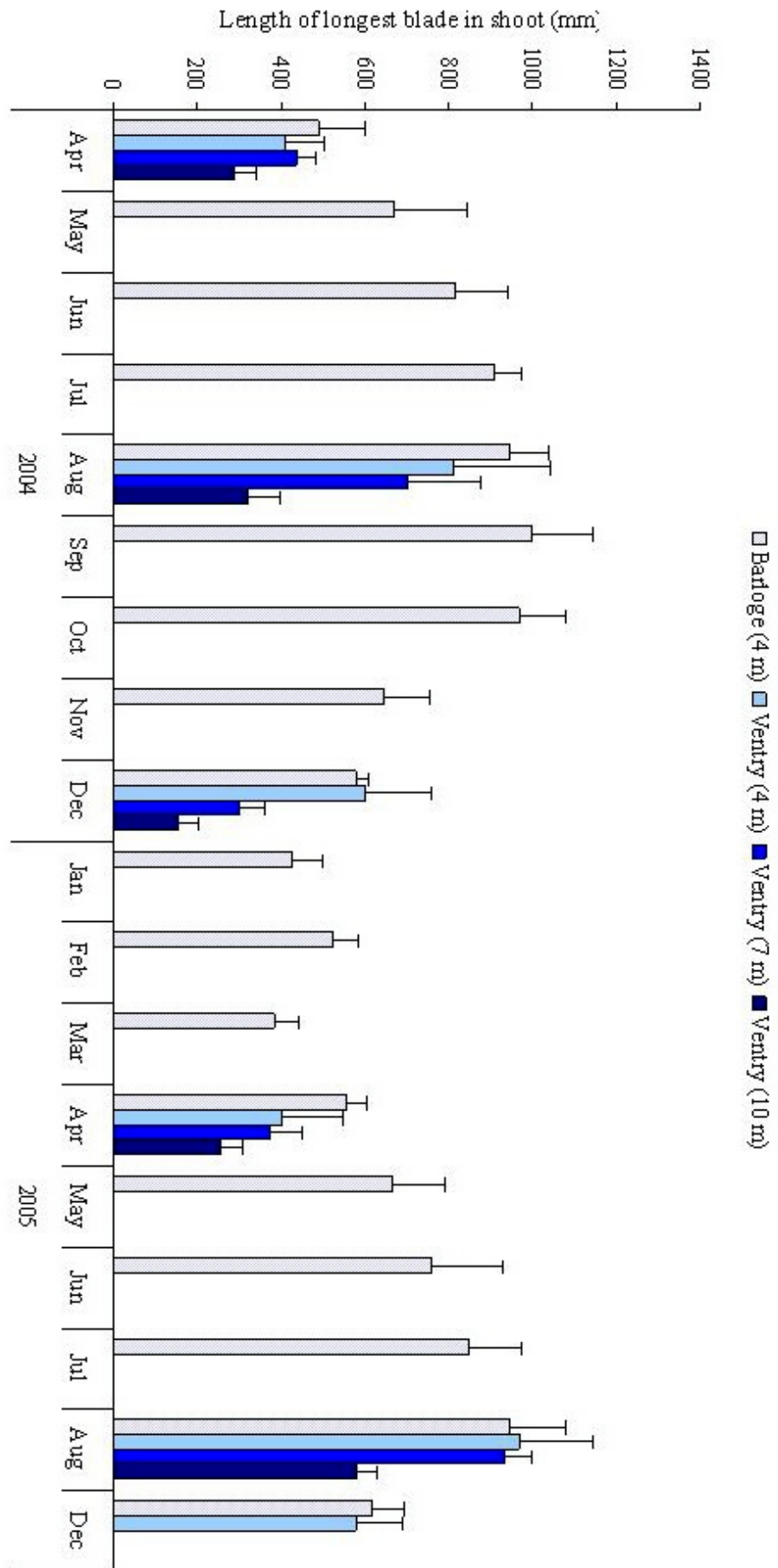


Figure 10. Monthly variation ( $\pm$  s.d.) in maximum blade length per shoot at Barloge Creek and Ventry Bay

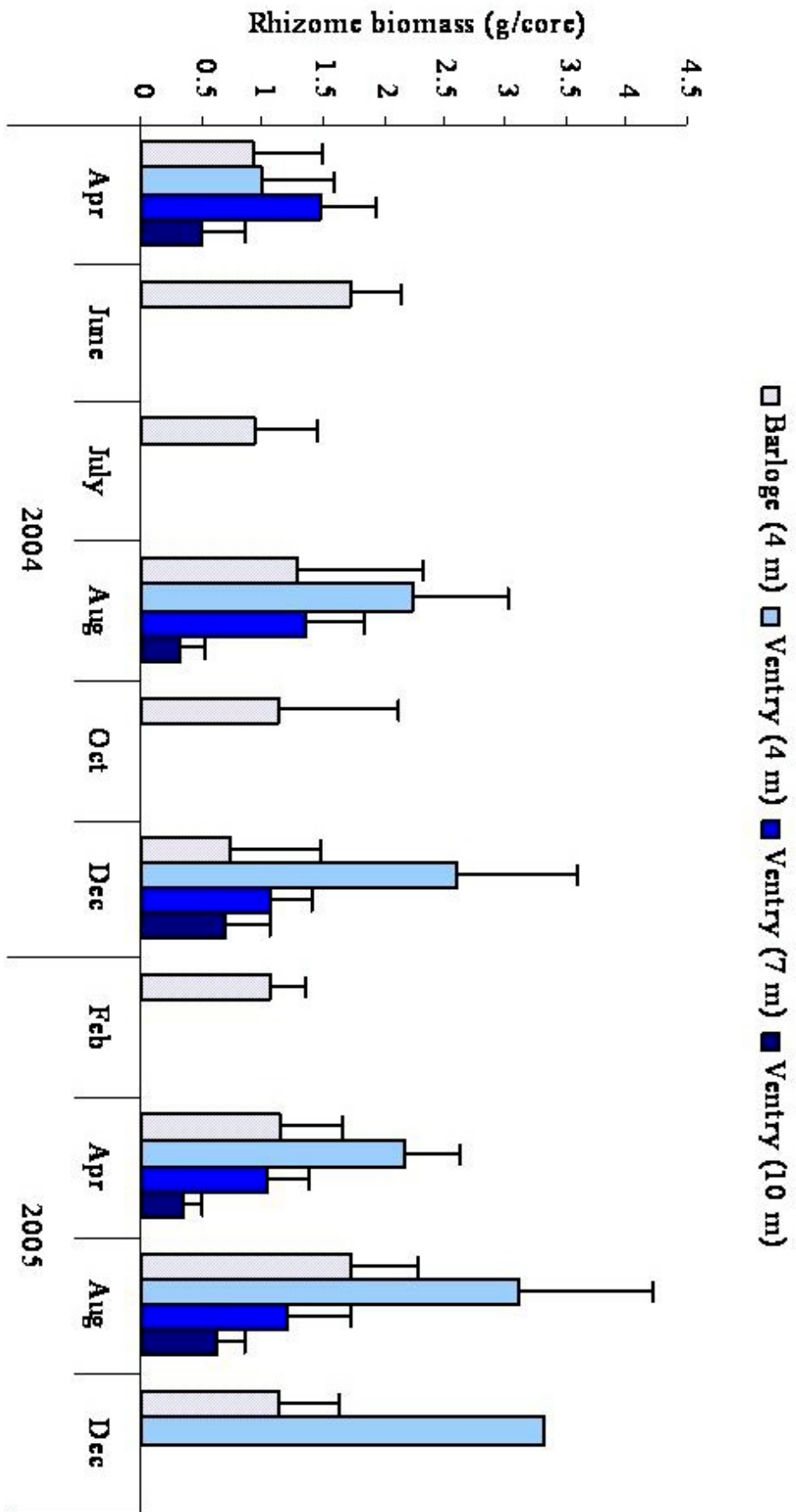


Figure 11. Seasonal variation ( $\pm$  s.d.) in *Zostera marina* root-rhizome biomass at Barloge Creek and Ventry Bay.

