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Response of benthic invertebrate communities to the large-scale dredging of Muuga Port

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Abstract. The dynamics of benthic invertebrate communities in Muuga Bay was described in connection with large-scale dredging activities. The spatial extent and duration of the effects were assessed by multivariate analysis and spatial modelling. In general, dredging had moderate effects on benthic invertebrates both in space and time. Still, dredging resulted in an elevated biomass of bivalves, namely that of *Macoma balthica*. These bivalves were more impacted on flat bottoms compared to steep slopes.

Key words: Baltic Sea, benthic invertebrates, dredging, interactive effects, spatial modelling.

INTRODUCTION

The coastal marine environment is severely threatened by increasing sand extraction and dredging of harbours (Dernie et al., 2003a, 2003b; Skilleter et al., 2006; Smith et al., 2006; Szymelfenig et al., 2006). Such activities result in physical disturbance of sediment structure and associated benthic communities at sites of dredging, and in organic enrichment and shifts in the community structure at sites adjacent to dredging. At moderate intensities dredging causes blooms of ephemeral or mobile species and at high intensities it may eventually defaunate large coastal areas either due to physical smothering or development of anoxia (Skilleter et al., 2006; Smith et al., 2006; Szymelfenig et al., 2006). Thus, dredging and dispersal of dredged material pose an important problem in coastal zone management (Van Dolah et al., 1984).

The multi-factorial nature of dredging impacts has been stressed in previous studies. These studies underline that the consequences of dredging represent an interplay of the spatial extent and temporal intensity of dredging with the characteristics of the habitat. Benthic communities recover faster at exposed areas whereas irreversible changes are likely at sheltered areas with poor water exchange (Szymelfenig et al., 2006). The recovery is delayed with increasing intensity and extent of dredging (Newell et al., 1998, 2004; Boyd et al., 2005). The effects of dredging on benthic invertebrates largely vary among regions. The regions that have a high proportion of mobile and opportunistic species are expected to be more resistant to the effects of dredging activities compared to regions with perennial, long-living, and sessile species (Ricklefs & Schluter, 1993; Whittaker et al., 2001).

The coastal ecosystem of the Baltic Sea is very dynamic and characterized by high physical disturbances such as resuspension of bottom sediments during extreme storm events and oxygen deficiency due to stagnant bottom water (Bonsdorff et al., 1996; Kotta et al., 2008b). Therefore it is likely that stress tolerant species of the Baltic Sea can easily cope with dredging activities unless oxygen deficiency develops (Bonsdorff, 1983). On the other hand, dredging may pose an additional challenge for Baltic species due to the presence of other stress factors such as low salinity and large temperature fluctuations (Kotta et al., 2008b). Therefore, the opposite scenario, i.e. low recovery, is also likely.

We analysed whether and how bottom topography, depth, and sediment type contributed to the influence of dredging on invertebrates. A simple generalized additive model (GAMS, Hijmans & Graham, 2006) was used directly to predict the impacts of dredging on benthic communities. Such models have been hardly used in marine systems despite the ease of their use and strong predictive capability (Kotta et al., 2008a).

MATERIAL AND METHODS

The Gulf of Finland is situated in the northern Baltic Sea. Due to high riverine loading from its eastern parts the salinity of the gulf is below 8. Muuga Bay is situated in the central Gulf of Finland. The bay is relatively exposed to the sea. The prevailing depths remain between 5 and 40 m, and bottom deposits consist mainly of clay, silt, and fine to medium sands. Hard bottoms, consisting of pebbles and boulders, are located in the vicinity of peninsulas and cover a small area.

Large-scale dredging activities were carried out in the south-eastern part of Muuga Bay due to harbour construction in 2004. Altogether 1 507 740 m^3 of material was extracted mainly in September–October. These activities affected sediment properties of the adjacent seabottom and caused significant enrichment with organic matter further away as demonstrated by an elevated concentration of sediment organic matter throughout the bay region in late 2004 and early 2005 compared to 2002–2003 (ANOVA, $p < 0.01$; database of the Estonian Marine Institute).

