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The Physical and Biological Impact of Sand Extraction: a Case Study of the Western Baltic Sea

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ABSTRACT

In autumn 1997, approximately 320,000 m³ of sand were extracted from a site located ca. 2.5 km off Wustrow, Germany, Western Baltic Sea. The physical impacts of dredging on the sea floor are assessed on the basis of side-scan sonar, sediment texture, and oxygen profile approaches. Benthic macrofaunal effects are analysed, in terms of species, abundance, and biomass (SAB), addressing the responses of sensitive and non-sensitive species.
Seabed modification was patchy within the dredging site. Morphology, texture, and oxygen characteristics returned to pre-dredging conditions over most of the site, during the first year of post-dredging. A smaller part of the area deepened by ca. 5 m and caused by multiple dredge furrows, was altered more drastically. During the year following the extraction, a shift to finer sediments with a higher organic carbon content and reduced oxygen levels was observed, at this location. Sensitive benthic species abundance did not recover to pre-impact levels, within a year after dredging. Slow recovery of the sensitive species can be overlooked easily by common environmental assessment measures, such as the SAB approach. Related to benthic habitats, environmentally-sound dredging practices should consider the various impacts that the creation of deep pits can have on the seabed, and compared those of shallow and isolated furrows.

ADDITIONAL INDEX WORDS: *aggregates, mining, side-scan sonar, sediment micro- profiles, macrozoobenthos, SAB approach, sensitive benthic species, dredging furrows, recolonisation, recovery.*

INTRODUCTION

Whilst several studies have investigated the effects of marine sand dredging on benthic associations (e.g. Essink, 1998; Øresundkonsortiet, 1998; van der Veer, Bergman, and Beukema, 1985), a substantially lower number of studies have dealt with gravel extraction (Boyd *et al.*, 2005; Cooper et al., 2007; Kenny and Rees, 1994, 1996); fewer still have investigated Baltic Sea conditions (Bonsdorff, 1980; Øresundkonsortiet, 1998). In order to develop an improved understanding of the physical recovery and macrofaunal recolonisation processes of dredged habitats in the western Baltic Sea, a dredging operation has been examined, at different spatial scales. At a broad scale, results were obtained from: (a) acoustic devices which covered an area > 1,000,000 m²; (b) grab samples, permitting examination of processes occurring over a few m²; and (c) sediment cores and oxygen profiles, providing information at a millimetre-scale. The overall objective of this study was to develop a better understanding of the physical and biological impact of dredging operations, within the Baltic Sea.

The western Baltic Sea (Schwarzer , this volume) can be regarded as a brackish water transition zone of decreasing salinity (5 - 20 psu). Faunistic diversity reaches its minimum within this salinity range (*horohaloclinicum*; Kinne, 1971, Remane, 1940). Macrofaunal species in this zone have often been described as \`transgressive´ or \`opportunistic´ (Gray, 1979; Pearson and Rosenberg, 1978); relatively \`immune´ to many anthropogenic stress factors (Wilson, 1994). However, Wilson and Elkaim (1991) have shown that not all estuarine species act as opportunists. The present study focuses upon the sensitive species, assuming that they are an important part of the local biodiversity: these mostly rare species, in accordance with the \`flush and crash´ speciation model (Piraino, Fanelli, and Boero, 2002), ensure the continuation of biological diversity.

For this study, species sensitive to physical disturbance of the seabed are defined following a concept of the \`Benthos Ecology Working Group´ of the International Council for the Exploration of the Sea (ICES), published in 1994 (ICES, 1994) and reviewed in 1995 (ICES, 1995). The ICES list of general criteria was published to develop "biogeographically-specific" lists for macrobenthic communities, to permit the prediction of disturbance related to any physical modification of the seabed. The list was based upon the assumption that, initially, any impact has a negative effect on individuals within a specific area. Sediment extraction can harm macrobenthic individuals, by damaging their bodies, burrows or shells, or by removing animals (potentially, with their tubes) from the substratum. However, disturbance can harm significantly the affected pop-

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ulations, only if the autecological characteristics of the population are characterised predominately by `sensitive´ features. The main criteria in the definition of `sensitive´ species are: a low growth rate; reduced regeneration capacities; low fecundity; infrequent recruitment; low propagule mobility; specialised habitat requirements; or narrow substratum tolerances. In contrast, macrobenthic populations with `non-vulnerable´ features are regarded as being resistant to the impact, e.g. r-strategic and highly mobile species. This ICES guideline does not draw up a list of species, but, rather, identifies standard criteria to establish regional lists. Table 2 from KRAUSE, VON NORDHEIM, and GOSSELCK (1996) provide an overview of the sensitive macrobenthic species, for the planned extraction sites of region under investigation.

SETTING

The study site was located 2.5 km northwest of Wustrow, Germany, off the coast of the Fischland Peninsula (Figure 1), in water depths of between 10 and 12 m, below Mean Sea Level. The site is situated to the south of the Darss Sill,a shoal that inhibits the inflow of bottom water of higher salinity, into the central Baltic Sea. As a consequence, salinities are usually higher (> 10 psu), to the southwest of the Darss Sill, than to the northeast of it (≈ 7 psu).

The dredging site "Wustrow II" encloses a surface area of approximately 1,100,000 m^2. The seafloor consists of moderately-sorted fine sands (Udden-Wentworth scale, TAUBER and LEMKE (1995)). The mean grain-size of surface sediments ranges from 210 to 250 µm, whilst the average thickness of the deposit amounts to 1.9 m, overlying till (StAUN Rostock, pers. comm.).In November 1997, approximately 320,000 m^3 of sand were extracted from the site, by trailer suction hopper dredging. This method leads to the production of shallow linear or

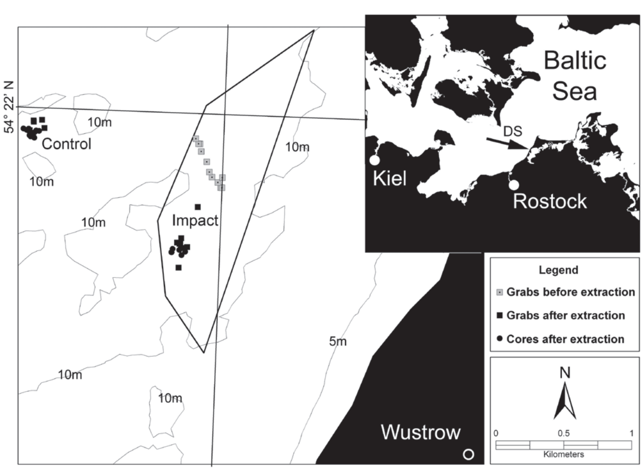

Figure 1. Map showing the location of the dredging site Wustrow II (black polygon in Figure), in the western Baltic Sea. Symbols of pre-extraction control samples are overlain by post-extraction symbols. Video imagery and observations of SCUBA divers suggested that impact sample sites be relocated after extraction to the more heavily impacted area. According to video imagery, control sites and impact sites were homogeneous sandy plains, before extraction. DS – Darss Sill.

curved furrows on the seafloor (BOYD *et al.*, 2004). Extracted sediments were utilised for beach replenishment, on the Fischland Peninsula.

METHODS

Side-Scan Sonar

Side-scan sonar surveys were carried out on 30 March, 1998 and 17 September, 1998, aboard *RV Littorina*. A Klein 595 dual-frequency (100 and 384 kHz) side-scan sonar was operated, in high-frequency mode, in order to permit the highest-resolution imaging, with a range of 75 m and a typical altitude of 8 - 10 m. Sonar returns were recorded on paper print-outs. Differential GPS was used for vessel positioning. The sonar tow fish had a layback of 15 - 20 m, whilst a sound velocity of 1500 ms^{-1} was assumed.

The analogue paper records were scanned and georeferenced, according to the method of KUBICKI and DIESING (2006). For every logged ship position (1 min^{-1}), four additional points were computed for the two side-scan sonar channels, describing both swath and water column extent. The latter points were assigned lying on the side-scan centre line . A geo-referencing method of 'rubber sheeting' was then applied (see KUBICKI and DIESING, 2006 for details); in this, the sonographs were stretched, or compressed, between two neighbouring geo-referenced points. In this way, the entire water column was removed . This method introduces positioning errors of the order of a few metres. In the absence of a more sophisticated method, to convert analogue print-outs into geo-referenced data, it was the "method-of-choice", bearing in mind its limitations.