### **Associated Faunal Diversity**

A total of 124 species representing 81 families from six phyla were recorded from infaunal samples. Thirty-two of these species were only found at Barloge Creek, while 45 were only found at Ventry Bay and a further 47 were found at both locations. Samples of epifauna occurring on *Zostera* blades obtained from a separate study at each of the sites included 15 species not represented in the infauna samples (see Appendices for species lists). The amphipods *Microdeutopus anomalis*, *M. versiculatus* and *Dexamine spinosa*, the tanaid *Leptochelia savigni* and the stalked jellyfish *Haliclystus salpinx* and *Lucernariopsis cruxmelitensis* are all noted to occur on *Zostera* and *Laminaria* (Hayward and Ryland 1995). Although these amphipods were found at both sites, the stalked jellyfish was only found at Ventry Bay. Many species occurred in low numbers, usually with <50 individuals collected over the course of the study but 27 species generated >50 individuals. The amphipods *Erichthonius punctatus* and *Corophium sextone*, the brittlestar *Amphipholis chiajei*, and the bivalves *Lucinoma borealis* and *Moerella donacina* occurred in numbers of >200 individuals each over the course of the study. Of the most common species (>50 individuals collected) three were only found at Barloge Creek (amphipods *Parametaphoxus fultoni* and *Socarnes erythrophthalmus*, and the tanaid *Apseudes talpa*) and seven were only found in Ventry Bay (amphipods *Pariambus typicus* and *Urothe elegans*, the brittlestar *Amphiuridae* spp. 1, cumacea *Diastylis rugosa* and *Eudorella truncatula*, bivalve *Thracia phaseolina* and the ostracod *Euphilomedes interpuncta*).

The high total abundance standard deviation shown for Barloge Creek in December 2004 (Fig. 12) is mainly due to the exceptionally high abundance of *Corophium sextone* (246 individuals) in one of the cores, although the bivalve *Mysella bidentata*, the brittlestar *Amphipholis chiajei*, and the crustacea *Apseudes talpa* and *Microdeutopus versiculatus* all also had abundances of >50 individuals for this core. ANOVA showed that the infaunal communities from Barloge Creek and the Ventry Bay deep site did not demonstrate any significant seasonal fluctuations in total abundance (N), species richness (S) or species diversity (H'). Infaunal samples from the shallow site at Ventry Bay showed no significant seasonal variation in total abundance (N), but the species richness (S) of samples collected in April 2004 was significantly lower from those collected in August 2004, August 2005 and December 2005 ( $P < 0.05$ ,  $F = 4.074$ ,  $df = 5$ ). Furthermore, species diversity (H') at the Ventry Bay shallow site was significantly lower in April 2004 than in all other sampling months ( $P < 0.001$ ,  $F = 10.641$ ,  $df = 5$ ) (Fig. 12). Barloge Creek infaunal total species abundance (N) ( $P < 0.001$ ,  $F = 6.982$ ,  $df = 3$ ) and species diversity (H') ( $P < 0.001$ ,  $F = 6.622$ ,  $df = 3$ ), but not species richness (S), was significantly different to that recorded at each of the Ventry Bay sites over the course of the study. At Ventry Bay there was no significant difference in total abundance (N), richness (S) or diversity (H') of species between depths.

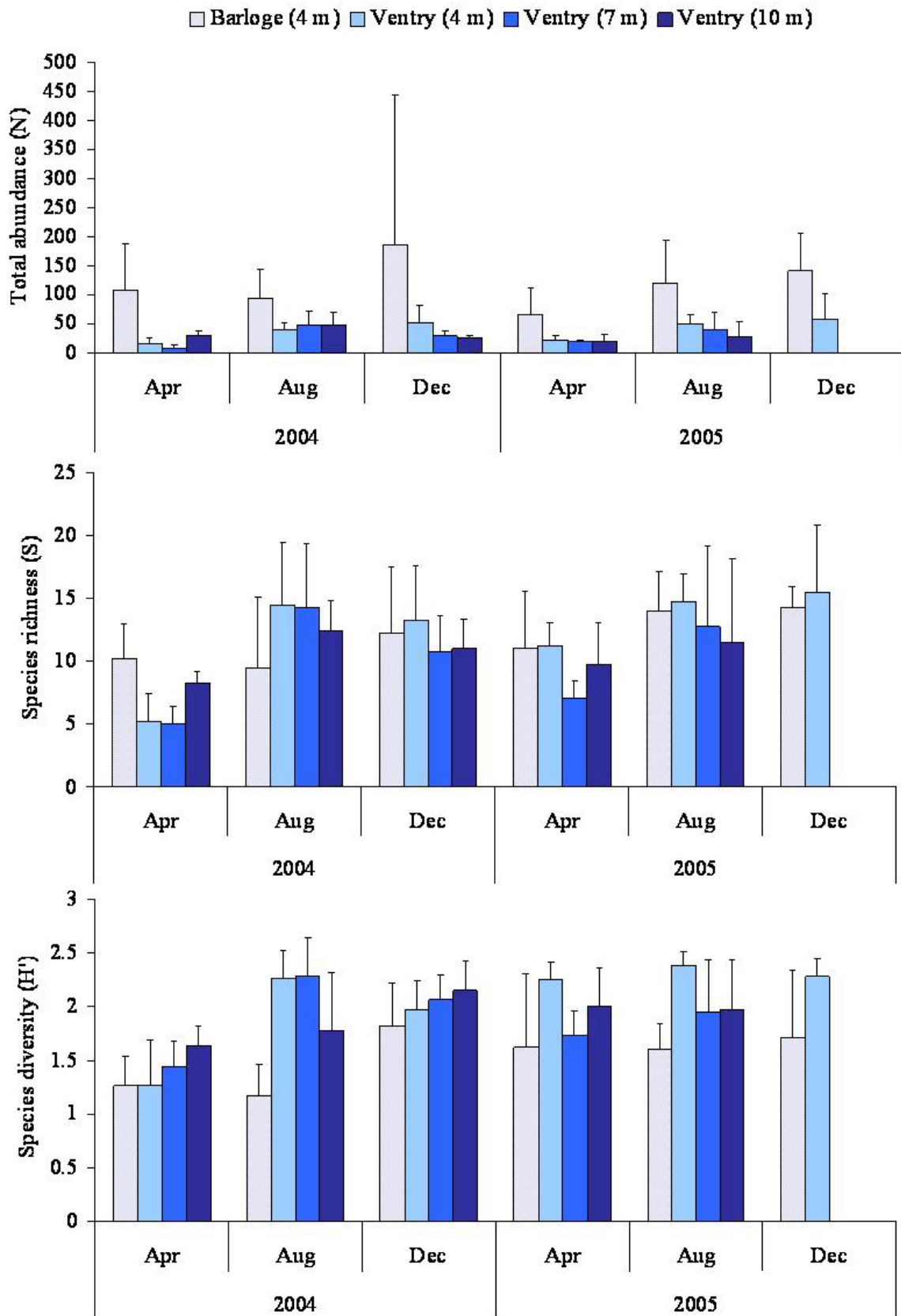


Figure 12. Seasonal variation (+ s.d.) in total abundance (N), species richness (S), and species diversity (H') of *Zostera marina* infauna at Barloge Creek and Ventry Bay.