The sampling of benthic invertebrates was performed in Muuga Bay annually during late summers 2002–2007. Altogether 10 stations were sampled covering the most important benthic habitats of the bay (Fig. 1, Table 1). At each station one sample was taken in each sampling occasion (a total of 60 samples). The sampling and sample analysis largely followed the guidelines developed for the HELCOM COMBINE programme (HELCOM, 2006). Macrozoobenthos sampling was performed by an Ekman type bottom grab (400 cm^2, weight 8 kg, penetration depth 10 cm). Macrozoobenthos samples were sieved through a 0.25 mm mesh

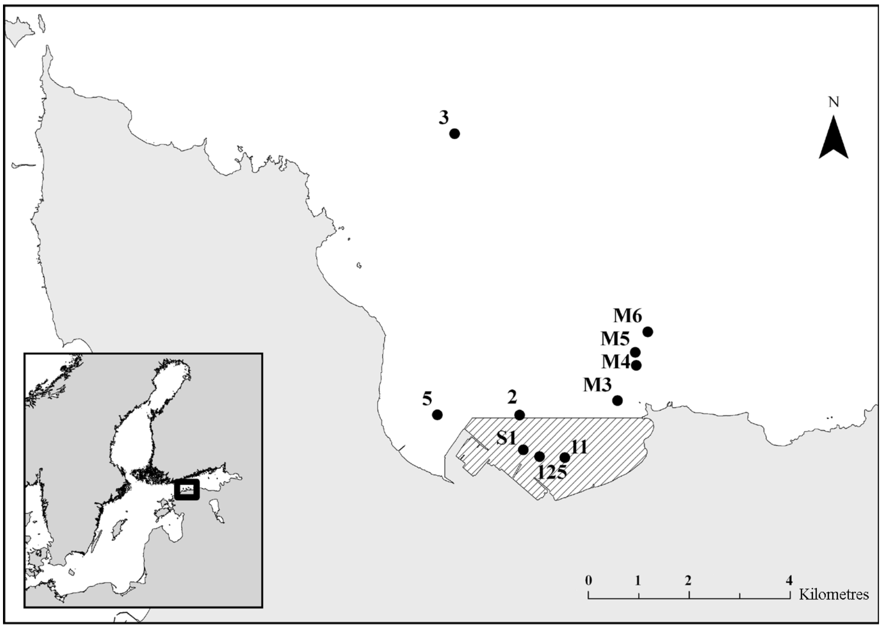

Fig. 1. Invertebrate sampling stations in Muuga Bay. The hatched area was dredged in 2004.

Table 1. Characteristics of the studied stations. Slope refers to the inclination of bottom slope in degrees at the respective spatial scale

Station	Pos N	Pos E	Depth	Sediment	Slope 100	Slope 500	Slope 1000
2	59.5051	24.9601	18.2	Silt	0.56	0.41	0.39
3	59.5142	24.9568	20.4	Silt	0.70	0.72	0.47
5	59.5052	24.9311	5.5	Sand	0.32	0.26	0.33
11	59.4974	24.9760	11.1	Sand	0.01	0.24	0.37
125	59.4976	24.9666	19.2	Silt	0.59	0.41	0.36
M3	59.5077	24.9939	6.6	Sand	0.69	0.36	0.49
M4	59.5138	25.0009	9.7	Sand	0.78	0.57	0.61
M5	59.5158	25.0006	18.4	Sand	0.78	0.57	0.61
M6	59.5193	25.0052	29.8	Silt	0.78	0.57	0.61
S1	59.4980	24.9607	18.4	Silt	0.59	0.41	0.36

and the residuals were preserved in a deep freezer at –20 °C. In the laboratory, animals were counted and identified under stereo dissecting microscope. Dry weights of all taxa were obtained after keeping the material 2 weeks at 60 °C.

As other natural factors may confound the response of invertebrates to dredging, we included measurements of the key environmental variables in the design. During sampling we recorded the depth and type of the bottom substrate. Based on depth charts (available at the Estonian Marine Institute), the inclination of coastal slopes was calculated at 100, 500, and 1000 m resolutions using the Spatial Analyst tool of ArcInfo software (Anon., 2004). High values of coastal slopes indicate the occurrence of topographic depressions at smaller spatial scale (100 m) and the occurrence of steep slopes at higher spatial scales (500–1000 m). Low values refer to flat bottoms at the measured spatial scales.

