

ORIGINAL ARTICLE

Effect of observation method on the perception of community structure and water quality in a brackish water ecosystem

Tiia Möller^{1,2}, Jonne Kotta¹ & Georg Martin¹

¹ Estonian Marine Institute, University of Tartu, Tallinn, Estonia

² Institute of Ecology and Earth Sciences, University of Tartu, Tartu, Estonia

Keywords

Baltic sea; macrophytes; benthic invertebrates; scale; water quality.

Correspondence

Tiia Möller, Estonian Marine Institute, University of Tartu, Mäealuse 10a, 12618 Tallinn, Estonia. E-mail: tiia.moller@sea.ee

Conflicts of interest

The authors declare no conflicts of interest.

doi:10.1111/j.1439-0485.2009.00325.x

Abstract

The EU Water Framework Directive is a Community legislative instrument in the field of environmental protection that establishes a common framework for keeping water quality at a favourable level. To implement the directive, classification systems need to be established that allow detection of human impacts at early stages and, thus, more effective management of coastal communities. Due to the spatial variability of communities, however, the results of any assessment are highly dependent on the selection of data. In this study we identified local spatial scales in which variability of macrophyte communities was maximised, quantified links between observed patterns of sediment types and communities and estimated how selection criteria impacted the outcome of the assessment of indicator class value in four different communities of the Northern Baltic Sea. The main findings of the study were that: (i) there were no clear local spatial scales in which the variability of benthic communities was maximised; (ii) hard-bottom communities were better predicted by the spatial arrangement of sediment characteristics than soft-bottom communities; (iii) the selection of method had no effect on the estimates of macrophyte cover and indicator class; but (iv) method impacted independently of habitat type on error estimates of macrophyte cover and indicator class. To conclude, in such homogeneous and low diversity macrophyte communities it is preferable to use methods that result in lower error estimates of algal coverage and, thus, result in lower uncertainties of estimates in the water quality class.

Problem

The EU Water Framework Directive is a Community legislative instrument in the field of environment protection that establishes a common framework for keeping water quality at a favourable level. To implement the directive, classification systems need to be established that allow us to assign each waterbody one of five ecological quality classes. This classification system has to be developed using mainly biological indicators (Water Quality Elements). Submerged Aquatic Vegetation is one of those Quality Elements and relevant indicators have to be

developed and intercalibrated to be useable in the classification system (Anonymous 2000). The choice of indicators and assessment method depends on the spatial patterns of the pressures and communities involved.

To identify the most important governing factors one needs to determine the scales at which communities have the largest variability and where the links between environmental and biotic patterns are the strongest (Platt & Denman 1975; Steele & Henderson 1994). Benthic communities have high structural variability at a multitude of scales and this variability is closely linked with physical setting. Therefore the mapping studies should incorporate

the relevant scales of variability to understand factors and processes generating patterns in biotic and abiotic components of ecosystems (Menge & Olson 1990; Levin 1992; Karlson & Cornell 1998). To quantify marine biodiversity, conserve marine ecosystems and assess impacts of human activities, understanding the spatial distributions of biota at local scales (10s or 100s of metres) is important for coastal management (Kotta *et al.* 2008c).

To date a number of quantitative methods have been developed to map benthic macrophyte communities at the local scale. Except for one study (Leujak & Ormond 2007), earlier studies have failed to provide any clear guidance as to which method is to be preferred. With the introduction of digital video cameras with greatly improved resolution, filming and assessing benthic communities along transects has become an increasingly practical and popular method (Murdoch & Aronson 1999; Riegl *et al.* 2001; Houk & Van Woessik 2006; Bucas *et al.* 2007). In general, the accuracy and precision of methods is dependent on the investment of resources but the resources are generally limited or fixed. Sample size is often the dominant factor in determining precision. If the sample size is too low, the study will lack the precision to provide reliable answers to the questions it is investigating. If the sample size is too large, time and resources will be wasted, often for minimal gain. Absolute accuracy and precision are not our ultimate goal; rather, an analysis of how data selection affects the outcome of water quality assessment is needed. Nevertheless, it is essential that the preferred method allows detection of already small changes in community structure (*i.e.* has low error estimate) and therefore can be used by the early warning systems.