#### 4. DISCUSSION

Many of the seagrass studies undertaken to date do not consider seasonal and inter-annual variation, with experiments being carried out over the course of a single month or season (e.g. Lewis and Stoner 1983; Somerfield *et al.* 2002). This present study was designed to explore aspects of the growth dynamics and associated faunal diversity of two subtidal *Zostera marina* beds in Ireland across spatial and temporal scales.

Seagrass tissues suffer osmotic stress at both high and low salinities. Salinities exceeding 45 psu can result in a decline in growth and mortality (Quammen and Onuf 1993). However, seagrass species can tolerate a range of salinities allowing them to occur in estuarine and lagoonal systems as well as fully marine environments. Subtidal *Zostera marina* is intolerant to desiccation and large fluctuations in salinities and so avoids brackish waters (Madden *et al.* 1993). Conversely, *Z. noltii* is more tolerant to large fluctuations in salinity and highly resistant to desiccation and so can be found in intertidal zones (Davison and Hughes 1998). None of the sites studied in this report are close to freshwater stream outlets and so water salinity measurements were within the range expected for a fully marine system and showed very little fluctuation over the course of a year. Water salinity at Ventry Bay fluctuated by 1.3, 1.0 and 1.4 psu for the shallow, intermediate and deep site respectively over the year whereas Barloge Creek showed slightly less fluctuation at only 0.7 psu. Similarly, measured seasonal fluctuations in bottom water temperature were comparable with monthly sea surface temperatures recorded in Irish waters (Irish Meteorological Service 2006).

High currents and wave action can generate movement of sediment, preventing roots from becoming established and causing damage to seagrass blades (Hemminga and Duarte 2000). Barloge Creek experiences minimal wave action as it is sheltered from strong winds by Carrigathorna Rocks at the mouth of the Creek and by steep banks on either side of the channel. There is a tidal current running through Barloge Creek. This current is strongest in the center of the channel where the root-rhizome mat appears to be unable to extend into this region (*pers. obs.*). This confines the *Zostera marina* bed to the peripheral margins of the Creek where currents are weaker and root-rhizomes may establish more readily. Ventry Bay is also sheltered from the exposed coastline. The wide opening of the embayment into Dingle Bay prevents channeling and so the tidal current is weaker than at Barloge Creek. Wave action is increased towards the center of the bay (as shelter from land and bedrock is reduced). However, the increase in depth buffers the seabed from the turbulence resulting in the sediment remaining undisturbed. Therefore, the presence of bedrock at the shallow site and restricted light penetration at the intermediate and deep sites appears to be the restricting factors in seagrass growth in Ventry Bay. Regular or extensive disturbance of the sediment can result in erosion of the root-

## Management of *Zostera* beds

rhizome mat and shoot burial leading to seagrass mortality in established beds (Hemminga and Duarte 2000). That no such effects were observed during this work suggests that the *Zostera* beds at both locations were not subject to physical disturbance.

Seagrasses are often nitrogen limited and while their growth can be stimulated by nutrient inputs (e.g. from sewage or agricultural drainage), excessive nitrate and ammonium concentrations are known to be directly toxic to seagrass. van Katwijk *et al.* (1997) reported water column ammonium concentrations of 25  $\mu\text{M}$  to cause death in *Zostera marina* after five weeks exposure whereas nitrate concentrations of 35  $\mu\text{M}$  can also result in death after 4-5 weeks (Burkholder *et al.* 1992). *Z. marina* blades have no method to regulate nitrogen uptake, whereas roots are able to regulate nitrogen uptake from the sediment by controlling the number of root hairs (Tennant 2006). This suggests *Z. marina* beds are particularly susceptible to damage from excessive nutrient loading of the water column but are better able to tolerate elevated nitrogen concentrations in the sediment. Temperate seagrass beds are more likely to experience seasonal nutrient limitation than tropical beds. This is due to the less stable temperature and light conditions in the temperate regions resulting in restricted periods of fast growth (Hemminga and Duarte 2000). In addition, excessive nutrient enrichment promotes algal growth and stimulates planktonic blooms resulting in reduced light penetration, smothering and ultimately death of the seagrass (Hemminga and Duarte 2000; Granger and Iizumi 2001; Robinson 2003). Studies into the nutrient limitation/loading in each system would be beneficial to further understanding the trends in *Zostera* growth at these sites.

Sediment grain-size composition at Barloge Creek has a much larger range of grain-size classes than Ventry Bay, which was predominantly sand (Table 1). This is partly due to the differing flow regimes at each site. The Barloge Creek *Zostera* bed is outside the main flow channel and the energy differential between it and the main flow channel results in regular deposition of suspended fine particles in the seagrass bed (or lower energy) area. Conversely, Ventry Bay is more open with no apparent energy differential that would result in deposition of suspended particles. Furthermore, the *Zostera* bed at Barloge Creek is located directly adjacent to a rocky shore with a high gravel component, while the abundance of detritus material within the bed aids the capture of fine particles passing in the current and hence increasing the sediment silt fraction. Conversely, the *Zostera* bed at Ventry Bay is located directly adjacent to bedrock and sandy beach resulting in a relatively uniform sediment grain-size composition. Sediment at Barloge Creek has a higher organic carbon content than Ventry Bay at all times of the year (Fig. 7), primarily as sediment organic carbon content is positively correlated with silt content. However, the Barloge Creek *Zostera* bed has a high level of epiphytic growth throughout the year as well as an abundance of detritus material on the sediment surface and the presence of several species of macrophyte (e.g. *Sargassum muticum*, *Himanthalia elongata*, *Ulva*

*lactuca*), all of which provide the system with an additional source of carbon. In contrast, the *Zostera* beds at Ventry Bay have a lower silt content, are relatively free of epiphytes and detritus material and have few macrophytes living among the *Zostera* shoots (pers. obs.).