Multivariate data analyses were performed with the statistical program 'PRIMER' version 6.1.5 (Clarke & Gorley, 2006). Invertebrate biomass data were not transformed prior to the multivariate analyses. The first-stage BEST analysis was used to evaluate the effect of different environmental variables on the biomass structure of benthic invertebrates. The analysis showed whether dredging significantly described the spatio-temporal variability of invertebrate communities. Next the second-stage BEST analysis was used to evaluate how environmental variability contributed to the effects of dredging. The second-stage BEST analysis showed which environmental variables best predicted the strength of dredging–invertebrate relationship. During analyses the variability of invertebrate communities was quantified separately in each station by using a zero-adjusted Bray–Curtis coefficient. The coefficient is known to outperform most other similarity measures and it enables samples containing no organisms at all to be included (Clarke et al., 2006). The resulting dissimilarity values were then combined to test for an interactive effect of dredging and other environmental variables on the spatio–temporal variability of benthic invertebrate communities. A Spearman rank correlation (r) was computed between the similarity matrices of environmental data (Euclidean distance, environmental variables were normalized prior to analyses) and different invertebrate communities (Bray–Curtis coefficient, biotic data were not transformed). A global BEST match permutation test was run to examine the statistical significance of the observed relationships between environmental variables and biotic patterns. The environmental variables that were selected as significant in the BEST analyses were used in spatial modelling. The statistical differences in the invertebrate community structure among years were assessed using ANOSIM analysis.

Ordinary linear multiple regression analysis was used to make spatial predictions of several response variables using point surveys of the response and predictor variables. Only best predictors were selected and tested. The Akaikes Information Criterion was used to select the best model. Partial regression analysis was run to evaluate a spatial trend surface of spatial autocorrelation of residuals (a 2nd order polynomial) (Rangel et al., 2006).

RESULTS

Altogether 25 taxa of infaunal and epifaunal invertebrates were collected in the study area. By far the most prevailing species were the bivalves *Macoma balthica*, *Mytilus trossulus*, and partly *Cerastoderma glaucum*. The cirriped *Balanus improvisus* and the bivalve *Mya arenaria* were abundant at some stations. The biomass of other taxa did not exceed 0.5% of total biomass (Table 2).

Table 2. Average biomasses of benthic invertebrate taxa (g dw m^{-2}) at the studied stations