In this study we: (i) attempted to identify local spatial scales at which the variability of benthic communities is maximised; (ii) quantified links between observed patterns of sediment types and communities; and (iii) estimated whether and how the criteria of data selection impacted the outcome of the assessment of indicator class value in four different communities of the Northern Baltic Sea. Our hypotheses were as follows: (i) the spatial patterns of macrophyte communities strongly reflect the patterns of seabed sediment, (ii) the communities that are more heterogeneous are more sensitive to the criteria of data selection and (iii) random sampling should produce better estimates of distribution of key species and communities as compared to uniform sampling.

Material and Methods

The study was carried out in four shallow water communities of the Eastern Baltic Sea in summer 2007 (Fig. 1). The Baltic Sea is a geologically young semi-enclosed sea. There exists a strong salinity gradient

from west to east. Sand, silt and clay sediment prevail and hard substrata are usually found in shallow and exposed bays. Due to the prevalence of clay substrate and strong wave action, the water transparency is often very low, while the low salinity and strong seasonality in temperature and light conditions result in the number of aquatic plant species inhabiting the Northern Baltic Sea being small (Kotta *et al.* 2008a,b).

Using a remote underwater video device we estimated the spatial patterns of benthic vegetation and invertebrates in four different communities of the Northern Baltic Sea. The camera was set at an angle of 35° below horizontal to maximise the field of view and the range of the forward view was about 2 m in clear waters. The camera sled was towed from a 5-m-long boat 1 m above the sea floor at an average speed of 50 cm·s⁻¹. Real-time video was captured with a digital video recorder. Depth and navigational data (from GPS) were recorded at 1-s intervals during camera deployments.

The four different communities investigated were: (i) bladder wrack *Fucus vesiculosus*, (ii) the red alga *Furcellaria lumbricalis*, (iii) the eelgrass *Zostera marina*, (iv) other higher plants (*Potamogeton* spp., *Myriophyllum spicatum*). The former two communities rely on hard substrate and the latter two on soft substrate.

For each community five transects of 87.5 m length were videotaped. In each transect the coverage of different sediment type (rock, boulders, pebbles, gravel, sand), benthic macrophyte and invertebrate species were estimated (i) in a continuous video mode and (ii) in a still picture mode based on 35 photographs taken with a step of 2.5 m. Additionally, the coverages were also estimated based on 5, 10 and 20 photographs selected using either uniform or random sampling of the respective pool of 35 photographs. The coverages were then averaged over transect and selection criteria, resulting in five replicate cover estimates of each community and selection criteria. These replicates were used later in statistical data analyses. A mean of sediment, algal and invertebrate coverages of all frames provided a baseline within the video technique when estimating the precision of different methods (Weinberg 1981; Leujak & Ormond 2007).

The ratios of annual to perennial macroalgae and of annual algae to *Z. marina* are potential indicators of eutrophication, as high nutrient concentrations are known to favour the growth of ephemeral flora (Sand-Jensen & Borum 1991; Kotta *et al.* 2000; Pedersen & Snoeijs 2001). The group of annual algae often includes ephemeral species such as *Chaetomorpha linum*, *Cladophora glomerata*, *Pilayella littoralis* and *Ectocarpus* sp. and sheet-formed species such as *Ulva lactuca*. Typically, they have thin tissues capable of fast rates of nutrient uptake and growth; they are relatively short-lived and represent

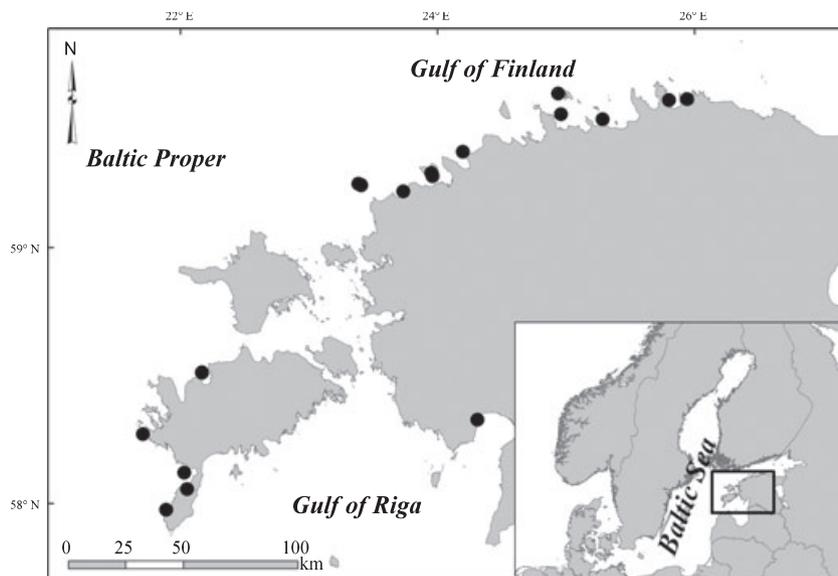


Fig. 1. Map of the studied transects in the NE Baltic Sea.