Shoot density at Barloge Creek showed seasonal and annual variation with densities being significantly higher in summer 2005 compared with summer 2004. The shallow site at Ventry Bay supports a far greater shoot density than Barloge Creek despite being at a similar depth (Fig. 8). This may be attributable to differences in light regimes between the two locations. Seagrass growth and photosynthesis requires approximately 10 % of surface irradiance (the amount of light that filters through the water column) (Greve and Binzer 2004). Irradiance is reduced as water depth and turbidity increases and so the maximum depth distribution is often determined by water clarity (Hemminga and Duarte 2000). Incidents of *Zostera* decline caused by reduced light availability are well documented (e.g. Sand-Jensen 1977; Moore *et al.* 1997; Moore and Wetzel 2000; Pergent-Martini *et al.* 2005). Barloge Creek has strong tidal currents resulting in regular re-suspension of fine silt material thus reducing light penetration through the water column, whereas Ventry Bay has weak tidal currents resulting in little re-suspension of the heavier sand material present there (Table 1). In addition, the Barloge Creek bed encompassed by steep banks may not experience as much direct sunlight as the more open landscape of Ventry Bay. The heavy epiphytic load on blades in the Barloge Creek bed may also in effect inhibit light availability to the plant. Sand-Jensen (1977) found that epiphytes were able to reduce photosynthesis of *Z. marina* blades by up to 58 % by acting as a barrier to carbon uptake and reducing light intensity, while Bulthuis and Woelkerling (1983) found that epiphyte biomass was able to accumulate on the blades of *Heterozostera tasmanica* fast enough to reduce the time in which positive net photosynthesis was possible (effectively reducing the leaf's lifespan to less than half). Additional light penetration at Ventry Bay could lead to a higher shoot density in shallow waters and an increased depth limit of the bed as a whole. A seasonal pattern was not observed at the Ventry Bay shallow and intermediate sites suggesting that light penetration levels remain relatively constant throughout the year. Shoot density at the deep site was significantly lower in August 2004 than in August 2005 suggesting that the same inter-annual fluctuation is being followed as at Barloge Creek. Furthermore, the significantly lower shoot density recorded in December 2004 suggests that this deep site may also follow a seasonal pattern, although data from subsequent years (not collected due to adverse weather conditions) would be needed to confirm this hypothesis.

Shoot density at Ventry Bay was shown to decrease with depth at all times of the year. The Ventry Bay shallow site is very sheltered by rocks and land and its shallow depth facilitates light penetration resulting in high shoot density. As the increase in depth protects the seabed from surface generated

disturbance and epiphyte load is low throughout the year, the observed decline in shoot density with depth at Ventry Bay (Fig. 8) is likely to be connected to the reduction in light availability with depth. Whelan (1986) also found a decrease in shoot density with depth at Ventry Bay. A separate study comparing Ventry Bay shoot density data presented here with data from similar locations collected by Whelan in the 1980s showed that shoot density now demonstrated less seasonal fluctuation during the period of this survey. The intermediate and deep sites both recorded a greater shoot density during this work than that measured 25 years ago. However, *Zostera marina* is no longer present at the 1 m depth site surveyed in the 1980s (recording 351 shoots per m<sup>2</sup>) and the shallow (4 m) site surveyed in 2004-2005 (recording 256 shoots per m<sup>2</sup>) was close to the boundary of the bed. This suggests that the Ventry Bay *Zostera* bed has contracted its range southwards into deeper water, although the reason for such a change or its significance is unclear at this time.

Seasonal patterns were less obvious but still apparent in maximum blade length data with a significant difference in maximum blade length between summer/autumn sampling and winter/spring sampling at Barloge Creek. Significant seasonal differences were not found at any of the Ventry Bay sites although the maximum length blade of the sampling periods was found in summer for both Barloge Creek and Ventry Bay sites. These results are also likely to be due to light penetration constraints as the summer months provide longer daylight hours and calmer weather conditions resulting in less sediment disturbance. Inter-annual differences between August 2004 and August 2005 were again apparent at the intermediate and deep sites of Ventry Bay with maximum blade length recorded in August 2004 being significantly lower than in August 2005. The growth patterns of all seagrasses are influenced by seasonal change although the magnitude of fluctuations in growth and the response to changes in light and temperature conditions are species specific (Marbà *et al.* 1996). Seasonal and inter-annual variation in several growth parameters (e.g. shoot biomass, shoot density, shoot elongation, leaf growth rate, rhizome elongation) have been reported for a number of species (e.g. Bulthuis and Woelkerling 1983; Marbà *et al.* 1996; Creed 1999). A study of Mediterranean seagrass growth patterns by Marbà *et al.* (1996) showed *Cymodocea nodosa* growth to be most influenced by temperature fluctuations whereas *Zostera marina* growth was more closely linked to seasonal fluctuations in light conditions. This observation is supported in data represented here, as both shoot density and maximum blade length decreased with depth despite water temperatures remaining relatively constant.

There were no obvious patterns in root-rhizome biomass during the course of the study. There appeared to be no seasonal variation across the sampling period but significant differences between sites were found in some months. As patterns in shoot density do not appear to correspond to root-

rhizome biomass data, above ground complexity (shoot density) should not be used as a direct proxy for below ground complexity (rhizome biomass).

Measurement to track the growth of *Zostera marina* into an adjacent area of bare sediment at Barloge Creek showed the majority of growth to occur between June and August 2004 (Fig. 9). The numbers of daylight hours available for photosynthesis, the level of organic carbon in the sediment and water temperature were all high during this time providing optimum conditions for seagrass growth. *Z. marina* rhizomes elongate by approximately 26 cm yr<sup>-1</sup> (Hemminga and Duarte 2000) so it is unlikely that rhizomes were completely absent from this area in June 2004. Root-rhizome material was not present in sediment cores taken along the transect in June and July 2004, but present in cores taken in August 2004 and thereafter. This suggests that the root-rhizome matrix was present in the bare area throughout the study but only became sufficiently dense to support a canopy and appear in cores from August onwards. The newly grown canopy did not retreat back to its original margins during the winter months as expected. This implies that the expansion of the bed is not a seasonal phenomenon and may reflect re-growth into an area that had previously died back due to a disturbance or new growth into an area that was previously unsuitable. The observation that the smaller patch of *Z. marina* to the north of the transect did not also grow into the bare area adjacent to its margin throughout the study suggests that substrate in this area is unsuitable for growth.

Infaunal data presented in this report may be considered an underestimation of both abundance and species diversity due to the omission of gastropods and annelids from the data set. There was no significant seasonal difference recorded in total species abundance (N), species richness (S) or species diversity (H') at Barloge Creek or the Ventry Bay deep site. Seasonal differences found in the Ventry Bay shallow and intermediate site, when present, were the result of infaunal data from April 2004 being unusually low in comparison to the rest of the sample months.

Borg *et al.* (2006) found rhizome material to be important to the infaunal communities of *Posidonia oceanica* beds, with a 'dead' root-rhizome matrix (without shoots) supporting a higher species diversity and total abundance of infauna than adjacent areas with living (with shoots) matrices. This indicates that the presence of rhizomes plays an important role in some seagrass systems by increasing structural complexity. Data represented here showed that faunal composition did not follow trends in rhizome biomass, suggesting that although there is likely to be a link between faunal composition and the presence of root-rhizome material other components of the habitat such as grain-size composition has more influence at these sites.

## Management of *Zostera* beds

Total species abundance (N) at Barloge Creek was significantly higher than at all of the Ventry Bay sites. This may be due to the high organic carbon content of the sediment coupled with the layer of detritus providing an abundant food source for infaunal populations. However, Barloge Creek infauna data showed very large fluctuations in total abundance of individuals (N) between replicates. This would suggest that total abundance cannot be used as a meaningful measure to assess differences between season or between sites without a much larger pool of replicate data.

### **Management Implications**

*Zostera* beds constitute a very important component of several habitats protected under the 1992 EU Habitats Directive. Indeed, the important role that seagrass beds play in biodiversity and the potential loss of ecosystem service that accrues through their destruction resulted in their placement on the 2004 Initial List of Threatened and Declining Species and Habitats by the OSPAR Commission. As well as having intrinsic value that merits protection, they are also potential indicators of broader habitat quality and the full realisation of this particular role will depend on a comprehensive understanding of the natural variability that characterise seagrass beds in the wild.

