TEXTE
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Final report

Development of concepts and methods for compilation and assessment of selected anthropogenic pressures in the context of the Marine Strategy Framework Directive

by:

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publisher:

German Environment Agency

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Umwelt Bundesamt

TEXTE 111/2021

Environmental Research Plan of the Federal Ministry for the Environment, Nature Conservation and Nuclear Safety

Project No. (FKZ) 3710 25 206

Report No. (UBA-FB) FB000547/ENG

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Imprint

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Report performed by:
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Report completed in:
March 2014

Edited by:
Section II 2.3 Protection of the Marine Environment
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Publication as pdf:
http://www.umweltbundesamt.de/publikationen

ISSN 1862-4804

Dessau-Roßlau, July 2021

Abstract: Development of concepts and methods for compilation and assessment of selected anthropogenic pressures in the context of the Marine Strategy Framework Directive

In 2008, the guideline for establishing a framework for Community action in the field of marine environment (Marine Strategy Framework Directive - MSFD, 2008/56/EG) was published. The overall objective of the guideline is to achieve and/or maintain a good status of the marine environment before the year 2020. The good environmental status has to be defined in accordance with qualitative Descriptors as listed in Annex I and specified through respective Criteria and Indicators given by the European commission.

While recent projects have focussed mainly on the so-called 'state' Descriptors of the MSFD, the focus of the current project has been on 'pressure' and 'impact' Descriptors. For these, assessment systems were not yet available. Within the project, we have been identifed existing deficits and presented possible solutions, for example by developing respective assessment systems.

For Descriptor 2 (non-indigenous, invasive species) an assessment system has been developed which considers the amount of foreign species within an ecosystem, the impact on native communities, and the trend indicator.

The approach proposed for Descriptor 6 (seafloor integrity) is based on modelling the impact by combining pressure-specific sensitivity information for benthic habitats with data on the spatial and temporal extent of physical loss and damage.

Due to large gaps in knowledge on pressures and impacts on hydrographical conditions and ecosystem components, it is currently not possible to present a detailed assessment concept for Descriptor 7 (hydrographical conditions). Instead, a first draft of an assessment concept is briefly outlined as a basic framework which should be open to changes and adaptable for future developments in research.

One of the most important stressors in the marine environment is the chemical pollution which is covered by Descriptor 8 (contaminants). Initially, relevance of the contaminants, environmental quality targets, the biological effects in common, and the effects on marine mammals in particular have been examined.

For Descriptor 10 (marine litter) an assessment system for litter on beaches has been developed. Concerning litter at the water surface, within the water column, and at the seafloor (particularly fisheries nets) existing data have been analysed to suggest environmental quality targets and effective monitoring strategies. In addition, the impact on marine birds has been examined when swallowing up litter.

Moreover, possible approaches for an overall assessment concept for 'the good environmental status' according to the MSFD have been developed, with special regard to the results of recent MSFD projects.

Kurzbeschreibung: Entwicklung von Konzepten und Methoden zur Erfassung und Bewertung ausgewählter anthropogener Belastungen im Rahmen der Umsetzung der Meeresstrategie-Rahmenrichtlinie

Im Jahr 2008 trat die Meeresstrategie-Rahmenrichtlinie (Marine Strategy Framework Directive - MSFD, 2008/56/EG) in Kraft. Das Ziel dieser Richtlinie ist es, einen guten Zustand der marinen Ökosysteme bis zum Jahr 2020 zu erreichen und/oder zu erhalten. Dieser gute Umweltzustand muss anhand von qualitativen Deskriptoren gemäß Anhang I der Richtlinie definiert und durch entsprechende Kriterien und Indikatoren, die durch die Europäische Kommission formuliert wurden, spezifiziert werden.