The georeferenced side-scan sonar data were projected in Universal Transverse Mercator (UTM) projection, then displayed in a geographical information system (Esri ArcView 3.2). Blocks of 100 m by 100 m, in size, were defined according to the UTM grid. The number of identifiable tracks, per block, was counted visually for each of the surveys. As a result, maps displaying the number of dredge tracks per 100 m by 100 m block, were created.

Ground-truthing was undertaken using an underwater video camera, with an on-board control and positioning system; it was towed at an altitude of < 1 m over the sea bottom. Two transects, running from east to west and north to south, were surveyed: 4 months before extraction (August 1997); and 1 month (December 1997), , 6 months (May 1998), and 10 months (September 1998) after.

Sampling

The effects of the extraction on the sediment and macrobenthic communities were examined, using a classical BACI (before-after-control-impact) approach (HEWITT, THRUSH and CUMMINGS, 2001). The various sample locations are plotted in Figure 1. Pre-extraction samples were collected from the centre of the intended dredging site, but sites were relocated after the extraction took place (as UW-video observations located the actual centre of the dredging activity). In contrast, the control sites were not relocated. Grab and core samples were collected from both the control and impact sites. Grab samples were collected using a `heavy´ van-Veen-grab (0.1 m^2), according to the international recommendations and calibration guidelines for monitoring macrozoobenthos in the Baltic Sea (HELCOM, 1988; ICES, 1990). A minimum of three replicates were collected at each sampling site. The minimum penetration depth was 10 cm. Grab samples were collected 4 and 2 months before and 1, 4, and 10 months after extraction. 6 and 10 months after extraction, 6 to 12 core samples were collected by SCUBA divers, at the control and impact sites, respectively. These cores were taken by means of large plastic tubes (10 cm diameter, 30 cm in lenght), which could be closed underwater, at both ends. The minimum penetration depth here was 15 cm. The cores were sampled directly, at the extraction site, from the furrows. Typically, the distance between the cores was 2 m. The sample positions were assumed to be the ship's position, as measured by differential GPS (SCUBA divers were attached to the boat by a rope).

Sediment Analyses

Grain Size and Carbon Content

From each site, a grab sample was transported to the laboratory, for analysis of grain-size distribution and organic carbon content. Mean grain-size was calculated from one sub-set, using the wet sieving method of Dyer (1986). Another portion was dried for 24 h at 60 °C and was powdered. These samples were subdivided; one half was treated with excess 1M HCl (hydrochloric acid), then dried for 12 h; the other remained untreated. The carbon content of both sub-samples was determined using a "CN-Analyser" (Fision Instruments). Finally, organic carbon content was calculated by subtracting the C-values of the treated sub-samples (inorganic C), from the C-values of the untreated subsamples (total C); this was noted as a proportion of the total sediment mass.

Oxygen Profiles

SCUBA divers collected all of the cores within a single day. The collecting time was approximately two hours, for 12 cores. Measurements on the core material were made within 12 h, in a field laboratory. During measurements, untreated cores were stored open in the dark and at a temperature of approximately 10 °C.

Oxygen concentration profiles were recorded in the sediment cores, with a robust stainless steel needle electrode (Microscale Measurements), containing a sensing tip (120µm in length) for oxygen (Au-plated Pt cathode) of 120 µm (Van Gemerden *et al.*, 1989). Electrodes were polarised at least one hour before commencement of the measurement. A cellulose-nitrate membrane was installed, prior to the measurements. These were replaced each time. Calibration was carried out in an extra tube with black sediment and saturated seawater (0 % and 100 % air saturation), in which the oxygen content was measured by a calibrated electrode (WTW EOT 196). A micromanipulator was used to insert the electrode, step-wise into the sediment. Signals from the sensor were digitised by an analogue-to-digital converter, then transmitted to a notebook using the software WINDAQ (Dataq Instruments). Each measurement was repeated at least once, for each core. Samples were processed in a method similar to those of Visscher, Beukema and van Gemerden (1991) and Vopel *et al.* (1998). For further data analyses, the mean oxygen concentration for each core was calculated, in 1 to 5 mm steps.

Macrofauna

Each grab sample replicate was sieved at sea, using a 1 mm screen, before preserving the sample with 4 % buffered formosaline solution. At the laboratory, organisms were extracted and identified to species level, except for Nematoda, Turbellaria, Bryozoa, Hydrozoa, Nemertini, Oligochaeta, and Diptera, as identification was impractical. Organisms were blotted (duration varied from a few seconds for small polychaete, to several minutes for large bivalves); afterwards, the wet mass was determined with a precision of 0.1 mg. Molluscs were weighed always with shell. Identified species were classified as endangered, according to existing local and biogeographic red lists (Helcom, 1998; Merck and von Nordheim, 1996) and as `sensitive´ or `non-vulnerable´, according to guidelines of the `Benthic Ecology Working Group of ICES´ (ICES, 1994; 1995), adapted for sediment extraction by Krause, von Nordheim and Gosselck (1996).

Average species abundance and biomass were calculated per square metre, for all the sampled taxa. Number of species, abundance, and biomass were compared between control and impact sites. To avoid double usage of the same data set, neither control site data nor these of impact sites were tested statistically, between times i.e. before/after dredging. The results showed non-normal distributions; thus, transformations were avoided, with differences tested by non-parametric Mann-Whitney U and Wilcoxon tests (Bortz, Lienert and Boehnke, 1990). Contents of grab and core samples were analysed separately. Analyses were performed using a spreadsheet calculation (MS Excel) and NCSS (Hintze, 2001).

Out of 50 taxa, only 28 species were used for the multivariate statistical analysis. All other taxa were excluded from the analyses, for various reasons, i.e. not systematically sampled species like epifauna, and hyperbenthic species (e.g. *Neomysis integer*); low taxonomic resolution (e.g. Diptera); patchily distributed (e.g. *Mytilus edulis*) and species which occurred within less than 1 % of all of the samples (e.g. *Calliopius laeviusculus*). The data were used for cluster analysis and non-metric multi-dimensional scaling (nMDS) ordination, according to Clarke and Warwick (1994). The analyses were based upon Bray Curtis (Steinhaus) similarity of the fourth-root transformed abundance data. In this study, the (hierarchical) `unweighted arithmetic average clustering´, also called `UPGMA´ (Legendre and Legendre, 1998) was used; it is a standard method used in benthic ecology (Clarke and Warwick, 1994).

All statistical calculations and procedures were performed with the software packages PCOrd (McCune and Mefford, 1999), Primer 5 (Clarke and Warwick, 1994) and NCSS

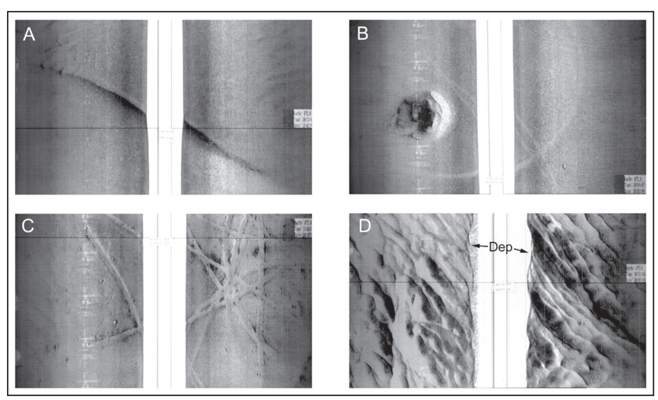

Figure 2. Representative examples of sea floor morphology revealed from 384 kHz side-scan sonar data, 4 months post-dredging: (a) sandy dunes in the control site; (b) pit of 15 m diameter and 3 m depth; (c) dredge furrows of approximately 2 m in width; and (d) multiple dredge tracks, building a depression of up to 5 m deep at the extraction site. Note: uncorrected sonographs, with a slant range of 75 m.