Station	2	3	5	11	M3	M4	M5	M6	125	S1
Balanus improvisus Darwin	1.503	0.000	0.000	0.000	0.000	10.098	2.786	1.666	0.000	0.000
Bathyporeia pilosa Lindström	0.000	0.000	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Cerastoderma glaucum (Poiret)	0.000	0.035	8.641	1.942	2.507	4.098	1.739	0.000	0.000	0.000
Chironomidae	0.000	0.000	0.017	0.002	0.020	0.008	0.000	0.000	0.000	0.000
Corophium volutator (Pallas)	0.007	0.000	0.000	0.011	0.071	0.452	0.012	0.000	0.010	0.016
Gammarus juv.	0.013	0.000	0.000	0.000	0.177	0.013	0.006	0.027	0.000	0.000
Gammarus oceanicus Segerstråle	0.000	0.000	0.000	0.000	0.069	0.065	0.013	0.000	0.000	0.000
Gammarus salinus Spooner	0.000	0.000	0.000	0.000	0.042	0.094	0.027	0.000	0.000	0.000
Halicryptus spinulosus von Siebold	0.000	0.015	0.000	0.000	0.000	0.000	0.000	0.635	0.014	0.000
Hediste diversicolor (O. F. Müller)	0.042	0.003	0.000	0.033	0.053	0.012	0.005	0.144	0.172	0.007
Hydrobia ulvae (Pennant)	1.304	0.466	0.066	0.643	0.372	1.319	0.271	0.026	0.143	0.351
Hydrobia ventrosa (Montagu)	0.111	0.008	0.020	0.052	0.009	0.000	0.000	0.000	0.000	0.026
Idotea balthica (Pallas)	0.022	0.000	0.022	0.000	0.000	0.000	0.000	0.044	0.000	0.000
Idotea chelipes (Pallas)	0.000	0.000	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Macoma balthica (L.)	84.282	113.984	31.172	17.565	17.656	4.611	46.602	132.403	169.372	109.408
Marenzelleria neglecta Sikorski and Bick sp. nov.	0.007	0.004	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Monoporeia affinis (Lindström)	0.000	0.009	0.000	0.000	0.004	0.000	0.067	0.213	0.000	0.000
Mya arenaria L.	0.000	0.000	0.349	0.307	1.787	10.291	0.578	0.000	0.168	0.895
Mytilus trossulus Gould	54.143	0.000	0.207	0.000	0.030	141.294	44.095	29.712	0.000	0.000
Oligochaeta	0.001	0.000	0.001	0.000	0.001	0.002	0.000	0.000	0.004	0.336
Piscicola geometra L.	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.000	0.000
Potamopyrgus antipodarum (J. E. Gray)	0.783	0.047	0.028	0.129	0.061	0.000	0.182	0.020	0.161	0.000
Cyanophtalma obscura (Schultze)	0.000	0.000	0.008	0.002	0.007	0.000	0.000	0.016	0.000	0.000
Saduria entomon (L.)	0.000	0.000	0.000	0.000	1.283	0.000	0.000	0.000	0.000	0.000
Theodoxus fluviatilis (L.)	0.159	0.000	0.055	0.000	0.000	0.515	0.000	0.000	0.000	0.000
Total biomass	142.377	114.571	40.590	20.686	24.149	172.874	96.383	164.906	170.044	111.039

The biomass dynamics of benthic invertebrates varied among species, stations, and years. The biomass of *M. balthica* was consistently higher in 2004–2005 compared to other studied years. Revised trends were due to stations that were located at the aquatory of Muuga Port. The bivalves *C. glaucum* and *M. trossulus* showed no consistent responses to dredging activities (Fig. 2).