Während sich bisherige Projekte hauptsächlich mit den sogenannten Zustandsdeskriptoren beschäftigt haben, lag der Focus des hier dargestellten Projektes auf der Betrachtung der Belastungsdeskriptoren. Für diese Deskriptoren gab es bisher keine Bewertungssysteme. Innerhalb des Projektes wurden bestehende Defizite identifiziert und mögliche Lösungen vorgeschlagen, beispielsweise durch die Erstellung entsprechender Bewertungssysteme.

Für Deskriptor 2 (nicht-einheimische, invasive Arten) wurde ein Bewertungssystem entwickelt, das neben dem Trendindikator auch die Menge bereits im System vorhandener gebietsfremder Arten sowie deren Auswirkungen auf die heimischen Gesellschaften berücksichtigt.

Der für Deskriptor 6 (Integrität des Meeresbodens) entwickelte Bewertungsansatz basiert auf der Modellierung der Beeinträchtigung, indem die Sensitivität der benthischen Habitate mit Informationen über die zeitliche und räumliche Ausdehnung der physischen Belastungen verknüpft wird.

Aufgrund großer Wissenslücken hinsichtlich der Belastungen und Auswirkungen auf die Lebensräume und Gemeinschaften ist es gegenwärtig für Deskriptor 7 (Hydrographische Bedingungen) nicht möglich, ein detailliertes Konzept zur Bewertung zu erstellen. Stattdessen wird in einem ersten Entwurf ein Konzept skizziert, das als grober Rahmen dienen und für zukünftige Erkenntnisse offen und anpassbar sein soll.

Einer der bedeutenden Stressoren in der Meeresumwelt ist die chemische Verschmutzung, die mit Deskriptor 8 (Schadstoffe) behandelt wird. Dazu wurden zunächst die Relevanz der Schadstoffe, die Umweltqualitätsziele, die biologischen Effekten im Allgemeinen und die Auswirkungen auf Meeressäuger im Speziellen untersucht.

Für den Deskriptor 10 (Meeresmüll) wurde ein Bewertungssystem für Strandmüll entwickelt. Für Müll an der Wasseroberfläche, in der Wassersäule und am Meeresboden (insbesondere Reste von Fischernetzen) wurden vorhandene Daten ausgewertet, um Umweltziele zu untersuchen und effektive Monitoringstrategien vorzuschlagen. Außerdem wurden die Auswirkungen von Müll beim Verschlucken von Meeresvögeln untersucht.

Abschließend erfolgte die Entwicklung möglicher Ansätze für ein übergreifendes Konzept (Gesamtbewertung über alle Deskriptoren) für den guten Umweltzustand unter Berücksichtigung weiterer Projekte zur Umsetzung der Meeresstrategie-Rahmenrichtlinie.

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List of Abbreviations

µ	mean value
µg	microgram
µM	micro Mol
µm	micrometer
ACI	Abundance Contamination Index
AHR	aryl hydrocarbon receptor
ANOVA	analysis of variance
ARNT	aryl hydrocarbon receptor nuclear translocator
As	Arsenic
ATR-FT-IR	Attenuated Total Reflection-Fourier Transform-Infrared Spectroscopy
AWI	Alfred-Wegener-Institute for Polar and Marine Research
BAC	Background Assessment Concentration
BCL	Biological Contamination Level
BEP	Best environmental practice
BfN	Bundesamt für Naturschutz; Federal Agency for Nature Conservation
BFR	brominated flame retardents
BITS	Baltic International Trawl Surveys
BLMP	Bund-Länder-Messprogramm
BMU	Bundesministerium für Umwelt
BPL	Biopollution Level Index
BSAP	Baltic sea action plan
BSH	Bundesamt für Seeschifffahrt und Hydrographie; Federal Maritime and Hydrographic Agency
BSII	Baltic Sea Impact Index
BSPI	Baltic Sea Pressure Index
BT	Beam trawl
BUND	Bund für Umwelt und Naturschutz Deutschland
CAS	Chemical Abstarts Service Number
Cd	Cadmium
CEMP	Coordinated Environmental Monitoring Programme
CLRTAP	Convention on Long-Range Transboundary Air Pollution
cm	centimeter