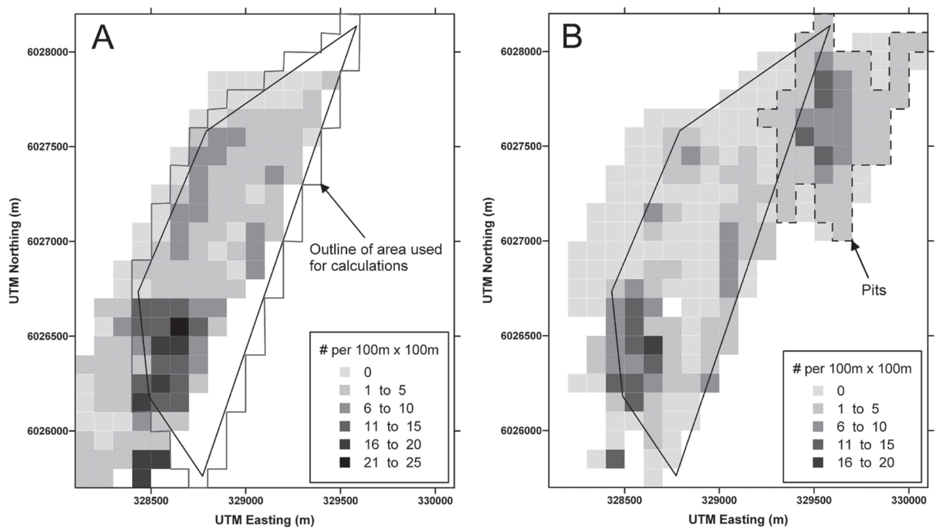

Figure 3. Number of dredge tracks per 100 m x 100 m block: (a) March 1998, 4 months post-dredging; (b) September 1998, 10 months post-dredging. Hatched line indicates area affected by static dredging causing pits (see Figure 2b.).

(HINTZE, 2001). Original data were stored and reassigned for calculations, using Access and Excel (Microsoft). The results were visualised using Grapher (Golden Software) and SigmaPlot (SPSS Inc.).

RESULTS

Seabed Morphology

From the UW video observations and side-scan sonar data, it is evident that the subaqueous dunes were present seafloor off Wustrow was predominantly sandy, with small patches of coarse-grained lag deposits to the west of the surveyed area, with wavelengths of 40 to 70 m (Figure 2a). In an area outside of the designated boundary, to the northeast of the dredging sites (Figure 2b), side-scan sonar imagery (validated by diver observations) revealed dredge pits of 10 to 40 m in diameter and up to 4 m in depth (Figure 2b). Dredge tracks of approximately 2 m in width, with a mean depth of 0.5 m (Figure 2c), were widely distributed. Where the seafloor was dredged repeatedly, multiple dredge tracks created larger depressions. In such areas, the seafloor was deepened by 3 to 5 m (Figure 2d).

Such multiple dredge tracks were present in the southwest of the designated extraction site (Figure 3a and 3b).

The intensity of the dredging impact was estimated, by evaluating the number of dredge tracks found in 100 m x 100 m blocks of the seabed. Some 142 blocks were assumed to lie within the extraction site, covering an area of 1,420,000 m^2 (Figure 3a). Four months after dredging, at least 59 % (840,000 m^2) of the site, had at least one visible furrow (Figure 3a). Ten months post-dredging, 53 % (750,000 m^2) of the dredging site had at least one furrow. This calculation does not consider the "out-of-area" dredge zone, to the SE and N (approx. 520,000 m^2, Figure 3b). Most disturbances occurred in the southwestern part of the extraction site (Figure 2d). This area contained fewer dredge tracks in September, than in March 1998.

Alteration of Sediment Characteristics

SCUBA diver observations, 6 and 10 months after extraction, indicated differences between the sediment characteristics between the control and impact sites. The flat sea bed at the control sites consisted of yellow sand whereas, following extraction at the impact site, the seafloor was composed of black and muddy sediments; occasionally, *Beggiatoa* mats were observed in the depressions and furrows.

Before dredging, the upper 10 cm of the control sites consisted of fine to medium sands (mean d = 2.4 ϕ); they were low in organic carbon (C_{org}) content (< 0.2 %) (Figure 4a). Sediments from the licensed dredging areas consisted of similar medium-sized sands (mean d = 1.8 ϕ), with a similar low organic carbon content (C_{org} < 0.35 %) (Figure 4a). Following extraction, the sediments of the dredged sites were altered fundamentally, whilst the sediments of the control sites were unchanged. Following dredging, the upper sediment layer consisted of fine sands (mean d = 2.7 ϕ) enriched in organic content ($C_{org} \approx 1.9$ %) (Figure 4a).

The core samples showed that sediments at the control sites were relatively homogeneous, not changing markedly over time. Conversely, the upper sediment layer at the impact sites revealed a fining trend, over time (Figure 4b). Thus, it may be concluded, at the impact sites, finer sediments enriched in organic carbon were accumulated on top of the original sediment (Figure 4b).

Six months after dredging the oxygen profiles from the impact sites were almost identical to these representative of pre-dredging conditions. There was a significant (n = 23; p = 0.05; Wilcoxon rank test) difference in the oxygen penetration depths, between the control site cores (mean depth: -10.9 ± 3.9 mm (11 cores; 22 profiles) and those collected from the bottom of the dredged pits (mean depth: -7.4 ± 3.6 mm (12 cores; 24 profiles)) (Figure 5a). Ten months after dredging, oxygen depletion was measured in sediment cores from the dredged furrows (mean depth: -1.6 ± 2.8 mm (3 cores; 6 profiles)), whereas the typical control site cores were more

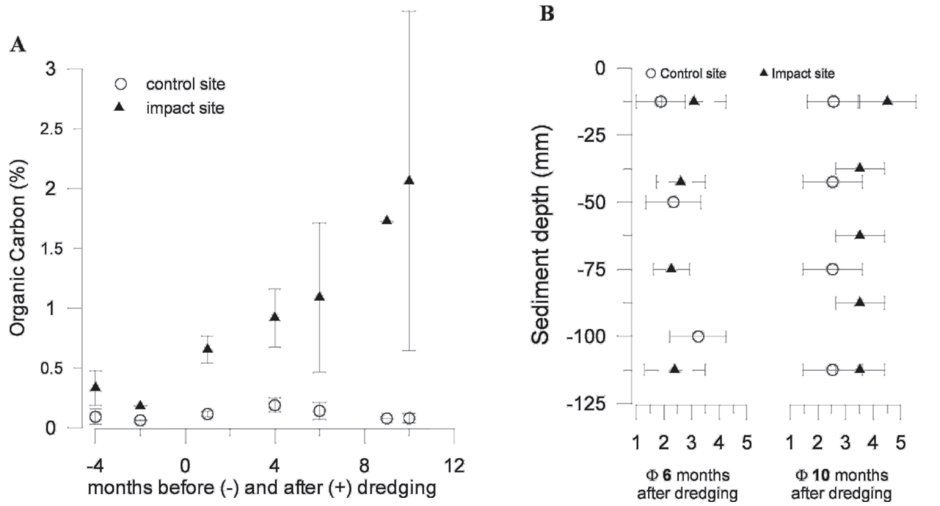

Figure 4. (a) Mean organic carbon content of one dissected representative, core at the control (open circles) and impact site (closed triangles), before and after dredging. Error bars indicate the standard deviation between the vertical sections of the core; and (b) Mean grain-size (ϕ) in 1, of the 6 sediment cores, used for the oxygen profile measurements, at the control site (open circles) and the impact site (closed triangles), 6 and 10 months after dredging. Error bars indicate the standard deviation of the sample.

oxygenated (mean depth: -38.3 ± 7.6 mm (3 cores, 6 profiles)) (Figure 5b).