the so-called r-strategy of population growth. In contrast to the annual species, perennial species are slow-growing, often have thicker and more complex tissues and are represented by, for example, the genera *Fucus*, *Furcellaria* and *Laminaria*. Although the division into annual and perennial types is straightforward in many cases, there are some species that do not fit this classification scheme. Nevertheless, this indicator is used in several schemes of water quality assessment, especially in cases where more traditional indicators (e.g. depth penetration of vegetation) is not applicable due to substratum limitation or shallowness of coastal areas. In the Mediterranean area this indicator is used either independently or is incorporated into different index calculations (e.g. Orfanidis *et al.* 2001; Juanes *et al.* 2008). The indicator is often calculated as the share of perennial (or annual) species to total vegetation cover or biomass. In Estonia the indicator is calculated as the coverage of species and divided into five classes: high (share of perennial macroalgae >72%), good (72–45%), moderate (45–27%), poor (27–9%) and bad (<9%) (Anonymous 2008).

Factorial ANOVA (Statistica version 8.0) and ANOSIM (PRIMER version 6.1) analyses were used to test for significant differences in the coverage of key species and communities among observation methods. Dissimilarities between each pair of algal and invertebrate cover estimates along transects were calculated using a zero-adjusted Bray–Curtis coefficient and dissimilarities between each pair of sediment cover estimates along transects were calculated using Euclidean distance. The scatterplot between the obtained dissimilarity matrixes of coverages and the respective point distances of samples was used as a proxy of the scale-specific spatial variability in sediment characteristics and community cover.

Results

Altogether, 16 taxa were identified. Most species were observed in *Zostera marina* (9) and *F. vesiculosus* (8) community. The mean coverage of community-forming species (i.e. *Z. marina*, other higher plants, *F. vesiculosus*, *F. lumbricalis*) exceeded 40% (Table 1).

The scatterplots between the point distances of samples and dissimilarity matrixes of coverage (i.e. community variability) show that there were no clear local spatial scales in which the variability of benthic communities was maximised (Fig. 2). Hard bottom communities were less patchy compared to soft bottom communities. *Fucus vesiculosus* and *F. lumbricalis* communities tended to be relatively homogeneous through the range of the studied spatial scales. The spatial pattern of *Z. marina* and other higher plant communities largely varied along transects. These scatterplots also show that some transects had the largest variability at 20 or 50 m (indicating that the patch size of macrophyte communities was on average either 20 or 50 m). The variability of other communities either increased with spatial scale or, similarly to hard-bottom communities, did not vary with spatial scale (Fig. 2).

Changes in sediment characteristics largely explained the spatial arrangement of *F. vesiculosus* and *F. lumbricalis* communities on hard bottom. However, *Z. marina* and other higher plant communities on soft bottoms were weakly related to the spatial patterns of sediment characteristics (Fig. 2).

Different sampling frequencies and distributions had no effect on the mean estimates of the key species (Fig. 3), community coverage and the indicator class (ANOVA and ANOSIM $P > 0.05$) (Table 2). However, different sampling frequencies and distributions produced

Table 1. Average coverage of benthic macrophyte and invertebrate species found in the investigated communities.