CONTIS	Continental Shelf Information System
CR	Contamination Ratio
CRP	c- reactive protein
CYP1	cytochrome P 450 proteincomplex
d	distance of rank values
DBP	Dibutylphthalat
DCA	Dichloranilin oder Dichloroacetic Acid
DDD	Dichlordiphenyl dichlorethane
DDE	Dichlordiphenyldichlorethen
DDT	Dichlordiphenyltrichlorethane
DE1	identification code of the German OSPAR beach 1
DE2	identification code of the German OSPAR beach 2
DE3	identification code of the German OSPAR beach 3
DE5	identification code of the German OSPAR beach 5
DEB	dynamic energy budged model
DEHP	Diethylhexylphthalat
DFG	derelict fishing gear
DNA	Deoxyribonucleic acid
DPSIR	Driver-Pressure-State-Impact-Response
dw	dry weight
Dx	dioxins
DYFS	Demersal Young Fish Survey
e.g.	exempli gratia
e.V.	eingetragener Verein
EC	European Commission
EcoQO	Ecological Quality Objective
EEA	European Environment Agency
EEZ	Exclusive Economic Zone
EIA	Environmental Impact Assessment
EIONET	European Environment Information and Observation Network
EPA	Environmental Protection Agency of the United States of America
EQR	ecological quality ratio
EQS	Environmental Quality Standard

EQS-AASW	Annual Average concentration of the Environmental Quality Standard (long-term EQS)
ES1	identification code of the Spanish OSPAR beach 1
ES2	identification code of the Spanish OSPAR beach 2
EU	European Union
F&E	Forschung und Entwicklung
FFH	Flora-Fauna-Habitat-Richtlinie
FS	Forschungsschiff
FT 4	free thyroxin
FTZ	Forschungs- und Technologie Zentrum Westküste
fw	fresh weight
FW	Freshwater
fwb	fat weight basis
g	gram
g mL^{-1}	gram per milliliter
GASEEZ	German Autumn Survey EEZ
GC/MS	gas chromatography combined with mass spectrometry
GES	Good Environmental Status
GESAMP	IMO/FAO/UNESCO-IOC/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection
GIS	Geographic Information Systems
GSBTS	German Small Scale Bottom Trawl Survey
HCB	Hexachlorbenzol
HCH	Hexachlorcyclohexan
HELCOM	Helsinki Commission; Baltic Marine Environment Protection Commission
Hg	Mercury
HH	Hansestadt Hamburg
HL	hidden layer
HP	haptoglobin
HpCDD	hepta- chlorinated dioxin
HSP	heat shock proteins
HxCDD	hexa- chlorinated dioxin
IAS	Invasive Alien Species
IBCI	Integrated Biocontamination Index
IBM Corp.	International Business Machines Corporation

IBPR	Integrated Biological Pollution Risk
IBTS	International Bottom Trawl Survey
ICES	International Council for the Exploration of the Seas
ICG ML	Intersessional Correspondence Group Marine Litter
ICPR	International Commission for the Protection of the Rhine
ID	identification code
IL	Interleukin
IMARES	Institute for Marine Resources & Ecosystem Studies
IMO	International Maritime Organization
in prep.	in preparation
INISI	Impact Non-Indigenous Species Index
IOW	Institut für Ostseeforschung Warnemünde, Leibniz Institute for Baltic Research
IPPC	Integrated Pollution Prevention and Control (RL 96/61/EG: IVU-Richtlinie)
ITAW	Institut für Terrestrische und Aquatische Wildtierforschung
JAMP	Joint Assessment and Monitoring Programme
JRC	Joint Research Centre of the European Commission
kg	kilogram
KIMO	Kommunenes Internasjonale Miljøorganisasjon (Local Authorities International Environmental Organization)
km	kilometre
L	litre
L. A.	Los Angeles
LC	Lethal concentration that result in x% mortality
LCPA	list of chemicals for priority action (OSPAR)
LDH	Lactatdehydrogenase
LKN-SH	Landesamt für Küsten- und Naturschutz Schleswig-Holstein
LLUR	Landesamt für Landwirtschaft, Umwelt und ländliche Räume Schleswig-Holstein, State Agency for Agriculture, Environment and Rural Areas
L-MPP	large microplastic particles
LRT	long range transport
LUNG	Landesamt für Umwelt, Naturschutz und Geologie Mecklenburg-Vorpommern
lw	lipid weight
m	meter
MCCP	medium chain chlorinated paraffins