Alteration of Macrofaunal Assemblages

Fifty macrobenthic taxa were sampled from the control and the impact sites; this is a representative number for this region of the Baltic Sea (Zettler, Bönsch and Gosselck, 2000). The most frequently-occurring taxa were: *Nereis* (*Hediste*) *diversicolor* (100 % occurrence, mean abundance (MA) = 85 individuals m^{-2}); *Hydrobia ulvae* (95 % occurrence, MA = 2475 individuals m^{-2}); *Pygospio elegans* (95 % occurrence, MA = 2703 individuals m^{-2}); *Macoma balthica* (95 % occurrence, MA = 77 individuals m^{-2}); *Scoloplos armiger* (91 % occurrence, MA = 281 individuals m^{-2}); and *Mytilus edulis* (86 % occurrence, MA = 313 individuals m^{-2}). Six species with `sensitive´ population characteristics were sampled before dredging. Four of these species were recorded in, at least, 50 % of control sites samples: *Mya arenaria* (91 % occurrence, MA = 407 individuals m^{-2}); *Bathyporeia pilosa* (82 % occurrence, MA = 121 individuals m^{-2}); *Travisia forbesii* (68 % occurrence, MA = 148 individuals m^{-2}); and *Cerastoderma lamarcki* (50 % occurrence, MA = 188 individuals m^{-2}). Of the other two sensitive species, *Ophelia rathkei* (29 % occurrence) was collected regularly, but only at the control sites; whereas *Bathyporeia pelagica* was generally rare (10 % occurrence).

Table 1. *Impact of dredging – SAB approach. Total number of taxa, abundance in individuals (ind m^{-2}), including standard deviation (sd) and biomass in grams (g m^{-2}), were examined at the control and the impact sites. Key: Time Before/After = month before and after extraction (year/month); (Sample type) Grab = Van Veen grab samples, Core = core samples obtained by SCUBA divers (number of samples from control site / number of samples from impact site); nd = no data; significance tested between samples from control and impact sites (Mann Whitney U-test); (ns) not significant, (*) p< 0.01, (**) p< 0.001.*

Time	Sample type	Total number of species			Abundance ind m-2 (sd)					Biomass g m-2 (sd)				
	(n control / n impact)	Control	Impact		Control		Impact			Control		Impact		
Before														
4 (97/07)	Grab (7/7)	27	31	ns	4031	(253)	6622	(316)	ns	88	(8)	119	(8)	ns
2 (97/09)	Grab (7/6)	24	25	ns	12894	(1308)	8297	(907)	ns	1072	(186)	222	(31)	ns
After														
1 (97/12)	Grab (6/3)	31	18	*	3122	(218)	2679	(366)	ns	56	(3)	87	(18)	*
4 (98/03)	Grab (5/3)	22	6	**	3508	(469)	199	(45)	**	53	(6)	0.6	(0.1)	ns
6 (98/05)	Core (5/5)	17	15	ns	12535	(1540)	4153	(448)	ns	88	(15)	6	(0.5)	ns
10 (98/09)	Grab (3/3)	23	19	**	3517	(329)	6195	(714)	**	62	(5)	21	(2)	**
10 (98/09)	Core (5/5)	11	2	**	5605	(759)	191	(0)	**	438	(120)	0	(0)	**
13 (98/12)	Grab (-/2)	nd	4		nd	1155	(473)		nd	2	(1)			

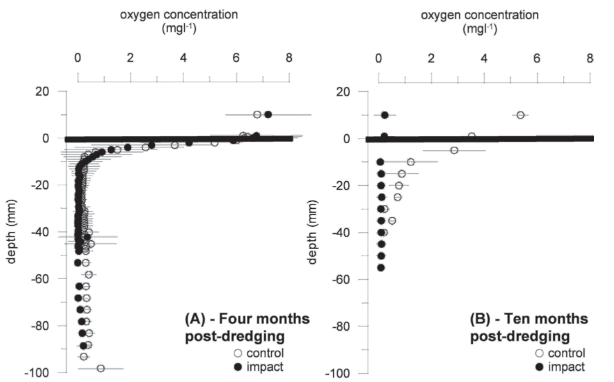

Figure 5. Sediment oxygen profiles at the control and impact site: (a) 4 months (N (cores control) = 11; N (cores impact) = 12); and (b) 10 months ((N (cores control) = 3; N (cores impact) = 3), after extraction, in November 1997. Error bars indicate one standard deviation.

Two cores and a single grab sample collected, respectively, 10 and 13 months after dredging, contained no macrobenthic animals. Comparison of Van Veen grab samples, from both the control and impact sites, 4 (n = 14) and 2 months (n = 13) before dredging, showed no significant differences (p > 0.05, Mann Whitney U-test) in number of species, abundance, or biomass (Table 1). One month after extraction, the number of species was lower at the impact sites, than before the dredging. Abundance and biomass in the control and impact sites decreased significantly, compared to the situation before dredging (Table 1). Four months after dredging, the number of species, abundance and biomass were lower at the impact sites, than at the control sites (significant, for richness and abundance) (Table 1). Ten months after dredging, control and impact sites differed significantly for the number of species, abundance, and biomass (Table 1).

The number of non-vulnerable species (Table 2) decreased after dredging at the impact sites (Table 3); they were less abundant 1 month after dredging (Table 3). Sensitive species were not found at the impact sites after dredging (Table 3),

Table 2. *List of `non vulnerable´ and `sensitive´ taxa. From 50 sampled taxa, 28 were used for multivariate statistical analysis. Taxa were determined according to Hayward and Ryland (1990) and Hartmann-Schröder (1996). Abbreviations: RL = red list of endangered species; MV = Mecklenburg-Vorpommern (local list); and BS = Baltic Sea. Class: O = Oligochaeta; P = Polychaeta; A = Amphipoda; C = Cumacea; I = Isopoda; and B = Bivalvia. Feeding Type (categories according to Fauchald and Jumars (1979)): gr = grazer; ff = filter feeder; sdf = selective deposit feeder; n-s df = non-selective deposit feeder; p = predator; gr = grazer; om = omnivore; and sc = scavenger. Category (ICES 1994; 1995): nks = biology not known sufficiently; nv = non-vulnerable; s = sensitive; and b = both. Red list: ** = presumably not endangered at present; P = potentially endangered; 3 = endangered; and 2 = critically endangered.*

Taxon	Class	Feeding type	Cat ICES	RL MV	RL BS
Arenicola marina (LINNAEUS 1758)	P	n-s df	nv		
Aricidea (Allia) suecica ELIASON 1920	P	n-s df	s	P	**
Bylgides sarsi (KINBERG 1865)	P	n-s df	nv		
Eteone longa (FABRICIUS 1780)	P	P	nv		
Marenzelleria viridis (VERRILL 1873)	P	ff & sdf	nv		
Neanthes succinea (FREY & LEUCKART 1847)	P	sc & om	nv		
Nereis (Hediste) diversicolor (O. F. MÜLLER 1776)	P	n-s df & p	nv		
Ophelia rathkei McINTOSH 1908	P	n-s df	s	P	P
Pygospio elegans CLAPARÉDE 1863	P	sdf / gr	nv		
Scoloplos armiger (O.F.MÜLLER 1776)	P	n-s df	nv		
Spio filicornis (O. F. MÜLLER 1776)	P	ff & sdf	nv		
Spio goniocephala THULIN 1957	P	ff & sdf	nks		
Travisia forbesii JOHNSTON 1890	P	n-s df	s	P	P
Heteromastus filiformis (CLAPARÉDE 1864)	P	n-s df	nv		
Capitella capitata (FABRICIUS 1780)	P	n-s df & p	nv		
Oligochaeta	O	sc & om	nv		
Tubificoides benedeni (UDEKEM 1855)	O	sc & om	nv		
Bathyporeia pelagica (BATE 1856)	A	sdf / gr	nks		
Bathyporeia pilosa LINDSTRÖM 1855	A	sdf / gr	s	P	P
Corophium volutator (PALLAS 1766)	A	n-s df	nv		
Gammarus oceanicus SEGERSTRÅLE 1947	A	p & gr	b		
Gammarus salinus SPOONER 1947	A	p & gr	b		
Diastylis rathkei KRØYER 1841	C	(non)-sdf / gr	nv	P	**
Cyathura carinata KRØYER 1848	I	p & om	s	3	3
Idotea baltica (PALLAS 1772)	I	sdf / gr	s		
Cerastoderma lamarcki (REEVE 1844)	B	ff	s	3	2
Macoma balthica (LINNAEUS 1758)	B	sdf / gr & ff	nv		
Mya arenaria LINNAEUS 1758	B	ff	b		