taxa	community			
	<i>Fucus vesiculosus</i>	<i>Furcellaria lumbricalis</i>	<i>Zostera marina</i>	higher plants
<i>Ceramium tenuicorne</i> (Kützinger) Waern	1.94	–	–	–
<i>Chara</i> sp.	–	–	–	3.94
<i>Chorda filum</i> (Linnaeus) Stackhouse	0.20	–	0.02	–
<i>Cladophora glomerata</i> (Linnaeus) Kützinger	11.31	–	0.03	–
<i>Cladosiphon zosterae</i> (J. Agardh) Kylin	–	–	2.75	–
<i>Fucus vesiculosus</i> Linnaeus	56.31	–	0.81	0.56
<i>Furcellaria lumbricalis</i> (Hudson) J.V. Lamouroux	–	43.51	–	–
<i>Myriophyllum spicatum</i> Linnaeus	–	–	–	11.07
<i>Pilayella littoralis</i> (Linnaeus) Kjellman	32.94	1.78	42.21	38.04
<i>Polysiphonia</i> sp.	–	38.81	–	–
<i>Potamogeton pectinatus</i> Linnaeus	–	–	4.48	41.19
<i>Potamogeton perfoliatus</i> Linnaeus	0.01	–	2.38	10.61
<i>Sphacelaria arctica</i> Harvey	–	0.37	–	–
<i>Zostera marina</i> Linnaeus	1.12	–	42.12	–
<i>Balanus improvisus</i> Darwin	1.04	3.76	–	–
<i>Mytilus trossulus</i> Gould	–	0.12	0.69	–

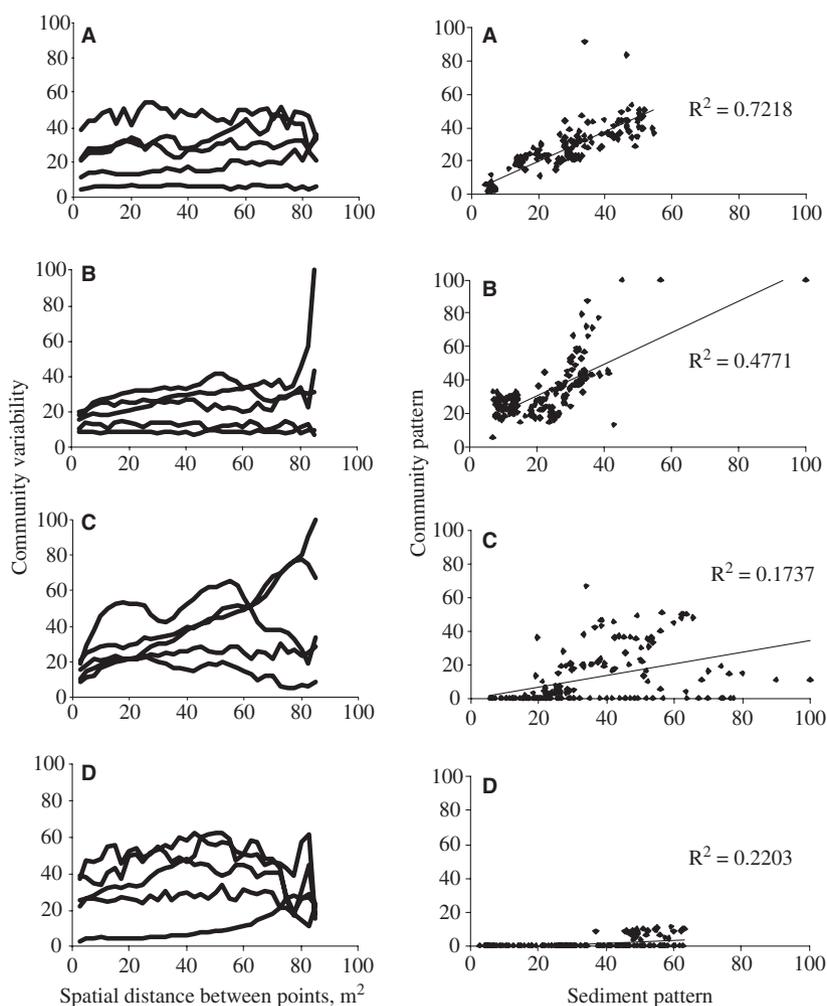


Fig. 2. Left column: variability of the macrophyte communities (Bray–Curtis dissimilarities) through the range of studied spatial scales. Right column: scatterplots of the dissimilarities of sediment characteristics and benthic macrophyte communities assessed separately for each pair of samples in a transect; R, linear correlation coefficient between the studied variables. Community code is as follows: A, *Fucus vesiculosus*; B, *Furcellaria lumbricalis*; C, *Zostera marina*; D, other higher plants.