mg	milligram
ml	millilitre
mm	millimeter
mRNA	messenger ribonucleic acid
MSF	methyl sulfone
MSFD	Marine Strategy Framework Directive
MSRL	Meeresstrategie-Rahmenrichtlinie
n	number of replicates
N3 M	identification code of sampling site on Norderney with visible macro-plastics burden
N3 O	identification code of sampling site on Norderney without visible macro-plastics burden
NABU	Naturschutzbund Deutschland e.V.
Nds	Niedersachsen; Lower Saxony
ng	nanogram
NGO	non-governmental organization
NIS	Non-Indigenous Species
NISI	Non-Indigenous Species Index
NL1	identification code of the Dutch OSPAR beach 1
NL2	identification code of the Dutch OSPAR beach 2
NL3	identification code of the Dutch OSPAR beach 3
NL4	identification code of the Dutch OSPAR beach 4
NLPV	Nationalparkverwaltung Niedersächsisches Wattenmeer
NLWKN	Niedersächsischer Landesbetrieb für Wasserwirtschaft, Küsten- und Naturschutz; Lower Saxony Water Management, Coastal Defence and Nature Conservation Agency
nm	nanometer
NOAA	National Oceanic and Atmospheric Administration (federal US agency)
NOEC	No observed effect concentration
NOEL	No observed effect level
NPE	nonylphenolethoxylate
OCDD	octa- chlorinated dioxin
OGewV	Oberflächengewässerverordnung, Verordnung zum Schutz der Oberflächengewässer
OOAO	One-out, all-out
OPE	octylphenolethoxylate
OSPAR	Oslo-Paris Convention ; Convention for the Protection of the Marine Environment of the North-East Atlantic

OT	Otter trawl
OWF	Offshore Wind Farm
p	level of significance
PAH	Polycyclic aromatic hydrocarbons
Pb	Lead
PBCR	Pathway-Specific Biological Contamination Rate
PBDE	Pentabromodiphenyl ether
PBDE	Polybrominated diphenyl ethers
PCB	Polychlorinated biphenyl
PCDD/F	Polychlorinated dibenzo-p-dioxins/furane
PCDFs	Polychlorinated dibenzofurans
PDV	Phocine distemper virus
PE	polyethylene
PEC	Predicted Exposure concentration
PeCDD	Penta- chlorinated dioxin
PELLETS-2D	two-dimensional version of a Lagrangian transport model of suspended particulate matter
PFCs	Perfluorcarbones
PFOA	Perfluorooctanesulfonic acid
PFOS	Perfluorooctane sulfonate
pg	pictogram
PhD	doctor degree
p-level	level of significance
PnCDD	Pentachlorodibenzo-p-Dioxin
PNEC	predicted non-effect concentration
POP	persistent organic pollutants
PP	polypropylene
PPAR α	Peroxisome proliferator- activated receptor
ppm	parts per million
ppt	parts per trillion
QNISI	Quantitative Non-Indigenous Species Index
QNISI-A	Quantitative Non-Indigenous Species Index on basis of Abundances
QNISI-B	Quantitative Non-Indigenous Species Index on basis of Biomass
QNISI-C	Quantitative Non-Indigenous Species Index on basis of Coverage