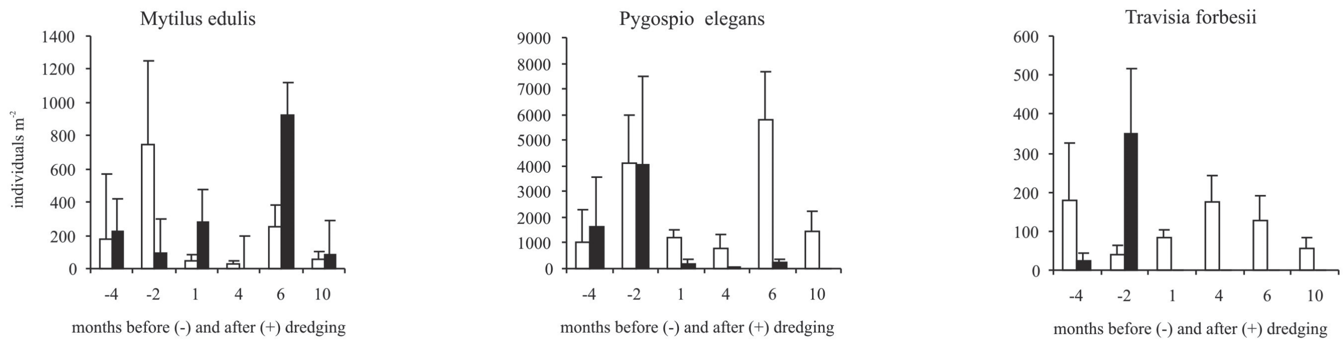

Figure 6. Mean abundance (ind m-2) of three macrobenthic species in samples, at the control and impact sites before (-) and after (+) extraction. Key: Control site - open bars; impact site - black bars. Error bars indicate one standard deviation.

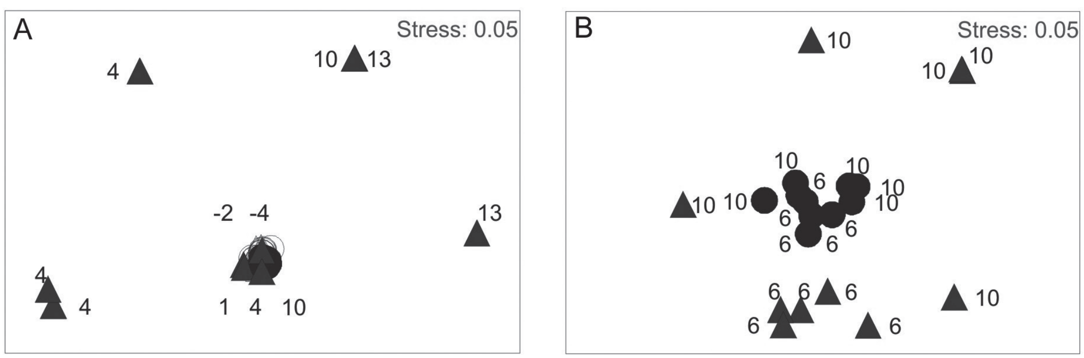

Figure 7. Non-metric MDS ordination of Steinhaus (Bray-Curtis) similarities, computed for fourth root transformed abundances, of 28 macro invertebrate species (Table 2.): (a) for 11 grab sample locations (n = 52); and (b) for 4 core sample locations (n = 20). Note that in (a), most symbols are grouped in the middle of the plot. Therefore, the numbers indicate only the months when samples in the cluster were collected, i.e. and not the specific one shown in (b). Key: open symbols-before extraction; closed symbols-after extraction; circles-control site; triangles- impact site; and numbers indicate months before (-) and after (+) dredging.

Table 3. *Effect of dredging on abundance of non-vulnerable and sensitive species. Total number of taxa and abundance (ind m^{-2} and standard deviation) as examined of the `non vulnerable´ and `sensitive´ species (Table 2.), from the total analysis. Time Before/After: Month before and after extraction (Year/Month); sample type: (Grab) Van Veen grab samples and (Core) core samples by SCUBA divers (number of samples from control site / number of samples from impact site); nd = no data; significance of each species type tested between samples from control and impact sites (Mann Whitney U-test): (ns) not significant, (*) p< 0.05, (**) p< 0.01.*

		Non vulnerable species (16 present)					**Sensitive species (7 present)**				
Time	**Sample type**	**Total number of taxa**		**Abundance ind m^{-2} (sd)**			**Total number of taxa**		**Abundance ind m^{-2} (sd)**		
	(n control / n impact)	Control	Impact	Control	Impact		Control	Impact	Control	Control	
Before											
4 (97/07)	Grab (7/7)	16	16	2307 (368)	4292 (403)	ns	5	4	949 (299)	217 (21)	ns
2 (97/09)	Grab (7/6)	16	15	6650 (997)	5029 (994)	ns	3	4	662 (270)	426 (142)	ns
After											
1 (97/12)	Grab (6/3)	15	10	1784 (298)	611 (61)	ns	7	0	158 (27)	0 (0)	**
4 (98/03)	Grab (5/3)	13	3	903 (132)	43 (4)	*	5	0	263 (67)	0 (0)	*
6 (98/05)	Core (5/5)	9	12	6714 (1388)	4108 (433)	ns	3	0	786 (147)	0 (0)	ns
10 (98/09)	Grab (3/3)	13	12	1681 (344)	3825 (751)	ns	5	2	890 (271)	331 (156)	ns
10 (98/09)	Core (3/5)	4	1	2229 (615)	127 (0)	*	3	0	828 (210)	0 (0)	ns
13 (98/12)	Grab (-/2)	nd	2	nd	38 (10)	-	nd	1	nd	28 (0)	-

except for a few small individuals of the bivalve *Cerastoderma lamarcki* and a single individual of the polychaete *Travisia forbesii*, 10 month after dredging (Table 3). At the same time, sensitive species were still sampled regularly at the control sites (Table 3). Differences in abundance before and after dredging of *Mytilus edulis* (non-vulnerable species), *Pygospio elegans* (non-vulnerable species), and *Travisia forbesii (sensitive species)* are illustrated in Figure 6. *Mytilus edulis*, as an indicator for 'non-vulnerable' and robust species, showed some abilities to cope directly with potentially-harmful alterations of the seafloor, after extraction. *Pygospio elegans* showed slow, but immediate recovery, most likely by migration, indicating 'non-vulnerability´ with good recolonising abilities. *Travisia forbesii*, as a 'sensitive' species, dwelled constantly at the control site, but its population did not recover at the impact site within the first year post- extraction (Figure 6).

Differences before and after extraction, at the control and impact sites (all the n values in Table 1) were analysed also by a non-metric multi-dimensional scaling (nMDS) (Figure 7), based upon Bray-Curtis (Steinhaus) similarities of the fourth-root transformed abundances of 28 macrobenthic species (Table 2), from the grab samples (Figure 7a) and core samples (Figure 7b). The results obtained illustrate that macrobenthic communities at the control sites were similar at all times. Additionally, no differences between the grab samples of the impact sites, before extraction, together with control sites before and after dredging, were plotted (Figure 7a). However, some 4,10, and 13 months after extraction, some samples at the impact sites showed a modified macrozoobenthic community;showed similarities (Figure 7a).

This observation illustrates the locally-heterogeneous effect of dredging. In particular, cores sampled by SCUBA divers from within the furrows in the impact sites, showed an overall difference between the macrobenthic community at the control and impact sites (Figure 7b).

DISCUSSION

In general, methods selected for analysis (as outlined above), focus on an improvement of the understanding of the effects of extraction on macrobenthic communities at different spatial scales. The number of samples is in general at the lower limit to describe alterations of benthic assemblages. However, the distribution of macrofauna in the Baltic Sea is due to only a few species occurring, which is less patchy than in other marine ecosystems (Zettler, Bönsch and Gosselck, 2000).

Seasonal effects have been controlled by sampling at the control and impact sites, at the same time. However, it was not possible to compensate for the influence of stochastic effects, such as locally-different salt-water intrusions in the study area (Prena *et al.*, 1997).