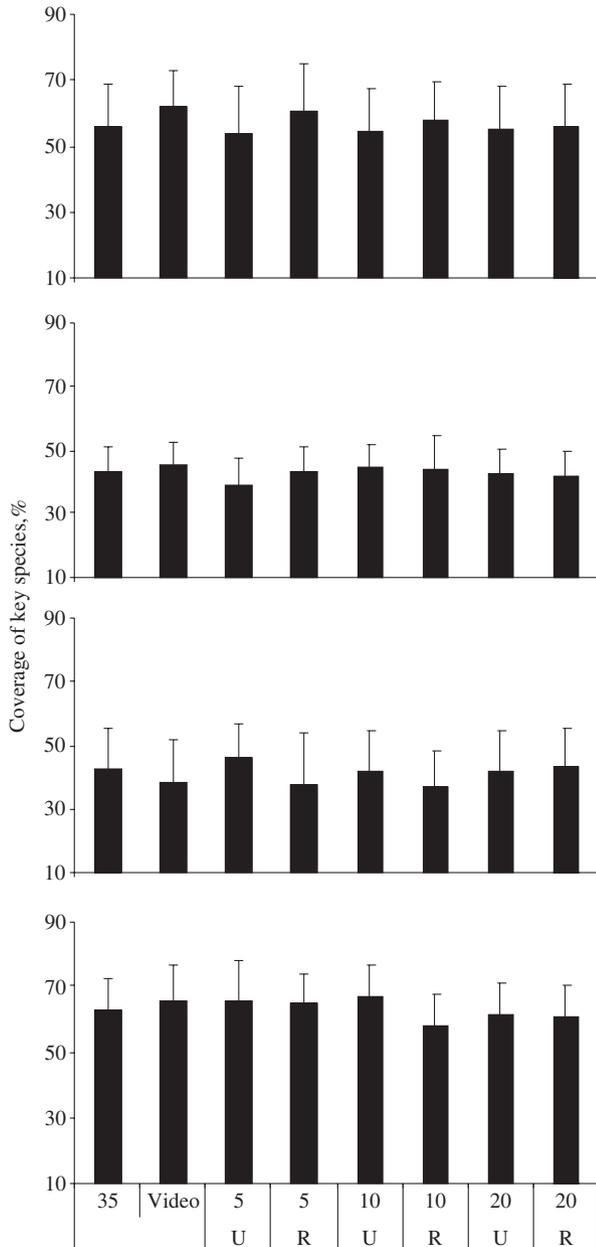


Fig. 3. Average coverage of key species (%) with SE according to different sampling frequencies and distributions applied. Treatments: video, estimates based on continuous video, 5, 10, 20 and 35 indicate the number of photographs included; U, uniform sampling; R, random sampling. Community code is as follows: A, *Fucus vesiculosus*; B, *Furcellaria lumbricalis*; C, *Zostera marina*; D, higher plants.

different error estimates. The cover estimates in a still picture mode based on 35 or 20 photographs had significantly lower error compared to other sampling rates. Similarly, the indicator class estimates in a continuous video mode and in a still picture mode based on 35 photographs had significantly lower error compared to other sampling rates (Fig. 4).

Table 2. Summary of the factorial anova analysis of the effect of sampling methods and community types on the evaluation of water indicator classes and key species coverage. Significant effects are indicated in bold.

effect	SS	DF	MS	F	P
key species coverage					
method	893	7	128	0.31	0.948
habitat	466	3	155	0.38	0.770
method × habitat	9305	21	443	1.08	0.381
key species coverage (absolute values)					
method	8803	7	1258	6.41	0.000
habitat	2260	3	753	3.84	0.011
method × habitat	4120	21	196	1.00	0.467
indicator classes					
method	287	7	41	0.27	0.964
habitat	951	3	317	2.09	0.104
method × habitat	1499	21	71	0.47	0.976
indicator classes (absolute values)					
method	2024	7	289	3.54	0.002
habitat	508	3	169	2.08	0.106
method × habitat	908	21	43	0.53	0.953