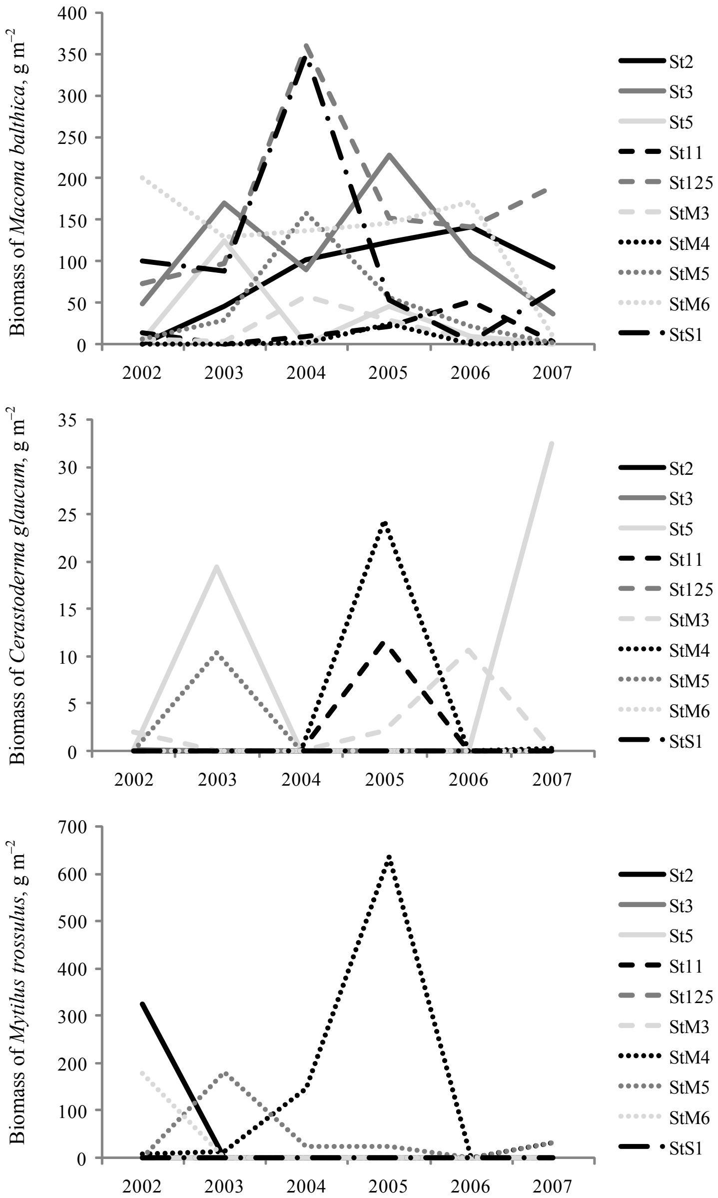

Fig. 2. Dynamics in the biomass of dominant benthic invertebrate species at the studied stations.

The biomass structure of invertebrate communities indicated a weak response to dredging as the effects of dredging were observed only at 4 stations (first-stage BEST analysis, Spearman rank correlations between 0.196 and 0.818). The biomass structure of invertebrate communities significantly differed between years, but this difference reflected environmental variability other than exposure to dredging. The second-stage BEST analysis indicated that bottom topography mainly affected the response of invertebrates to dredging (second-stage BEST analysis, Spearman rank $r = 0.484$). Flat bottoms were more sensitive to dredging compared to sites situated on slopes ($r = -0.58$). The type of sediment had also some effect on the response of invertebrates to dredging; however, the effect was not significant in the presence of coastal slope in the model, i.e. bottom topography described the variability due to sediment type. The distance of site to the dredged area was not important in the model. The coastal environment was fully recovered within 1 year after the dredging as then the pre-dredging and post-dredging communities did not diverge (ANOSIM $p > 0.05$).

The effects of dredging were mainly manifested as changes in the biomass of bivalves, especially in that of *Macoma balthica*. The biomass increased manifold during the year of dredging, especially at those areas that had lower water exchange, i.e. on flat bottoms (Fig. 3).

Spatial modelling identified large areas in western and eastern Muuga Bay that were highly sensitive to dredging as indicated by increased benthic biomasses. The spatial model described 76% of the overall variability of the community (Fig. 4).

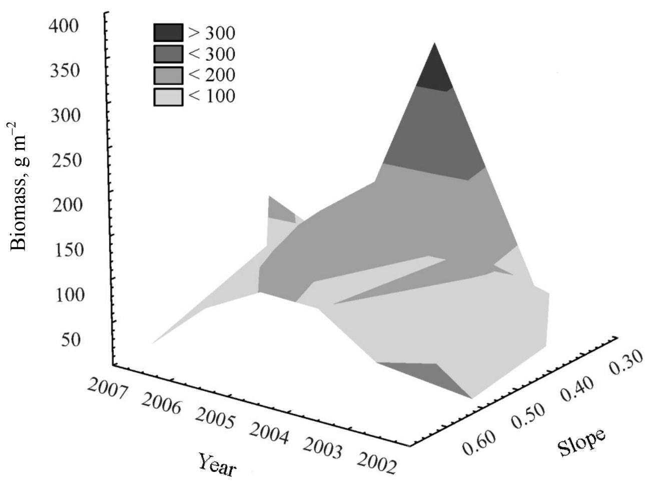

Fig. 3. Temporal trends in invertebrate biomass across different levels of coastal slopes in Muuga Bay. Biomasses above 100 g dw m^{-2} (i.e. above natural range) refer to strong impacts.