Additionally, only a small portion of the dredged area was sampled at high densities. This was possible for this site only, because of intensive observations (by both divers and a towed underwater video camera) showing that, prior to dredging, sediment surface texture and morphology were homogeneous. In the extraction area, the sample sites were relocated after extraction, to a more intensively dredged area. This approach was adopted to describe a `worst case' scenario and to avoid ambiguous results, as the singular extraction of 320,000 m^3 at Wustrow II could be considered to be a small- to medium-sized dredging operation.

Marine aggregates are used in at least 14 countries around the North Sea and the Baltic Sea, for different purposes (ICES, 2001). Such aggregates are extracted using different methods (trailer suction hopper and static suction hopper dredging), in differing environments under various hydrodynamic boundary conditions (e.g. wave-dominated vs. tide-dominated, low-energy vs. high-energy). As a consequence, the impacts of dredging, on benthic organisms and in different seas, are diverse and mostly poorly understood.

Marine aggregate extraction leads clearly to a physical disturbance of the seafloor and benthic communities. Therefore, an increasing number of studies examine physical and biological recovery, but mainly focus upon biological recolonisation. For example, Van Dalfsen and Essink (1997) and Essink (1998) have described the effects of sand extraction on benthic communities in the North Sea, reporting rapid recolonisation by surrounding fauna, within 2 to 4 years. Boers (2005) has measured the recovery of benthic communities, in a 5 - 12 m deep and 1,300 m x 500 m wide pit in the North Sea, on the Dutch Continental Shelf; this showed biomass recovery during a period of 1 to >4 years. Distribution trends in benthic communities of gravel sediments were studied, following a single experimental dredging (Kenny and Rees, 1994, 1996) and cessation of long-term dredging (Boyd *et al.*, 2003, 2004, 2005; Desprez, 2000; and Cooper *et al.*, 2007). Kenny and Rees (1996) have demonstrated rapid infilling of dredge tracks, with sand and gravel , together with rapid recolonisation by dominant species. However, the biomass was still reduced substantially, compared to its pre-dredged state, some 24 months after dredging. The results obtained by Boyd *et al.* (2005) show that the effects of high dredging intensities, on the composition of sediments and fauna, are discernable have some 6 years after the cessation of dredging.

Few studies have analysed the effects of extraction in the Baltic Sea. Bonsdorff (1980) investigated the causes of maintenance dredging in the Gulf of Finland. Øresundkonsortiet (1998), have recording the effects of sand extraction on a sandbank in the western Baltic Sea. Both of these studies have documented only minor impacts, with a rapid recovery of the original macrofauna. Recently, studies have described the physical recovery of extraction sites in the German Baltic Sea (Diesing *et al.*, 2006; Zeiler *et al.*, 2004). Diesing *et al.* (2006) showed a clear influence of water depth on physical recovery, under similar wave energy levels. Dredge tracks in Tromper Wiek were clearly discernable in 20 m of water after 12 years: similar tracks were almost obscured off Graal-Müritz, in 8 - 10 m water depth, within a year.

The modest short-term sediment extraction, examined in this study, resulted in heterogeneous alterations of the bottom morphology (Figures 2 and 3). During the first year post-dredging, physical recovery of areas with single furrows (Figure 2c) was more rapid, than in areas with dense furrows (Figure 2d); this was indicated by the decreasing numbers of detectable tracks by side-scan sonar, particularly in the northern part of the extraction field (Figure 3). Recently published results show that recognisable, but heavily weathered dredge tracks, were limited almost exclusively to the highly impacted southwestern part, some 30 months post-dredging (Diesing, 2007). Boyd *et al.* (2005) had described already such differences, in the long-term recovery of a gravel extraction site.

Quantification of large-scale heterogeneity was made possible by using side-scan sonar data sets. Therefore, its use is recommended as a standard tool for sediment extraction-related environmental impact assessments, albeit combined with adequate ground-truthing.

Major grain-size and organic carbon modifications occurred after the cessation of extraction activities. The new depression (which was previously a flat sea floor), functioned very likely as a sediment trap, collecting fine-grained and organic-rich material. Sediment fining after dredging has been described, elsewhere, by many authors (e.g. Jones & Candy, 1981; Kaplan *et al.*, 1975; van der Veer, Bergman and Beukema, 1985; and Zeiler *et al.*, 2004),but was ascribed to onboard screening of fine material (e.g. Desprez, 2000). According to the licensing procedures, this was not allowed for the extraction site, at Wustrow II.

In many studies, interpretation of the recovery trends is based upon the concept that the impact on the benthic communities ceased immediately following the cessation of dredging, when recolonisation can commence. However, in the present study, 6 to 10 months after dredging, oxygen deficiencies developed in the depression of the impact site: this was related to enrichment of organic material and the presence of stagnant water bodies in the furrows. The accumulation of silt and the development of a black and nearly anaerobic surface layer have been observed by van der Veer, Bergman and Beukema (1985), in the Wadden Sea. Similarly, in the Baltic Sea, the accumulation of organic detritus and oxygen deficiency, during periods of water body stratification, has been described for deep extraction, resilting in the creation of pits (Norden-Andersen, Nielsen, and Leth (1992). Conversely, in a 65 m deep natural pit, formed by gas eruption in the North Sea, Thatje, Gerdes and Rachor (1999) observed faunal changes; however, no oxygen deficiencies were detected. It is postulated that these dramatic consequences, on the macrobenthic communities, might occur more easily in low-energy seas, such as the Baltic Sea. However, as relevant measurements are rare, this effect might have been overlooked in previous studies.

The accumulation of fine sediment enriched in organic material and oxygen deficiencies, during summer, can be regarded as a local pattern ascribed usually to eutrophication (Boesch, 1985). According to local monitoring programmes (Lung, 1999; Lung, 2001), these alterations occurred in an area previously affected less by eutrophication (see also Rumohr, Bonsdorff and Pearson, 1996). Whereas eutrophication is often widespread, the observed dredging effects were restricted locally and disturbed areas were located adjacent to less disturbed, or undisturbed, areas.

Physical and biological recovery is interlinked and the recolonisation of benthic communities can be expected to differ, depending upon: (a) the nature of the physical impact; (b) the physical recovery stages; and (c) the regional structure of the benthic community. Typically for the Western Baltic Sea, the original fauna was structured mostly by the physical environment, characterised as a marginal sea with estuarine circulation, low salinity, an annual high temperature fluctuation, and stochastic events such as saltwater intrusions (Prena *et al.*, 1997). The salinity gradient from south to north, in combination with the geological youth of the Baltic Sea, are considered to be the primary factors for the poor taxon richness (Bonsdorff and Pearson, 1999; Remane, 1940). The benthic communities in such an environment are considered to be able to cope well with additional physical stress factors (Wilson, 1994). In their classical model, Pearson and Rosenberg (1978) have described different successional states of macrofauna, under increasing eutrophication levels. At the extraction site studied here, the common species of the western Baltic Sea, e.g. the bivalve *Mytilus edulis*, were collected regularly after dredging. Conversely, sensitive species such as the polychaete *Travisia forbesii* and the amphipod *Bathyporeia pilosa,* present continuously at the control sites, were not recorded at the impact sites after dredging (Figure 6). According to the concepts of sensitive and non-vulnerable species (ICES, 1994, 1995; Krause, von Nordheim and Gosselck, 1996) and the Pearson-Rosenberg model, that at least 3 distinct categories of macrobenthic species population recovery are proposed.

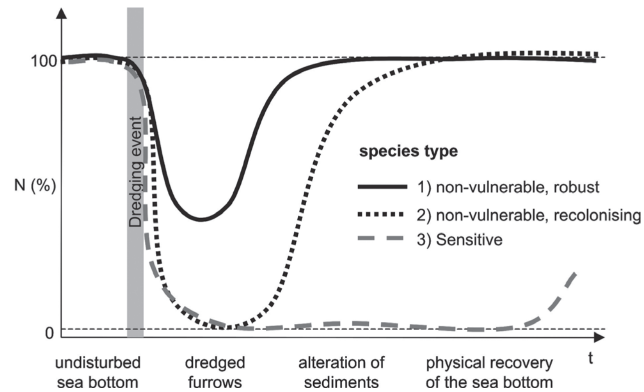

Figure 8. Conceptual model of the changes in the relative abundance (N), throughout time (t) of: (1) a `non-vulnerable´ species having the ability to cope with direct alterations of the physical conditions after dredging (i.e. robust); (2) a `non-vulnerable´ species that recolonises very rapidly after extraction, due to high reproduction rates (non-vulnerable, recolonising); and (3) a `sensitive´ species which cannot cope with post-dredging physical alterations and having limited migration and reproductive abilities, respectively.