Monitoring methods using seagrass growth parameters such as shoot density and canopy height are relatively cheap, easy to carry out and non-destructive. They can provide a good evaluation of habitat quality, as well as acting as a proxy measure of habitat complexity and refuge capability. However, *Zostera marina* plants can be either annual or perennial with some annual beds known to disappear completely during some months of the year (Muñiz-Salazar *et al.* 2005). Comprehensive work by both Whelan (1986) and the current study, while showing very high annual and seasonal variation and forming a foundation for future baseline studies, highlights the need for studies that integrate both flora and fauna simultaneously. The findings of this work whilst detailing the extent of natural variability suggest that a broader approach within the context of robust baseline data may be more appropriate for monitoring abundance.

When monitoring the growth parameters of a seagrass bed at a site, it is therefore important to establish:

- if the plants have an annual or perennial life cycle,
- natural growth fluctuations over time and depth,
- baseline data that facilitates the use of broad abundance scales.

Previous studies on patch size and edge effect suggest that small seagrass beds have reduced infaunal species diversity and organism abundance (e.g. Frost *et al.* 1999; Bowden *et al.* 2001). Data from

Barloge Creek showed that small beds were able to support a rich faunal community despite covering a much smaller area and having a coarser sediment grain-size than the Ventry Bay beds. It cannot be assumed therefore that dense beds support a richer faunal diversity or abundance than sparse beds and this highlights the need to obtain data from beds ranging in size and across physico-chemical regimes.

Faunal communities require considerably more work effort and expertise to monitor effectively than growth parameters of the seagrass themselves. As infaunal communities are likely to be least affected by above-ground seagrass degradation or fragmentation, any change observed outside natural fluctuations will indicate a major disturbance in the system. Furthermore, significant changes noted in the relatively stable infaunal communities could be amplified for larger organisms that are more susceptible to above-ground deterioration or fragmentation (Frost *et al.* 1999). Differences in species composition between beds can often be accounted for by changes in a few common species rather than large-scale shifts or total species replacement. Further research into the dominant species in Irish seagrass beds is an essential next step in developing our understanding the contribution seagrass beds make to marine biodiversity in Ireland.

## 5. CONCLUSIONS

The ecological importance and highly diverse nature of *Zostera* habitats coupled with cheap and relatively non-destructive monitoring techniques makes them, at first glance, an attractive potential ecological monitoring tool. However, this preliminary study has shown natural fluctuations to be high and clear correlations between *Zostera* complexity and associated infaunal communities difficult to elucidate. Seasonal trends cannot be accurately identified without first implementing frequent, long term and intensive sampling to gain a comprehensive baseline data set. Giving due consideration to the ecosystem services provided by seagrass beds, it would be a worthwhile compromise to collect basic data on faunal communities and then concentrate on monitoring shoot density, maximum blade lengths and the distribution of a variety of beds that encompass a range of environmental conditions. By doing this, changes outside the natural fluctuations will be detected, giving an indication of when more in-depth monitoring protocols should be resumed. There is an urgent need to build on the current knowledge on Irish seagrass beds with frequent and long term monitoring as any further decline in *Zostera* habitats in Ireland would undoubtedly have an adverse affect on Irish marine biodiversity.

## 6. REFERENCES

- Anon. (1995). Habitat action plan for seagrass beds *in*: Biodiversity: The UK Steering Group Report – Volume II: Action Plans, December 1995, Tranches 1, Vol. 2. 262pp. Also available at [www.ukbap.org.uk/UKPlans.aspx?ID=35](http://www.ukbap.org.uk/UKPlans.aspx?ID=35).
- APG (1998). An ordinal classification for the families of flowering plants. *Annals of the Missouri Botanical Garden* 85: 531-553.
- Borg, J. A., Rowden A. A., Attrill, M. J., Schembri, P. J., Jones. M. B. (2006). Wanted dead or alive: high diversity of macroinvertebrates associated with living and 'dead' *Posidonia oceanica* matte. *Marine Biology* 149(3): 667-677.
- Borum, J., Duarte, C.M., Krause-Jensen, D., Greve, T. M. (2004). Preface. *In*: Borum, J., Duarte, C. M., Krause-Jensen, D. (eds.), *European Seagrasses: An Introduction to Monitoring and Management*. EU Project Monitoring and Managing of European Seagrasses (M&MS) EVK3-CT-2000-00044.
- Bowden, D. A., Rowden, A. A., Attrill, M. J. (2001). Effect of patch size and in-patch location on the infaunal macroinvertebrate assemblages of *Zostera marina* seagrass beds. *Journal of Experimental Biology and Ecology* 259: 133-154.
- Buchanan, J. B., Kain, J. M. (1971). Measurement of the physical and chemical environment: sediments. pp. 30 – 52. *In*: Holme, N. A., McIntyre, A. D. (eds.), *Methods for the study of marine benthos*. Blackwell Scientific Publications, Oxford. 387pp.
- Bulthuis, D. A., Woelkerling, Wm. J. (1983). Seasonal variation in standing crop, density and leaf growth rate of the seagrass *Heterozostera tasmanica*, in Western Port and Port Phillip Bay, Victoria, Australia. *Aquatic Botany* 16: 111-136.
- Burkholder, J. M., Mason, K. M., Glasgow, H. B. (1992). Water-column nitrate enrichment promotes decline of eelgrass *Z. marina*: evidence from seasonal mesocosm experiments. *Marine Ecology Progress Series* 81: 163-178.
- Creed, J. C. (1999). Distribution, seasonal abundance and shoot size of the seagrass *Halodule wrightii* near its southern limit at Rio de Janeiro state, Brazil. *Aquatic Botany* 65(1):47-58.
- Cullinane, J., O'Mahony, J., Whelan, P. (1985). Algal epiphytes of subtidal *Zostera marina* L. on the south coast of Ireland. *Cryptogamie: Algologie* VI 4:239-251.
- Davison D. M., Hughes D. J. (1998). *Zostera* biotopes (volume I): an overview of dynamics and sensitivity characteristics for conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project). 95pp.
- Fonseca, M.S. (1992). Restoring seagrass systems in the United States. pp79–110. *In*: Thayer, G. W. (eds.) *Restoring the nation's marine environment*. Maryland Sea Grant College, Maryland. 716pp.