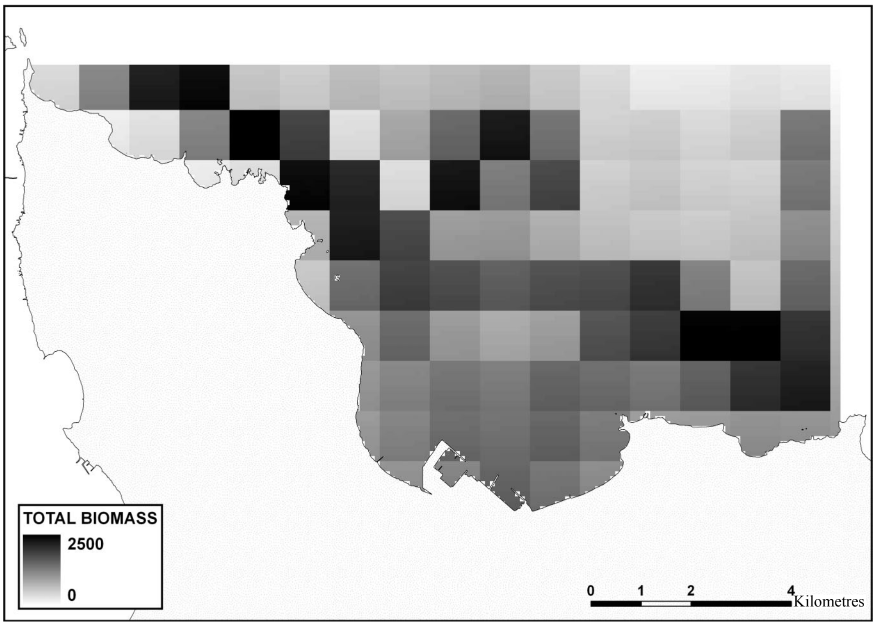

Fig. 4. Spatial prediction of total invertebrate biomasses (g dw m^{-2}) in the dredged scenario in Muuga Bay.

DISCUSSION

The impacts of dredging on biological communities are difficult to predict because physical conditions interact with the community response. Furthermore, biological systems are complex and impacts often result from indirect effects rather than direct smothering (Newell et al., 1998). Often it is factors other than dredging that largely determine the community structure, resulting in confounded effects of the dredging. Thus, the study designs that do not include the measurement of other environmental factors or lack the baseline data must be interpreted with particular care. As our study involved the key environmental variables and the structure of invertebrate communities in the design, we were able to separate the effects of dredging from those of other environmental variables and to identify the interactive effects of dredging and environmental variability on invertebrate communities.

Usually the effects of dredging persist over several years (Kenny & Rees, 1994, 1996; Newell et al., 2004) and in extreme cases over a decade (Boyd et al., 2005; Robinson et al., 2005). Our study clearly indicated that dredging had weak but consistent effects on benthic invertebrate assemblages and recovery of the communities took place within a year. This supports our hypothesis that stress-tolerant species of the Baltic Sea can easily cope with dredging activities. Often communities that are characterized by opportunistic species show weak effects and fast recoveries of dredging (Gorzelany & Nelson, 1987; Bolam & Rees, 2003; Robinson et al., 2005). Hinchey et al. (2006) and Powilleit et al. (2006) also demonstrated that *M. balthica* dominated communities were weakly affected by disposal of dredged material and communities recovered within a short period of time. On the other hand, Olenin (1992) demonstrated that several common benthic invertebrate species of the Baltic Sea are sensitive to dredging. However, the effects were significant for species abundances only and not for biomasses. Weak impacts and a high recovery potential of benthic communities are supported by a relatively high exposure of the study area (Newell et al., 1998). Our study area has a good water exchange with the deeper sea and hypoxic conditions are not likely in Muuga Bay. A rapid recovery is also expected due to a very strong seasonality of the Baltic Sea, strong natural physical disturbance, and short generation times of most near-coastal animal species (Hällfors et al., 1981).

Among invertebrate functional feeding groups, deposit feeders and suspension feeders significantly gained biomass. Our study showed that flat bottoms were more impacted by dredging compared to sites situated on slopes. This could be expected as flat bottoms are characterized by lower water exchange than slopes. With increasing water exchange the amount of deposited organic matter, i.e. food of benthic invertebrates, decreases (Newell et al., 1998).

With the rise of new powerful statistical techniques and GIS tools, there has been rapid progress in the development of predictive habitat distribution models in ecology covering as diverse aspects as biogeography, conservation biology, climate change research, and habitat or species management (Guisan & Zimmermann, 2000). Our study demonstrated that spatial predictive modelling is a useful and cost-efficient tool in near-coastal zone management as the modelled layers provide managers a possibility of reducing the overall environmental impact of future dredging activities.