(1) Robust non-vulnerable species: species for which occurrence and abundance decrease only slightly after dredging, due to the robustness of individuals to physical disturbances and post-dredging alterations of sediments and water column characteristics (e.g. oxygen conditions). Additionally, such species are omnipresent in the surrounding area, and, as such, larvae and adults can migrate easily into the dredged area (e.g. *Mytilus edulis*, Figure 6)

(2) Recolonising non-vulnerable species: species which are removed almost completely from the sediment, due to dredging and which cannot cope with post-dredging alterations of sediment and water column characteristics. However, due to their omnipresence in the region and rapidly migrating adults and larvae, these species are able to recolonise a dredged site rapidly (e.g. *Pygospio elegans*, *Macoma balthica*; Figure 6).

(3) Sensitive species: species which cannot cope with post-dredging, grain size alterations and oxygen depletion and cannot recolonise an area rapidly, due to low migrating abilities of the adults and larvae (e.g. *Travisia forbesii*; Figure 6).

In Figure 8, idealised development curves of relative abundances are proposed, for the species types described above. Species Type 1, due to its robustness, never disappears from the dredged area. Species Type 2 disappears but, due to well-developed migrating abilities of the larval and adult forms, reappears rapidly in the area. Species Type 3, without the abilities of the two first types, can only recolonise the region after a long phase (time) lag, in which the physical conditions of the sediments and water column have recovered and naturally-sporadic larvae settlements appear.

It should be noted that non-vulnerable species contribute most to the overall abundance of the local benthic community: non-vulnerable species types dominate the fluctuation of total abundance, within a given community. Therefore, population trends, following sediment extraction of the sensitive species and which are equivalent parts of the local biodiversity, might be overseen when using the SAB-approach only. Even multivariate analyses, which reduce the dominance of abundant species in statistic analyses, could be misleading.

CONCLUSIONS

The dredging operation has been tend to have had varied effects on the seabed. The response of areas with isolated furrows differed from areas with dense furrows, or pits. The impacts did not leave with the cessation of dredging. The sediment composition continued to change and oxygen deficiency developed, at the base of the dredged furrows. For a complete evaluation of the biological effects, it is insufficient to analyse only the overall richness, abundance, and biomass, as any effects on less abundant species can be overlooked. Such species may be characteristic and sensitive indicators for the region. Therefore, analysis at multiple scales is required, to detect any changes.

Dredging-induced impacts on benthic communities can be only minimal and short-lived, when the physical impact is limited and the site recovers rapidly to pre-dredging conditions. However, when the physical alteration cascades into subsequent prolonged harsh conditions, e.g. oxygen deficiency in summer time, a potentially rapid recovery is interrupted, until physical recovery occurs.

The planning and licensing of extraction areas and methods used in the western Baltic Sea need to be based upon solid guidelines, to retain the physical impact at a minimum, in space and time. Likewise, they need to consider the different regional conditions of the physical and biological environmental parameters, of the various sea regions.

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LITERATURE CITED

Boers, M., 2005. Effects of a deep sand extraction pit*; Final report of the PUTMOR measurements at the Lowered Dump Site*. Rijkswaterstaat, report RIKZ/2005.001, 87p.

Boesch, D.F., 1985. Effects on benthos of oxygen depletion in estuarine and coastal waters. *Estuaries*, 8, 43-55.

Bonsdorff, E., 1980. Macrozoobenthic recolonisation of a dredged brackish water bay in SW Finland. *Ophelia. Supplement*, 1, 145-155.

Bonsdorff, E. and Pearson, T.H., 1999. Variation in the sublittoral macrozoobenthos of the Baltic Sea along environmental gradients: A functional-group approach. *Australian Journal of Ecology,* 24, 312-326.

Boyd, S.E.; Limpenny, D.S.; Rees, H.L.; Cooper, K.M., and Campbell, S., 2003. Preliminary observations of the effects of dredging intensity on the re-colonisation of dredged sediments off the south-east coast of England (Area 222). *Estuarine, Coastal and Shelf Science*, 57, 209-223.

Boyd, S.E.; Cooper, K.M.; Limpenny, D.S.; Kilbride, R.; Rees, H.L.; Dearnaley, M.P.; Stevenson, J.; Meadows, W.J., and Morris, C.D., 2004. Assessment of the re-habilitation of the seabed following marine aggregate dredging. Sci. Ser. Tech. Rep., CEFAS Lowestoft, 121, 154p.

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

Bortz, J.; Lienert, G.V., and Boehnke, K., 1990. *Verteilungsfreie Statistik Methoden in der Biostatistik.* Springer, Berlin, 935p.

Clarke, K.R. and Warwick, R.M., 1994. *Change in marine communities: an approach to statistical analysis and interpretation.* Natural Environment Research Council, Plymouth, UK, 144p.

Cooper, K.; Boyd, S.; Eggleton, J.; Limpenny, D.; Rees, H., and Vanstaen, K., 2007. Recovery of the seabed following marine aggregate dredging on the Hastings Shingle Bank off the southeast coast of England. *Estuarine, Coastal and Shelf Science*, 75, 547-558.

Desprez, M., 2000. Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short- and long-term post-dredging restoration. ICES, *Journal of Marine Science,* 57, 1428-1438.

Diesing, M., 2007. What does "physical regeneration" of marine aggregate dredging sites mean? Coastal Sediments '07, *Proceedings of the Sixth International Symposium on Coastal Engineering and Science of Coastal Sediment Processes* (New Orleans, Louisiana), pp2394-2401.

Diesing, M.; Schwarzer, K.; Zeiler, M., and Klein, H., 2006. Comparison of marine sediment extraction sites by means of shoreface zonation. *Journal of Coastal Research, Special Issue* 39, 783-788.

Dyer, K.R., 1986. The sediment. *In:* Wiley, J. and Sons ltd (eds.), *Coastal and estuarine sediment dynamics*, pp.13-46.

Essink, K., 1998. *Risk analysis of coastal nourishment techniques (RIACON):* Final evaluation report. National Institute for Coastal and Marine Management (RIKZ). Haren. *RIKZ-97.031*, 98p.

Fauchald, K. and Jumars, P.A., 1979. The diet of worms: a study of polychaete feeding guilds. *Oceanography and Marine Biology: an Annual Review*, 17, 193-284.

Gray, J.S., 1979. Pollution - induced changes in populations. *Philosophical Transaction of the Royal Society of London*, Part B 286, 545-561.

Hartmann-Schroeder, g., 1996. *Annelida, Borstenwürmer, Polychaeta*. 2nd edn., Jena, Fischer, , 648p.

HAYWARD, P.J. AND RYLAND, J.S. (eds), 1990. *The marine fauna of the British Isles and North-West Europe*. Vol.1 and Vol.2., New York, Oxford University Press.

HELCOM, 1988. *Guidelines for the Baltic Monitoring Programme for the third stage (1988)*: Helsinki Commission (HELCOM). Baltic Sea Environmental Proceedings (Biological Determinants), 27D, 91-100.

HELCOM, 1998. *Red list of marine and coastal biotopes and biotope complexes of the Baltic Sea, Belt Sea and Kattegat*. Baltic Sea Environmental Proceedings, 75, 115p.

HEWITT, J.E.; THRUSH, S.E., and CUMMINGS, V.J., 2001. Assessing environmental impacts: Effects of spatial and temporal variability at likely impact scales. *Ecological Applications*, 11, 1502-1516.

HINTZE, J., 2001. *NCSS and PASS. Number cruncher statistical systems*. NCSS softwareICES, 1990. *Soft bottom macrofauna: collection and treatment of samples*. Copenhagen, International Council for the Exploration of the Sea (ICES). Techniques in marine environmental sciences. (ICES) 8, 18p.