QS	Quality standard
r	correlation coefficient
RCI	Richness Contamination Index
REACH	Registration, Evaluation, Authorisation of Chemicals
RID	Riverine Inputs and Direct Discharges
ROV	remote operating vehicle
rs	Spearman correlation coefficient
RT- q- PCR	Real-time polymerase chain reaction
SBCI	Specific Biocontamination Index
SBPR	Species-Specific Biological Pollution Risk
SCCP	short chain chlorinated paraffins
SD	standard deviation
SE3	identification code of the Swedish OSPAR beach 3
SEA	Strategic Environmental Assessment
SIN	substitute it now
S-MPP	small microplastic particles
Spearman-r	Spearman correlation coefficient
SPSS	Statistical Package for the Social Sciences
SW	Saltwater
T- Hg	total Mercury
TBBP-A	Tetrabromobisphenol A
TBT	Tributyltin compounds
TCB	Trichlorbenzol
TGF- ß	transforming growth factor
Th	helper T cells
TiHo	Stiftung Tierärztliche Hochschule Hannover
TMAP	Trilateral Monitoring and Assessment Program
TOAO	two-out, all-out
TOC	Total organic carbon
TSEG	Trilateral seal expert group
TSG ML	Technical Subgroup Marine Litter
TT 3	triiodthyronin
TT 4	total thyroxin

UAS	unmanned aircraft systems
UBA	Umweltbundesamt; German Environment Agency
UG	uteroglobin
UK	United Kingdom
UK2	identification code of the British OSPAR beach 2
UK20	identification code of the British OSPAR beach 20
UNEP	United Nations Environment Programme
USA	United States of America
USF	Institute of Environmental Systems Research
UV	ultra violet
UWWD	Urban Wastewater Directive
VMS	Vessel Monitoring System
VO-WRRL	Verordnung zur Umsetzung der Wasserrahmenrichtlinie
WFD	Water Framework Directive
ww	wet weight
x	single value
x-axis	horizontal axis
XXT	2,3-bis (2-Methoxy-4-nitro-5-sulfophenyl)-5-[(phenylamino) carbonyl]-2H-tetrazolium hydroxide
y-axis	vertical axis
z	standardized value

Executive summary

Objective of the project

In 2008, the guideline for establishing a framework for Community action in the field of marine environment (Marine Strategy Framework Directive - MSFD, 2008/56/EG) was published. The overall objective of the guideline is to achieve and/or maintain a good status of the marine environment before the year 2020. The good environmental status has to be defined in accordance with qualitative Descriptors as listed in Annex I and specified through respective Criteria and Indicators given by the European commission.

The initial assessment includes the analysis of main characteristics of the current environmental status of the assessed waters and covers the respective physical and chemical features, the habitat types, as well as the biological features and the hydromorphology. Moreover, the most important impacts and effects, also of anthropogenic origin, on the environmental state of the targeted waters have to be analysed and qualitative and quantitative aspects of the different impacts as well as detectable trends covered.

While recent projects have focussed mainly on the so-called 'state' Descriptors of the MSFD, the focus of the current project has been on 'pressure' and 'impact' Descriptors. For these, assessment systems were not yet available, such as D2 (non-indigenous/'invasive' species), D6 (sea-floor integrity), D7 (hydrographical conditions), D8 (contaminants) and D10 (marine litter). Within the project, we have been identifed existing deficits and presented possible solutions, for example by developing respective assessment systems. Moreover, possible approaches for an overall assessment concept for 'the good environmental status' according to the MSFD have been developed, with special regard to the results of recent MSFD projects.

Work package 1: Non-indigenous species (Descriptor 2)

Within the Marine Strategy Framework Directive, non-indigenous species are considered in Descriptor 2 as a separate component, having an impact on ecosystems and biological communities. Via various naturally and anthropogenically influenced input vectors, alien species may possibly enter into the system, become established and have a strong and long-lasting impact on the native organism communities, which goes beyond the natural variability. In these cases, the species will be considered as being invasive.