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REFERENCES

Anon., 2004. *ArcGIS 9: Getting Started with ArcGIS*. ESRI.

Bolam, S. G. & Rees, H. L. 2003. Minimizing impacts of maintenance dredged material disposal in the coastal environment: a habitat approach. *Environ. Manage.*, **32**, 171–188.

Bonsdorff, E. 1983. Recovery potential of macrobenthos from dredging in shallow brackish water. In *Proceedings of the 17th European Marine Biology Symposium* (Cabioch, L., Glémarec, M. & Samain, J.-F., eds). *Oceanol. Acta*, **Spec. Vol.**, 27–32.

Bonsdorff, E., Diaz, R. J., Norkko, A. & Cutter, G. R. 1996. Characterization of soft-bottom benthic habitats of the Åland Islands, Northern Baltic Sea. *Mar. Ecol. Prog. Ser.*, **142**, 235–245.

Boyd, S. E., Limpenny, D. S., Rees, H. L. & Cooper, K. M. 2005. The effects of marine sand and gravel extraction on the macrobenthos at a commercial dredging site (results 6 years post-dredgings). *ICES J. Mar. Sci.*, **62**, 145–162.

Clarke, K. R. & Gorley, R. N. 2006. *Primer v6. User Manual/Tutorial*. Primer-E, Plymouth, UK.

Clarke, K. R., Somerfield, P. J. & Chapman, M. G. 2006. On resemblance measures for ecological studies, including taxonomic dissimilarities and a zero-adjusted Bray–Curtis coefficient for denuded assemblages. *J. Exp. Mar. Biol. Ecol.*, **330**, 55–80.

Dernie, K. M., Kaiser, M. J. & Warwick, R. M. 2003a. Recovery rates of benthic communities following physical disturbance. *J. Anim. Ecol.*, **72**, 1043–1056.

Dernie, K. M., Kaiser, M. J., Richardson, E. A. & Warwick, R. M. 2003b. Recovery of soft sediment communities and habitats following physical disturbance. *J. Exp. Mar. Biol. Ecol.*, **285–286**, 415–434.

Gorzelany, J. F. & Nelson, W. G. 1987. The effects of beach replenishment on the benthos of a subtropical Florida beach. *Mar. Environ. Res.*, **21**, 75–94.

Guisan, A. & Zimmermann, N. E. 2000. Predictive habitat distribution models in ecology. *Ecol. Model.*, **135**, 147–186.

Hällfors, G., Niemi, Å., Ackefors, H., Lassig, J. & Leppäkoski, E. 1981. Biological oceanography. In *The Baltic Sea* (Voipio, A., ed.), pp. 219–274. *Elsevier Oceanogr. Ser.*, **30**. Amsterdam.

HELCOM. 2006. Manual for Marine Monitoring in the COMBINE Programme of HELCOM. http://sea.helcom.fi/Monas/CombineManual2/CombineHome.htm (accessed 2008-04-02).

Hijmans, R. J. & Graham, C. H. 2006. The ability of climate envelope models to predict the effect of climate change on species distributions. *Glob. Change Biol.*, **12**, 2272–2281.

Hinchey, E. K., Schaffner, L. C., Hoar, C. C., Vogt, B. W. & Batte, L. P. 2006. Responses of estuarine benthic invertebrates to sediment burial: the importance of mobility and adaptation. *Hydrobiologia*, **556**, 85–98.

Kenny, A. J. & Rees, H. L. 1994. The effects of marine gravel extraction on the macrobenthos: early post-dredging recolonization. *Mar. Pollut. Bull.*, **28**, 442–447.

Kenny, A. J. & Rees, H. L. 1996. The effects of marine gravel extraction on the macrobenthos: results 2 years post-dredging. *Mar. Pollut. Bull.*, **32**, 615–622.

Kotta, J., Aps, R. & Herkül, K. 2008a. Predicting ecological resilience of marine benthic communities facing a high risk of oil spills. In *Environmental Problems in Coastal Regions VII* (Brebbia, C. A., ed.), pp. 101–110. *WIT Transactions on The Built Environment*, **99**.