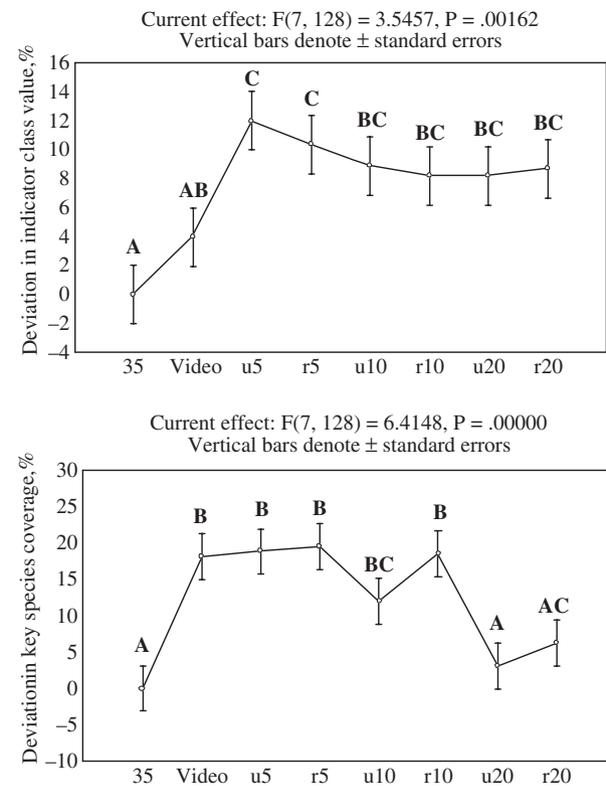


Fig. 4. Deviation in the estimates of indicator class and key species coverage for different sampling frequencies and distributions applied. A, B, C indicate different homogeneous groups (Fisher LSD *post-hoc* test).

Discussion

The main findings of the study were that (i) there were no clear local spatial scales in which the variability of benthic communities was maximised; (ii) hard-bottom communities were better predicted by the spatial arrangement of sediment characteristics than soft bottom communities; (iii) the selection of method had no effect on the estimates of macrophyte cover and indicator class, but (iv) impacted independently of community the error estimates of macrophyte cover and indicator class.

In the Northern Baltic Sea bottom substrate is very heterogeneous and often characterised by mixed sediments. Different substrate types are found either within very small areas (e.g. patches of sand or stones that do not exceed 50–200 cm in diameter) or, alternatively, different sediment types are mixed at the same location (e.g. mixture of clay, sand and pebbles) (Kotta *et al.* 2008a,b). Our results suggest that the cover values of true hard-bottom species *i.e.* *F. vesiculosus* and *F. lumbricalis* clearly reflected such sediment heterogeneity. By far the strongest correlations between the cover of different substrate types and the patterns of macrophyte communities were observed in *F. vesiculosus* community. The availability of light and mineral nutrients is known to play a crucial role, with upwelling, turbulence, turbidity and grazing intensity as secondary factors in regulating the distribution of macroalgae (Field *et al.* 1998). The availability of hard substrate is a prerequisite for the establishment of the macroalgal species found in the study area and other limiting factors are known to have minor effects on the macroalgal communities in the study area (Kotta & Orav 2001; Eriksson & Bergström 2005; Kotta *et al.* 2006; Orav-Kotta *et al.* 2009).

On the other hand, there was only a weak link between sediment heterogeneity and soft-bottom macrophyte communities. It is likely that wave exposure rather than sediment characteristics determine the distribution of true soft-bottom macrophyte species. Large waves may cause considerable resuspension of sediments and result in prolonged periods of poor light conditions (Madsen *et al.* 2001) but, similarly, they may exert a mechanical disturbance of the benthic communities (Dernie *et al.* 2003; Kotta *et al.* 2007; Schiel & Lilley 2007).

It is plausible that the weak effects of the choice of methods can be attributed to the homogeneous distribution of benthic macrophyte communities in the study area and to the small range of scales investigated. Only a few transects on soft sediments were characterised by the patchy distribution of the key macrophyte species. Providing the low diversity of macrophyte communities in the study area (Kotta *et al.* 2008a,b) already, a few observations may capture the pattern of macrophyte

communities and provide accurate estimates of their parameters.

Earlier studies have shown that the variance of estimates decreases (and precision increases) with increasing sample size when sampling units are scaled up to standard units for comparison (Elliot 1979; Morin 1985). Similarly, in our study the precision of the estimate increased with sample size. Besides, the precision of different sampling frequencies and distributions varied depending on the variable considered (cover of key species *versus* indicator class value). Observation that was based on a continuous video mode resulted in large variability of estimates of the coverage of dominant macroalgal species (*F. vesiculosus*, *F. lumbricalis* and higher plants) compared to still picture mode. On the other hand, both video and still picture mode precisely captured the cover patterns of annual and perennial algae. This may be due to more clumped distribution (reflected in the greater variance of estimates) of the dominant species in relation to the broader categories such as perennial and annual species. In such macrophyte clumps an average speed of 50 cm·s⁻¹ was too high to capture the patterns of dominant macrophytes precisely.