There are already different approaches existent of quantifying the impact of non-indigenous species on domestic systems and thus, assessing the ecological status.

- The HELCOM trend indicator evaluates the overall number of newly introduced species over a period of six years. The good status will be reached, if no new alien species are occurring within this period.
- The three-step "Black List" system is assigning the non-indigenous species according to their potential impact on the system to a black list (species with high potential risk), to a grey list (potential risk) or to a white list (no risk).
- The Biopollution Level Index (BPL) quantifies the impact of alien species on domestic communities on different levels. The assessment consists of a five-step scale and considers abundance and distribution as well as the impact on native species, communities, habitats and ecosystem functions. From a matrix, the overall assessment will be achieved.
- The Biocontamination Index considers the abundance of the alien species and their impact on the diversity of the communities by assessing the richness.

Apart from the new entry (trend) of non-indigenous species, the quality and the status of a marine ecosystem is also determined by the number of existing alien species and their effect on native organisms and communities. The three indicators of the descriptor 2 have been defined respectively. On this basis, the multimetric "Non-indigenous Species Index" (NISI) has been developed.

- Within a five-step scale (being scalable with regard to the two-step scale of the Marine Strategy Framework Directive), the percentage of the non-indigenous species in the communities will be assessed by taking into account – dependent on the organism group – either the abundances or an indicator for the biomass.
- The impact of the alien species will be assessed by using a "Black List" system, assigning different weightings to the species being combined with abundance or biomass. This will also be a basis for a five-step assessment.
- Both classifications will be averaged to an overall index. The trend indicator will be used for an additional up- or downgrading or for a separate evaluation.

In order to find and identify the respective alien species on all organism and ecosystem levels, a spatially and temporally adapted and effective monitoring system will have to be established.

Work package 2: Seafloor integrity - Physical damage, having regard to substrate characteristics (Descriptor 6)

Within the framework of the research and development project 'Compilation and assessment of selected anthropogenic pressures in the context of the Marine Strategy Framework Directive' BioConsult Schuchardt & Scholle GbR was commissioned with the development of a concept to assess indicator 6.1.2: 'Extent of the seabed significantly affected by human activities for the different substrate types'. The approach proposed is based on modelling the impact by combining pressure-specific sensitivity information for benthic habitats with data on the spatial and temporal extent of physical loss and damage. In order to assess the spatial extent of pressures the area affected by each human activity is defined. The temporal extent is assessed by means of a five-step scale ranging from rare to persistent. Habitat sensitivity is determined by resistance (the ability to withstand disturbance or stress) in relation to a specific pressure and recoverability following the disturbance. Resistance and recovery time are categorized in sensitivity ranks both for the physical habitat features and the characteristic species. A matrix combining pressure intensity in terms of the temporal extent and habitat sensitivity supports the assessment of physical impact. A percentage value is assigned to each impact rank which should provide an approximation of the relative damage on the habitat. The cumulative physical impact for each habitat results from the sum of individual values for the relative physical impact. This method provides the advantage of easily comparing the different impacts of the pressures physical loss (reduction in extent) and physical damage (impairment of condition) and results in a single percentage value of physical degradation for each habitat.