Kotta, J., Lauringson, V., Martin, G., Simm, M., Kotta, I., Herkül, K. & Ojaveer, H. 2008b. Gulf of Riga and Pärnu Bay. In *Ecology of Baltic Coastal Waters* (Schiewer, U., ed.), pp. 217–243. Series: Ecological Studies, **197**. Springer.

Newell, R. C., Seiderer, L. J. & Hitchcock, D. R. 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. *Oceanogr. Mar. Biol. Annu. Rev.*, **36**, 127–178.

Newell, R. C., Seiderer, L. J., Simpson, N. M. & Robinson, J. E. 2004. Impacts of marine aggregate dredging on benthic macrofauna off the south coast of the United Kingdom. *J. Coast. Res.*, **20**, 115–125.

Olenin, S. 1992. Changes in a south-eastern Baltic soft-bottom community induced by dredged spoil dumping. In *Proceedings of the 12th Baltic Marine Biologists Symposium* (Bjørrnestad, E., Hagerman, L. & Jensen, K., eds), pp. 119–123. International Symposium Series. Olsen & Olsen, Fredensborg.

Powilleit, M., Kleine, J. & Leuchs, H. 2006. Impacts of experimental dredged material disposal on a shallow, sublittoral macrofauna community in Mecklenburg Bay (western Baltic Sea). *Mar. Pollut. Bull.*, **52**, 386–396.

Rangel, T. F. L. V. B., Diniz-Filho, J. A. F. & Bini, L. M. 2006. Towards an integrated computational tool for spatial analysis in macroecology and biogeography. *Global Ecol. Biogeogr.*, **15**, 321–327.

Ricklefs, R. E. & Schluter, D. 1993. Species diversity: regional and historical influences. In *Species Diversity in Ecological Communities: Historical and Geographical Perspectives* (Ricklefs, R. E. & Schluter, D., eds), pp. 350–364. University of Chicago Press, Chicago.

Robinson, J. E., Newell, R. C., Seiderer, L. J. & Simpson, N. M. 2005. Impacts of aggregate dredging on sediment composition and associated benthic fauna at an offshore dredge site in the southern North Sea. *Mar. Environ. Res.*, **60**, 51–68.

Skilleter, G. A., Pryor, A., Miller, S. & Cameron, B. 2006. Detecting the effects of physical disturbance on benthic assemblages in a subtropical estuary: a beyond BACI approach. *J. Exp. Mar. Biol. Ecol.*, **338**, 271–287.
Smith, R., Boyd, S. E., Rees, H. L., Dearnaley, M. P. & Stevenson, J. R. 2006. Effects of dredging activity on epifaunal communities – surveys following cessation of dredging. *Estuar. Coast. Shelf Sci.*, **70**, 207–223.
Szymelfenig, M., Kotwicki, L. & Graca, B. 2006. Benthic re-colonization in post-dredging pits in the Puck Bay (Southern Baltic Sea). *Estuar. Coast. Shelf Sci.*, **68**, 489–498.
Van Dolah, R. F., Calder, D. R. & Knott, D. M. 1984. Effects of dredging and open-water disposal on benthic macroinvertebrates in a South-Carolina estuary. *Estuaries*, **7**, 28–37.
Whittaker, R. J., Willis, K. J. & Field, R. 2001. Scale and species richness: towards a general, hierarchical theory of species diversity. *J. Biogeogr.*, **28**, 453–470.

Muuga sadama suuremastaapsete süvendamistööde mõju põhjaloomastikule

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On kirjeldatud suuremastaapsete süvendustööde mõju Muuga lahe põhjaloomastiku kooslustele. Mõjude ruumilist ulatust ja ajalist kestust uuriti mitmemõõtmeliste statistilise analüüsi ning ruumimodelleerimise meetoditega. Süvendamine mõjutas mõõdukalt põhjaloomastiku kooslusi. Erandina suurendas süvendamine oluliselt *Macoma balthica* biomassi. Järskude rannanõlvadega võrreldes mõjutas süvendamine *M. balthica* biomassi enam tasastel merepõhjadel.