To implement the EU WFD we need to build monitoring strategies that provide data with sufficient statistical confidence and develop efficient indicators (Anon 2003a,b). When measuring biological indicators a large amount of monitoring data is needed to reach this level of confidence. This goal is often difficult to achieve due to the limitation of resources. Currently evaluated techniques enable us to produce high amounts of data with acceptable quality, enabling the development of adequate assessment schemes.

To conclude, our estimates in a continuous video mode and in a still picture mode based on 35 and 20 photographs produced less variable results compared to other methods. In any long-term monitoring programme a high precision (ability to detect differences) is more important than high accuracy (ability to detect true value) (Andrew & Mapstone 1987). While reducing uncertainties of estimates, these methods should be preferred in the mapping of benthic macrophyte communities or in the assessment of water quality in the relatively homogeneous environments of the Northern Baltic Sea. When a method can only detect large changes, then it is often too late to undertake remedial actions.

Acknowledgements

Funding for this research was provided by target financed projects SF0180013s08 of the Estonian Ministry of Education, by the Estonian Science Foundation grants 6015 and 7813 and by EU project Marine Protected Areas

in the Eastern Baltic Sea (Baltic MPAs) LIFE 05 NAT/LV/000100.

References

- Andrew N.L., Mapstone B.D. (1987) Sampling and the description of spatial patterns in marine ecology. *Oceanography and Marine Biology. An Annual Review*, **25**, 39–90.
- Anonymous (2000) Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. Legislative acts and other instruments. ENV221 CODEC 513. European Union.
- Anonymous (2003a) Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Monitoring under the Water Framework Directive. Guidance Document No 7. Produced by Working Group 2.7 – Monitoring: 160 pp.
- Anonymous (2003b) Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance on typology, reference conditions and classification systems for transitional and coastal waters. Produced by Working Group 2.4 – Coast: 121 pp.
- Anonymous (2008) *Quality Classes of Surface Waterbodies and Coastal Water, Values of Quality Indicators Corresponding to Quality Class Boundaries and Guidelines for Assessing the Quality Status*. Government regulation, Ministry of Environment, Estonia. [In Estonian].
- Bucas M., Daunys D., Olenin S. (2007) Overgrowth patterns of the red algae *Furcellaria lumbricalis* at an exposed Baltic Sea coast: the result of a remote underwater video data analysis. *Estuarine, Coastal and Shelf Science*, **75**, 308–316.
- Dernie K.M., Kaiser M.J., Warwick R.M. (2003) Recovery rates of benthic communities following physical disturbance. *Journal of Animal Ecology*, **72**, 1043–1056.
- Elliot J.M. (1979) *Some Methods for the Statistical Analysis of Samples of Benthic Invertebrates*, 2nd edn. Scientific Publication Number 25. Freshwater Biology Association, Ambleside, UK.
- Eriksson B.K., Bergström L. (2005) Local distribution patterns of macroalgae in relation to environmental variables in the northern Baltic Proper. *Estuarine Coastal and Shelf Science*, **62**, 109–117.
- Field C.B., Behrenfeld M.J., Randerson J.T., Falkowski P. (1998) Primary production of the biosphere: integrating terrestrial and oceanic components. *Science*, **281**, 237–240.
- Houk P., Van Woesik R. (2006) Coral reef benthic video survey facilitates long-term monitoring in the Commonwealth of the Northern Mariana Islands: towards an optimal sampling strategy. *Pacific Science*, **60**, 177–189.
- Juanes J.A., Guinda X., Puente A., Revilla J.A. (2008) Macroalgae, a suitable indicator of the ecological status of coastal rocky communities in the NE Atlantic. *Ecological Indicators*, **8**, 351–359.
- Karlson R.H., Cornell H.V. (1998) Scale-dependent variation in local vs. regional effects on coral species richness. *Ecological Monographs*, **68**, 259–274.
- Kotta J., Orav H. (2001) Role of benthic macroalgae in regulating macrozoobenthic assemblages in the Väinameri (north-eastern Baltic Sea). *Annales Zoologici Fennici*, **38**, 163–171.
- Kotta J., Paalme T., Martin G., Mäkinen A. (2000) Major changes in macroalgae community composition affect the food and habitat preference of *Idotea baltica*. *International Review of Hydrobiology*, **85**, 693–701.
- Kotta J., Orav-Kotta H., Paalme T., Kotta I., Kukkk H. (2006) Seasonal changes in situ grazing of the mesoherbivores *Idotea baltica* and *Gammarus oceanicus* on the brown algae *Fucus vesiculosus* and *Pylaiella littoralis* in the central Gulf of Finland, Baltic Sea. *Hydrobiologia*, **554**, 117–125.
- Kotta J., Lauringson V., Kotta I. (2007) Response of zoobenthic communities to changing eutrophication in the northern Baltic Sea. *Hydrobiologia*, **580**, 97–108.
- Kotta J., Jaanus A., Kotta I. (2008a) Haapsalu and Matsalu Bay. In: Shiewer U. (Ed.), *Ecology of Baltic Coastal waters: Ecological Studies*. Springer, Berlin, Heidelberg, **197**, 245–258.
- Kotta J., Lauringson V., Martin G., Simm M., Kotta I., Herkül K., Ojaveer H. (2008b) Gulf of Riga and Pärnu Bay. In: Shiewer U. (Ed.), *Ecology of Baltic Coastal Waters. Ecological Studies*. Springer, Berlin, Heidelberg, **197**, 217–243.
- Kotta J., Paalme T., Püss T., Herkül K., Kotta I. (2008c) Contribution of scale-dependent environmental variability on the biomass patterns of drift algae and associated invertebrates in the Gulf of Riga, northern Baltic Sea. *Journal of Marine Systems*, **74** (Suppl. 1), S116–S123.
- Leujak W., Ormond R.F.G. (2007) Comparative accuracy and efficiency of six coral community survey methods. *Journal of Experimental Marine Biology and Ecology*, **351**, 168–187.
- Levin S.A. (1992) The problem of pattern and scale in ecology. *Ecology*, **73**, 1943–1967.
- Madsen J.D., Chambers P.A., James W.F., Koch E.W., Westlake D.F. (2001) The interaction between water movement, sediment dynamics and submersed macrophytes. *Hydrobiologia*, **444**, 71–84.
- Menge B.A., Olson A.M. (1990) Role of scale and environmental factors in regulation of community structure. *Trends in Ecology and Evolution*, **5**, 52–57.
- Morin A. (1985) Variability of density estimates and the optimization of sampling programs for stream benthos. *Canadian Journal of Fisheries and Aquatic Sciences*, **42**, 1530–1534.
- Murdoch T.J.T., Aronson R.B. (1999) Scale-dependent spatial variability of coral assemblages along the Florida reef tract. *Coral Reefs*, **18**, 341–351.
- Orav-Kotta H., Kotta J., Herkül K., Kotta I., Paalme T. (2009) Seasonal variability in the grazing potential of the invasive amphipod *Gammarus tigrinus* and the native amphipod *Gammarus salinus* in the northern Baltic Sea. *Biological Invasions*, **11**, 597–608.