A first application of the proposed assessment concept was carried out for the German EEZ of the North Sea. In terms of area, 'abrasion' caused by bottom trawling is the main pressure which covers nearly the complete seabed of the EEZ (98.9 %). Areas subject to physical loss currently account for less than 0.01 % of the total area. Impacts which interfere with each other are areas with aggregate extraction and bottom trawling as well as pipelines and bottom trawling. Other human uses are mutually exclusive, for example construction works and bottom trawling or operational wind farms, where fishing is excluded. The calculated cumulative impact values range from 20.1 % for sandbanks on the Borkum Reef Ground / Sylter Outer Reef to 47.8 % for reef habitats. The cumulative impact of predominant habitats adds up to approximately 40 % (sublittoral sand 37.9 %, sublittoral mud 40.3 %, sublittoral coarse sediment 43.8 %). The impact values mainly arise from high impacts of bottom trawling as major parts of the benthic habitats are fished more than once a year. The comparatively low cumulative impact value for sandbanks on the Borkum Reef Ground / Sylter Outer Reef originates from the lower fishing pressure on the Borkum Reef Ground. The high impact value for reefs is mainly caused by the high sensitivity towards 'abrasion' determined for this habitat.

Work package 3: Hydrographical conditions (Descriptor 7)

Within the framework of the research and development project 'Compilation and assessment of selected anthropogenic pressures in the context of the Marine Strategy Framework Directive' BioConsult Schuchardt & Scholle GbR was commissioned to develop an assessment concept for Descriptor 7 on hydrographical conditions. Descriptor 7 consists of three indicators to assess the spatial characterisation and the impact of permanent hydrographical changes: Indicator 7.1.1 determines the actual extent of hydrographical changes, indicator 7.2.1 describes the spatial extent of benthic and pelagic habitats affected and indicator 7.2.2 on changes in functioning of the habitats provides information on impacts on functional groups. The importance of this descriptor is seen in the opportunity to assess hydrographical alterations of large-scale cumulative impacts outside coastal waters which are currently not satisfactorily considered within national policy. It is proposed to mainly follow the advice provided by OSPAR, which is to focus on the EEZ and future large-scale projects with permanent effects on hydrographical conditions. While the emphasis should thus be on offshore wind farms, other activities leading to hydrographical alterations should also be considered in order to account for cumulative effects.

As a basis for the assessment concept, human activities which may impact hydrographical conditions and their associated pressures are described, as well as impacts on ecosystem components as far as currently foreseeable. Due to large gaps in knowledge on pressures and impacts on hydrographical conditions and ecosystem components, it is currently not possible to present a detailed assessment concept for Descriptor 7. Instead, a first draft of an assessment concept is briefly outlined as a basic framework which should be open to changes and adaptable for future developments in research. It is suggested to base the assessment on impact modelling as a pragmatic and cost-effective approach. As a first step the extent of pressures on hydrographical conditions should be modelled, i.e. changes in temperature, salinity, currents and waves, and categorized regarding their intensity. If extent and intensity of hydrographical alterations result in significant and permanent changes the next step is to examine effects on ecosystem components. This implies the linkage of the modelled pressure map with a habitat map of the North and Baltic Seas to identify benthic and pelagic habitats affected. Subsequently, the impacts on structure and function of benthic and pelagic communities have to be considered. The focus should initially be on the pelagic system and the phyto- and zooplankton communities, as first effects should be detected within these groups. If significant changes can be modelled or observed within these communities, the abundance and distribution of selected indicator species of zoobenthos and fish should be assessed.

In conclusion, it is referred to further work required for developing the assessment concept and the importance of this descriptor especially with regard to the further expansion of offshore wind farms in the North Sea and the possibly fundamental changes in the marine ecosystem is highlighted.

Work package 4: Pollutants in the marine environment (Descriptor 8)

One of the main pressures affecting the marine environment today results from chemical pollution: Work package 4 deals with pollutants in the marine environment (descriptor 8), focussing on the relevance of contaminants (chapter 5.1), environmental quality targets (chapter 5.2), biological effects (chapter 5.3) and effects on marine mammals (chapter 5.4).

Within the subchapter 5.1 the project aims towards a selection of relevant substances to be monitored under the WFSD.

Given the multitude of substances and the complexity of effects, this work takes a pragmatic approach, clustering the substances based on their use pattern and compiling the arguments for inclusion or exclusion of a substance for monitoring under the MSFD from the available data. All relevant substance inventories were summarized and consolidated. The resulting table builds the basic totality of all considered substances. Beyond this the main sources for direct emission of hazardous substances into the sea are taken into consideration.