## Management of *Zostera* beds

- Frost, M.T., Rowden, A. A., Attrill, M. J. (1999). Effect of habitat fragmentation on the macroinvertebrate infaunal communities associated with the seagrass *Zostera marina* L. *Aquatic Conservation: Marine and Freshwater Ecosystems* 9: 255-263.
- Granger, S., Iizumi, H. (2001). Water quality measurement methods for seagrass habitat. pp393-406. In: Short, F. T., Coles, R. G. (eds.), *Global Seagrass Research Methods*. Elsevier Science, Amsterdam. 482pp.
- Greve, T. M., Binzer, T. (2004). Which factors regulate seagrass growth and distribution? pp19-23. In: Borum, J., Duarte, C. M., Kause-Jensen, D. (eds.), *European Seagrasses: An Introduction to Monitoring and Management*. EU Project Monitoring and Managing of European Seagrasses (M&MS). 88pp.
- Hayward, P. J., Ryland, J. S. (eds.) (1995). *Handbook of the marine fauna of north-west Europe*. Oxford University Press. 800pp.
- Hemminga, M. A., Duarte, C. M. (2000). *Seagrass ecology*. Cambridge University Press. 298pp.
- Irish Meteorological Service (2006). [www.met.ie/marine/marine\\_climatology.asp](http://www.met.ie/marine/marine_climatology.asp). Accessed 5th December 2006.
- Kikuchi, T. (1980). Faunal relationships in temperate seagrass beds. pp153-172. In: Phillips, R. C., and McRoy, C. R. (eds.), *Handbook of seagrass biology: an ecosystem perspective*, Garland Publishing. 353pp.
- Lewis, F. G., Stoner, A. W. (1983). Distribution of macrofauna within seagrass beds: an explanation for patterns of abundance. *Bulletin of Marine Science* 33(2): 296-304.
- Madden, B., Jennings E., Jeffrey, D. W. (1993). Distribution and ecology of *Zostera* in Co. Dublin. *Irish Naturalists' Journal* 24: 303-310.
- Marbà, N., Cebrian, J., Enriquez, S., Duarte, C. M. (1996). Growth patterns of Western Mediterranean seagrasses: species-specific responses to seasonal forcing. *Marine Ecology Progress Series* 133: 203-215.
- Moore, K. A., Wetzel, R. L., Orth, R. J. (1997). Seasonal pulses of turbidity and their relations to eelgrass (*Zostera marina* L.) survival in an estuary. *Journal of Experimental Marine Biology and Ecology* 215(1): 115-134.
- Moore, K. A., Wetzel, R. L. (2000). Seasonal variation in eelgrass (*Zostera marina* L.) responses to nutrient enrichment and reduced light availability in experimental ecosystems. *Journal of Experimental Marine Biology and Ecology* 244: 1-28.
- Muñiz-Salazar, R., Talbot, S. L., Sage, G. K., Ward, D. H., Cabello-Pasini, A. (2005). Population genetic structure of annual and perennial populations of *Zostera marina* L. along the Pacific coast of Baja California and the Gulf of California. *Molecular Ecology* 14: 711-722.
- OSPAR Commission (2005). Case reports for the Initial List of Threatened and/or Declining Species in the OSPAR Maritime Area. OSPAR Commission. 149pp.
- Pergent-Martini, C., Pasqualini, V., Ferrat, L., Pergent, G., Fernandez, C. (2005). Seasonal dynamics of *Zostera noltii* Hornem. in two Mediterranean lagoons. *Hydrobiologia* 543: 233-243.
- Phillips, R. C., Meñez, E. G. (1988). *Seagrasses*. 104 pp. Smithsonian Contributions to the Marine Sciences No. 34. Smithsonian Institute, Washington DC. 104pp.
- Plymouth Marine Laboratory (2006). PRIMER-E software, Version 6.1.6. Plymouth Marine Lab, Plymouth, U.K.

## Management of *Zostera* beds

- Quammen M. L., Onuf, C. P. (1993). Laguna Madre: Seagrass changes continue decades after salinity reduction. *Estuaries* 16: 302-10.
- Rasmussen, E. (1977). The wasting disease of eelgrass (*Zostera marina*) and its effects on environmental factors and fauna, pp1-51. In: McRoy, C. P., Helfferich, C. (eds.) *Seagrass Ecosystems: A Scientific Perspective*. Marcel Dekker, Inc. New York. 314pp.
- Robinson, J. A. (2003). All-Ireland review of intertidal eel-grass (*Zostera*) beds. Unpublished report to the Heritage Council. 43pp.
- Sand-Jensen, K. (1977). Effect of epiphytes on eelgrass photosynthesis. *Aquatic Botany* 3: 55-56.
- Somerfield, P. J., Yodnarasri, S., Aryuthaka, C. (2002). Relationships between seagrass biodiversity and infaunal communities: implications for studies of biodiversity effects. *Marine Ecology Progress Series* 237: 97-109.
- Spalding, M., Taylor, M., Ravilious, C., Short, F., Green, E. (2003). The distribution and status of seagrasses, pp5-26. In: Spalding, M., Taylor, M., Ravilious, C., Short, F., Green, E. (eds.), *World Atlas Of Seagrasses*. University of California Press. 298pp.
- SPSS for Windows, Rel. 12.0.1. (2003). Chicago: SPSS Inc.
- Tennant, G. (2006). Experimental effects of ammonium on eelgrass (*Zostera marina* L.) shoot density in Humboldt Bay, California. Masters Thesis. Humboldt State University. 86pp.
- van Katwijk, M. M., Vergeer, L. H. T., Schmitz, G. H. W., Roelofs, J. G. M. (1997). Ammonium toxicity in eelgrass *Zostera marina*. *Marine Ecology Progress Series* 157: 159-173.
- Whelan, P. M. (1986). The Genus *Zostera* in Ireland. Ph.D Thesis, University College Cork, Ireland. 398pp.
- Whelan, P. M., Cullinane, J. P. (1985). The algal flora of a subtidal *Zostera* bed in Ventry Bay, South-west Ireland. *Aquatic Botany* 23: 41-51.

## 7. APPENDICES

**Infauna Species List**

A total of 124 species representing 81 families from six phyla were recorded from Barloge Creek (1) and Ventry Bay (2) study sites over the course of the study.