- Orfanidis S., Panayotidis P., Stamatis N. (2001) Ecological evaluation of transitional and coastal waters: a marine benthic macrophytes-based model. *Mediterranean Marine Science*, **2**, 45–65.
- Pedersen M., Snoeijs P. (2001) Patterns of macroalgal diversity, community composition and long-term changes along the Swedish west coast. *Hydrobiology*, **459**, 83–102.
- Platt T., Denman K.L. (1975) Spectral analysis in ecology. *Annual Review of Ecology and Systematics*, **6**, 189–210.
- Riegl B., Korrubel J.L., Martin C. (2001) Mapping and monitoring of coral communities and their spatial patterns using a surface-based video method from a vessel. *Bulletin of Marine Science*, **69**, 869–880.
- Sand-Jensen K., Borum J. (1991) Interactions among phytoplankton, periphyton, and macrophytes in temperate freshwaters and estuaries. *Aquatic Botany*, **41**, 137–175.
- Schiel D.R., Lilley S.A. (2007) Gradients of disturbance to an algal canopy and the modification of an intertidal community. *Marine Ecology Progress Series*, **339**, 1–11.
- Steele J.H., Henderson E.W. (1994) Coupling between physical and biological scales. *Philosophical Transactions of the Royal Society of London, Series B*, **343**, 5–9.
- Weinberg S. (1981) A comparison of coral reef survey methods. *Bijdragen tot de Dierkunde*, **51**, 199–218.