ICES, 1994. *Benthos issues - Indicator species with reference to physical disturbance of the seabed*. Cooperative research report. Copenhagen, International Council for the Exploration of the Sea. (ICES) 204, pp.55-57.

ICES, 1995. *Environmental effects monitoring of extraction of marine aggregates*. ICES. ACME Copenhagen, International Council for the Exploration of the Sea (ICES). Report, pp.73-76.

ICES, 2001. *Effects of extraction of marine sediments on the marine ecosystem*. WGEXT. Copenhagen, International Council for the Exploration of the Sea (ICES), 247, 80p.

JONES, G. and CANDY, S., 1981. Effects of dredging on the macrobenthic infauna of Botany Bay. *Australian Journal of Marine Freshwater Research*, 32, 379-398.

KAPLAN, E. H.; WELKER, J. R.; KRAUS, M. G., and MCCOURT, S., 1975. Some factors affecting the colonisation of a dredged channel. *Marine Biology*, 32, 193–204.

KENNY, A.J. and REES, H.L., 1994. The effects of marine gravel extraction on the macrobenthos - early post-dredging recolonization. *Marine Pollution Bulletin*, 28, 442-447.

KENNY, A.J. and REES, H.L., 1996. The effects of marine gravel extraction on the macrobenthos: Results 2 years post-dredging. *Marine Pollution Bulletin*, 32, 615-622.

KINNE, O., 1971. I*nvertebrates. In:* KINNE, O.(ed.), *Marine Ecology* 1 (2), 683-1244.

KRAUSE, J.C.; VON NORDHEIM, H., and GOSSELCK, F., 1996. Effects of submarine sand and gravel extraction on benthic macrofauna in the Baltic Sea, Mecklenburg-Vorpommern, Germany. Actual problems of the marine environment. Lectures at the 6th Scientific Symposium 14 and 15 May 1996 in Hamburg, *Deutsche Hydrographische Zeitschrift,* Supplement 6, pp.189-199.

KUBICKI, A. and DIESING, M., 2006. Can old analogue sidescan sonar data still be useful? An example of a sonograph mosaic geo-coded by the DECCA Navigation System. *Continental Shelf Research*, 26, 1858-1867.

LEGENDRE, P. and LEGENDRE, L., 1998. *Numerical Ecology*, 2nd edn. Amsterdam, Elsevier Science, 853p.

LUNG, 1999. *Gewässergütebericht 1996/1997: Ergebnisse der Güteüberwachung der Fließ-, Stand- und Küstengewässer und des Grundwassers in Mecklenburg-Vorpommern.* (Water quality report Mecklenburg-Vorpommern). Landesamt f. Umwelt, Natur und Geologie (LUNG), Güstrow, 124p.

LUNG, 2001. *Gewässergütebericht 1998/1999: Ergebnisse der Güteüberwachung der Fließ-, Stand- und Küstengewässer und des Grundwassers in Mecklenburg-Vorpommern.* (Water quality report Mecklenburg-Vorpommern). Landesamt f. Umwelt, Natur und Geologie (LUNG), Güstrow, 106p.

MCCUNE, B. and MEFFORD, M.J., 1999. *Multivariate analysis of ecological data.* Gleneden Beach, MJM Software Design, 237p.

MERCK, T. and VON NORDHEIM, H., 1996. Rote Listen und Artenlisten der Tiere und Pflanzen des deutschen Meeres- und Küstenbereichs der Ostsee (Red data book for animals and plants fort the German Baltic Sea). Bundesamt f. Naturschutz (BfN), *Schriftenreihe für Landschaftspflege und Naturschutz*, Bonn-Bad Godesberg, 48, 1-108.

NORDEN ANDERSEN, O.G.; NIELSEN, P.E., and LETH J., 1992. Effects on sea bed, benthic fauna and hydrography of sand dredging in Køge Bay, Denmark. *In: Proceedings of the 12th Baltic Marine Biologists Symposium* (Helsingør, Denmark) ,International Symposium Series, 16p.

ØRESUNDKONSORTIET, 1998. *The Øresund Link. Assessment of the Impacts on the Marine Environment of the Øresund Link. Report*, Update March 1998. Øresundkonsortiet, Malmö & Copenhagen, 175p. unpublished.

PEARSON, T.H. and ROSENBERG, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanographic Marine Biological Annual Review,* 16, 229-311.

PIRAINO, S.; FANELLI, G., and BOERO, F., 2002. Variability of species' roles in marine communities: change of paradigms for conservation priorities. *Marine Biology*, 140, 1067-1074.

PRENA, J.; GOSSELCK, F.; SCHROEREN, V., and VOSS, J., 1997. Periodic and episodic benthos recruitment in southwest Mecklenburg Bay (western Baltic Sea). *Helgoländer Meeresuntersuchungen*, 51, 1-21.

REMANE, A., 1940. Einführung in die zoologische Ökologie der Nord- und Ostsee. *In:* GRIMPE, G. and (WAGLER, E.) REMANE, A. (eds.), *Die Tierwelt der Nord- und Ostsee*. Leipzig: Akademische Verlagsgesellschaft Becker & Erler,. 238p.

RUMOHR, H.; BONSDORFF, E., and PEARSON, T.H., 1996. Zoobenthic succession in Baltic sedimentary habitats. *Archive of Fishery and Marine Research*, 44, 179-213.

SCHWARZER, K., this volume. Aggregate resources and extraction in the Baltic Sea – An Introduction. *Journal of Coastal Research.*

TAUBER, F. and LEMKE, W., 1995. Map of sediment distribution in the western Baltic Sea (1:100,000), Sheet "Darß". *Deutsche Hydrographische Zeitschrift*, 47(3), 171-178.

THATJE, S.; GERDES, D., and RACHOR, E., 1999. A seafloor crater in the German Bight and its effects on the benthos. *Helgoland Marine Research*, 53, 36-44.

VAN DALFSEN, J. and ESSINK, K., 1997. *RIACON – Risk analysis of coastal nourishment techniques in The Netherlands.* Part A, Part B, Part C. EC DG XII, Mast II. (*No. MAST2-CT94-0084*), National Institute for Coastal and Marine Management / RIKZ, Haren, pp1-98.

VAN DER VEER, H.W.; BERGMAN, M.J.N., and BEUKEMA, J.J., 1985. Dredging activities in the Dutch Wadden Sea: effects on macrobenthic infauna. *Netherlands Journal of Sea Research*, 19, 183-190.

VAN GEMERDEN, H.; TUGHAN, C.S.; DE WIT, R., and HERBERT, R.A., 1989. Laminated microbial ecosystems on sheltered beaches in Scapa Flow, Orkney Islands. *Fems Microbiology Ecology*, 62, 87-102.

VISSCHER, P.; BEUKEMA, J.J., and VAN GEMERDEN, H., 1991. In situ characterization of sediments: Measurements of oxygen and sulfide profiles with a novel combined needle electrode. *Limnology and Oceanography,* 36, 1476-1480.

VOPEL, K.; DEHMLOW, J.; JOHANSSON, M., and ARLT, G., 1996. Effects of anoxia and sulphide on populations of *Cletocamptus confluens* (Copepoda, Harpacticoida). *Marine Ecology Progress Series*, 175, 121-128.

WILSON, J.G., 1994. The role of bioindicators in estuarine management. *Estuaries,* 17(1a), 94-101.

WILSON, J.G. and ELKAIM, B., 1991. The toxicity of freshwater: Estuarine bioindicators. *In:* JEFFREY, D.W. and MADDEN, B. (eds.), *Bioindicators and Environmental Management*. London: Academic Press, pp311-322.

ZEILER, M.; FIGGE, K.; GRIEWATSCH, K.; DIESING, M., and SCHWARZER, K., 2004. Regenerierung von Materialentnahmestellen in Nord- und Ostsee. *Die Küste*, 68, 67-98.

ZETTLER, M.; BÖNSCH, R., and GOSSELCK, F., 2000. Verbreitung des Makrozoobenthos in der Mecklenburger Bucht (südliche Ostsee) - rezent und im historischen Vergleich. *Meereswissenschaftliche Berichte*, 42, 1-144.