Phylum	Class	Order	Family	Species
Arthropoda	Pycnogonida	Pantopoda	Ammotheidae	Genus <i>Achelia</i> spp. 1 <sup>(2)</sup> Genus <i>Achelia</i> spp. 2 <sup>(1,2)</sup>
			Anoplodactylidae	<i>Anoplodactylus petiolatus</i> <sup>(1,2)</sup>
			Endeidae	<i>Endeis laevis</i> <sup>(1)</sup>
			Pallenidae	<i>Callipallene emaciata</i> <sup>(2)</sup>
Cnidaria	Anthozoa	Actiniaria	Actiniidae	Actiniidae sp. <sup>(1,2)</sup>
	Scyphozoa	Stauromedusae	Lucernariidae	<i>Halicylistus salpinx</i> <sup>(2)</sup> <i>Lucernariopsis cruxmelitensis</i> <sup>(2)</sup>
Crustacea	Copepoda	Calanoida	Temoridae	<i>Temora longicornis</i> <sup>(2)</sup>
		Harpacticoida	Porcellidiidae	<i>Porcellidium sarsi</i> <sup>(2)</sup>
	Malacostraca	Poecilostomatoida	Clausidiidae	Genus <i>Hemicyclops</i> spp. 1 <sup>(1)</sup>
		Amphipoda	Caprellidae	<i>Caprella acanthifera</i> <sup>(1,2)</sup> <i>Pariambus typicus</i> <sup>(2)</sup>
			Ampeliscidae	<i>Ampelisca diadema</i> <sup>(2)</sup> <i>Ampelisca brevicornis</i> <sup>(2)</sup> <i>Photis longicandata</i> <sup>(2)</sup>
			Ampithoidae	<i>Ampithoe ramondi</i> <sup>(2)</sup> <i>Ampithoe rubricata</i> <sup>(1,2)</sup> <i>Ampithoe</i> sp. (juvenile) <sup>(2)</sup>
			Amphiloichidae	<i>Amphiloichus neapolitamis</i> <sup>(1)</sup>
			Aoridae	<i>Leptocheirus pectinatus</i> <sup>(1)</sup> <i>Aora gracilis</i> <sup>(2)</sup> <i>Aoridae</i> sp. (juvenile) <sup>(1)</sup>
			Atylidae	<i>Atylus swammerdamei</i> (juv.) <sup>(1,2)</sup>
			Calliopiidae	<i>Apherusa bispinosa</i> <sup>(1,2)</sup>
			Cheluridae	<i>Erichthonius punctatus</i> <sup>(1,2)</sup>
			Corophidae	<i>Corophium sextonae</i> <sup>(1,2)</sup>
			Dexaminidae	<i>Dexamine thea</i> <sup>(1,2)</sup> <i>Dexamine spinosa</i> <sup>(1,2)</sup>
			Gammaridae	<i>Gammarus locusta</i> <sup>(2)</sup>
			Haustoriidae	<i>Urothe elegans</i> <sup>(2)</sup>
			Isaeidae	<i>Gammaropsis maculosa</i> <sup>(1)</sup> <i>Microdeutopus anomalis</i> <sup>(1,2)</sup> <i>Microdeutopus versiculatus</i> <sup>(1,2)</sup> <i>Microprotopus maculosus</i> <sup>(1,2)</sup>
			Liljeborgiidae	<i>Liljeboria kinahani</i> <sup>(1,2)</sup>
			Lysianassidae	<i>Socarnes erythrophthalmus</i> <sup>(1)</sup>
			Melitidae	<i>Abludomelita obtusata</i> <sup>(1,2)</sup> <i>Ceradocus semiserratus</i> <sup>(1)</sup> <i>Cheirocratus sundevallii</i> <sup>(1,2)</sup> Genus <i>Cheirocratus</i> sp. <sup>(1)</sup>
			Oedicerotidae	<i>Gammarella fucicola</i> <sup>(1,2)</sup> <i>Periocolodes longimanus</i> <sup>(1,2)</sup>
			Phoxocephalidae	<i>Pontocrates altamarinus</i> <sup>(2)</sup>
			Stenothoidae	<i>Harpinia crenulata</i> <sup>(1,2)</sup> <i>Parametaphoxus fultoni</i> <sup>(1)</sup> <i>Stenothoe monoculoides</i> <sup>(1,2)</sup>

Management of *Zostera* beds

Phylum	Class	Order	Family	Species	
Crustacea	Malacostraca	Cumacea	Bodotriidae	<i>Bodotria scorpioides</i> <sup>(2)</sup>	
			Diastylidae	<i>Diastylis rugosa</i> <sup>(2)</sup>	
			Leuconidae	<i>Eudorella truncatula</i> <sup>(2)</sup>	
		Decapoda	Cancridae	<i>Cancer pagurus</i> (juvenile) <sup>(1)</sup>	
			Hippolytidae	<i>Hippolyte varians</i> <sup>(1)</sup>	
				<i>Thoralus cranchii</i> <sup>(1)</sup>	
			Majidae	<i>Eurynome spinosa</i> <sup>(2)</sup>	
			Paguridae	<i>Anapagurus hyndmanni</i> <sup>(1)</sup>	
				<i>Pagurus bernhardus</i> <sup>(1)</sup>	
				<i>Pagurus pubescens</i> <sup>(1)</sup>	
			Palemonidae	<i>Palemon</i> sp. (juvenile) <sup>(2)</sup>	
			Porcellanidae	<i>Pisidia longicornis</i> <sup>(1)</sup>	
			Portunidae	<i>Carcinus maenas</i> <sup>(2)</sup>	
				<i>Liocarcinus arcuatus</i> <sup>(1)</sup>	
				<i>Liocarcinus corrugatus</i> <sup>(1)</sup>	
				Processidae	<i>Processa edulis crassipes</i> <sup>(1,2)</sup>
				Xanthidae	<i>Xantho incisus</i> <sup>(1)</sup>
		<i>Xantho pilipes</i> <sup>(1)</sup>			
		Isopoda		n/a	shrimp larvae <sup>(1)</sup>
			Arcturidae	<i>Astacilla longicornis</i> <sup>(2)</sup>	
			Cirolanidae	<i>Eurydice spinigera</i> <sup>(2)</sup>	
			Idoteidae	<i>Idotea baltica</i> <sup>(2)</sup>	
			Janiridae	<i>Janira maculosa</i> <sup>(1,2)</sup>	
			Gnathiidae	Gnathiidae sp. (juvenile) <sup>(1)</sup>	
			Leptostraca	Nebaliidae	Nebaliidae sp. <sup>(1,2)</sup>
				Mysidacea	Mysidae
			n/a		Mysidacea sp. <sup>(2)</sup>
Tanaidacea	Mysidae		<i>Siriella clausii</i> <sup>(2)</sup>		
	Apseudidae	<i>Apseudes talpa</i> <sup>(1)</sup>			
	Paratanaidae	<i>Leptochelia savignyi</i> <sup>(1,2)</sup>			
Ostracoda	Myodocopidea	Cylindroleberididae	<i>Synasterope norvegica</i> <sup>(2)</sup>		
		Philomedidae	<i>Euphilomedes interpuncta</i> <sup>(2)</sup>		
Ctenophora	Tentaculata	Cydippida	Pleurobrachiidae	<i>Pleurobranchia pileus</i> <sup>(2)</sup>	
Echinodermata	Asteroidea	Phanerozonia	Astropectinidae	<i>Astropecten irregularis</i> <sup>(2)</sup>	
	Echinoidea	Spatangoida	Loveniidae	<i>Echinocardium flavescens</i> <sup>(2)</sup>	
	Ophiuroidea	Ophiurae	Amphiuridae	<i>Amphiura brachiata</i> <sup>(1,2)</sup>	
				<i>Amphipholis chiajei</i> <sup>(1,2)</sup>	
				Amphiuridae spp. 1 <sup>(2)</sup>	
			Ophiolepidae	<i>Ophiura ophiura</i> <sup>(1,2)</sup>	
Ophiotrichidae	<i>Ophiotrix fragilis</i> <sup>(1)</sup>				

Management of *Zostera* beds

Phylum	Class	Order	Family	Species	
Mollusca	Bivalva	Myoida	Corbulidae	<i>Corbula gibba</i> <sup>(1,2)</sup>	
			Hiatellidae	<i>Hiatella arctica</i> <sup>(1,2)</sup>	
		Mytiloida	Mytilidae	<i>Modiolula phaseolina</i> <sup>(1,2)</sup>	
				<i>Musculus costulatus</i> <sup>(1,2)</sup>	
		Mytiloida		<i>Mytilus edulis</i> <sup>(1,2)</sup>	
		Nuculoida	Nuculidae	<i>Nucula nitidosa</i> <sup>(1,2)</sup>	
		Ostreoida	Anomiidae	<i>Anomia ephippium</i> <sup>(1)</sup>	
			Pectinidae	<i>Chlamys varia</i> <sup>(1)</sup>	
		Pholadomyoida	Thraciidae	<i>Thracia phaseolina</i> <sup>(2)</sup>	
		Tellinoidea	Donacidae	<i>Moerella donacina</i> <sup>(1,2)</sup>	
			Psammbiidae	<i>Gari ferrensis</i> <sup>(2)</sup>	
		Veneroida		Tellinidae	<i>Angulus squalidus</i> <sup>(2)</sup>
				Cardiidae	<i>Parvicardium ovale</i> <sup>(1,2)</sup>
				Lasaeidae	<i>Lasaea rubra</i> <sup>(1,2)</sup>
				Leptonidae	<i>Lepton squamosum</i> <sup>(2)</sup>
					<i>Mysella bidentata</i> <sup>(1,2)</sup>
				Lucinidae	<i>Lucinoma borealis</i> <sup>(1,2)</sup>
				Mactridae	Mactridae spp. 1 <sup>(1,2)</sup>
				Montacutidae	<i>Tellimya ferruginosa</i> <sup>(2)</sup>
				Scrobiculariidae	<i>Abra alba</i> <sup>(1,2)</sup>
					<i>Phaxas pellucidus</i> <sup>(2)</sup>
				Solenidae	<i>Solen marginatus</i> <sup>(2)</sup>
					<i>Thyasira flexuosa</i> <sup>(1,2)</sup>
				Veneridae	<i>Chamelea gallina</i> <sup>(2)</sup>
					<i>Circomphalus casina</i> <sup>(1,2)</sup>
		<i>Dosinia exolata</i> <sup>(2)</sup>			
	<i>Tapes rhomboides</i> <sup>(1,2)</sup>				
	Veneridae spp. 1 <sup>(1,2)</sup>				
n/a	n/a	Bivalve spp. 1 <sup>(1)</sup>			
		Bivalve spp. 2 <sup>(1,2)</sup>			
		Bivalve spp. 3 <sup>(1)</sup>			
		Bivalve spp. 4 <sup>(1)</sup>			
		Bivalve spp. 5 <sup>(2)</sup>			
		Bivalve spp. 6 <sup>(2)</sup>			
		Bivalve spp. 7 <sup>(2)</sup>			
Polyplacophora	Acanthochitonina	Acanthochitonidae	<i>Acanthochitona crinitus</i> <sup>(1)</sup>		
	Lepidopleuridae	Lepidopleuridae	<i>Leptochiton cancellatus</i> <sup>(1,2)</sup>		

**Epifauna Species List**

*Zostera* shoots collected for epifauna were enclosed in labelled plastic bags and gathered at the base before cutting 2 cm above the substrate to avoid sediment disturbance. In the laboratory shoot blade and water contained in the collection bag were sieved through a 63 µm sieve to obtain fauna. Organisms were identified to the lowest possible taxonomic level and stored in 70 % alcohol. Samples were collected monthly for one year (June 2004 – May 2005) at Barloge Creek and in August, December 2004 and April 2005 at Ventry Bay.

Phylum	Class	Order	Family	Species	
Annelida	Polychaeta	n/a	n/a	Polychaete	
	Oligochaeta	n/a	n/a	Oligochaete	
Arthropoda	Pycnogonida	Pantopoda	Ammonotheidae	Genus <i>Achelia</i> spp. 1 Ammonotheidae sp.	
		n/a	Endeidae n/a	<i>Endeis laevis</i> Pycnogonida spp. 1	
Chelicerata	Arachnida	Acari	n/a	Acari sp.	
		Araneae	n/a	Araneae sp.	
Cnidaria	Anthozoa	Actinaria	Actiniidae	<i>Anemonia viridis</i>	
	Scphozoa	Stauromedusae	Eleutherocarpidae Lucernariidae	Genus <i>Halicyclystus</i> sp. <i>Lucernariopsis cruxmelitensis</i>	
Crustacea	Cirripedia	n/a	n/a	cyprid larvae	
		Copepoda	Calanoida	n/a	Calanoid copepods
	Harpacticoida		n/a	Harpacticoid copepods	
	Malacostraca	Amphipoda	n/a	n/a	Caprellid amphipod
			n/a	n/a	Gammarid amphipods
		Cumacea	Bodotriidae	<i>Bodotria scorpioides</i>	
			Diastylidae	<i>Diastylis rugosa</i>	
		Decapoda	Majidae	<i>Macropodia rostrata</i>	
			Palaemonidae	Palaemonidae sp. (juvenile)	
		Leptostraca	Nebaliidae	Nebaliidae sp.	
		Tanaidacea	n/a	Tanaidacea sp.	
	Ostracoda	Myodocopidea	Philomedidae	<i>Euphilomedes interpuncta</i> <i>Synasterope norvegica</i>	
		n/a	n/a	Ostracoda spp. 1	
n/a		n/a	Ostracoda spp. 2		
n/a		n/a	Ostracoda spp. 3		
n/a		n/a	Ostracoda spp. 4		
Echinodermata	Echinoidea	n/a	n/a	Echinoidea sp. (juvenile)	
	Ophiuroidea	Ophiurae	Amphiuridae	Amphiuridae spp. 1	
			Ophiolepidae	<i>Ophiura ophiura</i>	
		n/a	Ophiurae sp.		
Ascidacea	Pleurogona	n/a	Pleurogona sp.		

Management of *Zostera* beds

Phylum	Class	Order	Family	Species	
Mollusca	Bivalvia	Myoidea	Hiatellidae	<i>Hiatella arctica</i>	
		Mytiloidea	Mytilidae	<i>Modiolula phaseolina</i>	
				<i>Musculus costulatus</i>	
		Nuculoidea	Nuculidae	<i>Mytilus edulis</i>	
				<i>Nucula nitidosa</i>	
		Ostreoida	Anomiidae	<i>Anomia ephippium</i>	
		Veneroidea	Cardiidae	Cardiidae sp. (juvenile)	
			Leptonidae	<i>Mysella bidentata</i>	
		n/a	n/a	Bivalve spp. 1 (juvenile)	
		n/a	n/a	Bivalve spp. 2 (juvenile)	
		n/a	n/a	Bivalve spp. 3 (juvenile)	
		Gastropoda	Archaeogastropoda	Patellidae	<i>Helicon pellucidum</i>
					Patellidae sp.
	Rissoidae				Rissoidae sp. (juvenile)
	Trochidae				<i>Gibbula cineraria</i>
	Cephalaspidea		Retusidae	<i>Bittium reticulatum</i>	
				<i>Retusa truncatula</i>	
				n/a	Gastropod spp. 1 (juvenile)
				n/a	Gastropod spp. 2 (juvenile)
				n/a	Gastropod spp. 3 (juvenile)
				n/a	Gastropod spp. 4 (juvenile)
				n/a	Gastropod spp. 5 (juvenile)
				n/a	Gastropod spp. 6 (juvenile)
				n/a	Gastropod spp. 7 (juvenile)
				n/a	Gastropod spp. 8 (juvenile)
	n/a	Gastropod spp. 9 (juvenile)			
	n/a	Gastropod spp. 10 (juvenile)			
n/a	Gastropod spp. 11 (juvenile)				
n/a	Gastropod spp. 12				
n/a	Gastropod spp. 13				
Nudibranchia	Dotoidae	<i>Doto</i> sp.			
	Eubranichidae	<i>Eubranichus farrani</i>			
		<i>Eubranichus pallidus</i>			
	n/a	Nudibranchia sp.			
	n/a	Nudibranchia spp. 1			
n/a	Nudibranchia spp. 2				
Platyhelminthes	Turbellaria	Polycladida	Leptoplanidae	Leptoplanidae sp.	
		n/a	Monocelidae	<i>Monocelis fusca</i>	
			n/a	n/a	Turbellaria spp. 1