The PEC/PNEC ratio - where available - should be used as the highest priority criteria for assessing a substance as relevant. Reliable values for PNEC and PEC in the marine environment are hardly available. The other considered criteria are: persistence and mobility of a substance, the potential for bioaccumulation, toxicity and long-range transport, contaminated sediment as secondary source, large production volumes, diffuse sources and the regulation status.

As a result, the highest priority groups are heavy metals and organotins, brominated flame retardents (BFR), Polycyclic aromatic hydrocarbons (PAH) and the large group of industry chemicals with Polyflourated Chemicals PFC, Chloralkanes and Phenols; followed by dioxins and biocides; followed by pharmaceuticals.

With CHASE an assessment toll was developed which gives each of the HELCOM objectives a status (bad, poor, moderate, good, high). As a result of the calculation mode of the CR (dividing by the squareroot) the envrionmental status is not absolutely dependent on the number of indicators, but the cumulative effects of several smaller stressors are taken into account. Supplemented by the OSPAR Assessment criteria, geographically structured and based on a confidential database, the CHASE tool in its expanded form as CHASE 2.0 seems to be a good and relatively easily applicable assessment tool.

Subchapter 5.2 deals with the environmental quality targets with particular attention to river-basin-specific pollutants according to the WFD.28 substances considered significant in the meaning of the MSRL were selected (see subchapter 5.1) and supplemented by 31 river-basin-specific pollutants by the German Environment Agency.

An approach for deriving environmental quality standards (EQS) for the marine environment is included the “Technical Guidance for Deriving Environmental Quality Standards” (Guidance Document No. 27, 2011). Depending on the quantity of data and the taxonomic groups the uncertainty varies accordingly and different assessment factors (AF) are applied to obtain an EQS. The AFs (assessment factors) for the protection of saltwater organisms are generally higher than the AFs used for the protection of freshwater organisms reflecting larger uncertainty. Concerning the EQS required for the assessment under the MSRL the question came up whether the approach of the Guidance Document No. 27 can be followed. The proce-dure is used for deriving EQS values for the selected substances and the suitability of this approach for deriving quality standards under the MSRL is discussed. However, a thorough verification and updating of the existing EQS or the derivation of new EQS would be a time- and labour-consuming task due to the large number of the selected substances (n=59). Therefore, it was not possible to perform this work in the framework of the presented project. For an overview, to check the procedures described in the Technical Guidance Document 27 (2011) and to obtain a rough estimate of saltwater EQS values for the selected substances a limited data search was performed using the comprehensive ECOTOX Database (US EPA). The general results are that current data situation for the selected substances regarding saltwater organisms is generally poor and therefore a reliable derivation of EQS values based on marine data only is not possible (except a few substances, e.g. copper). Within the pooled data from fresh- and saltwater a freshwater species is the most sensitive. However, this is regarded a result of the larger database for freshwater organisms.

Due to the limited data set for marine organisms for most of the substances and the lack of additional taxonomic groups specific for the marine environment there is a high uncertainty deriving EQS only using marine data. Therefore, the procedure of the Guidance Document 27 (2011) for using pooled data and applying higher assessment factors is considered correspondingly applicable for the MSRL, unless a large amount of marine data is available.

In subchapter 5.2 EQS_{marin} values are presented as additional information currently assessed in a project of the German Environment Agency (FKZ 3712 28 232).

Within the subchapter 5.3 the biological effects and bioindicators of the selected substances are adressed. The monitoring of biological effects in the North Sea and Baltic Sea is an issue of major concern for the new EU marine strategy framework directive MSRL. The guideline declares that concentrations of environmental pollutants should not impair aquatic organisms. Therefore, biological indicators are required to identify and verify the adverse environmental effects of contaminants and to attribute the changes to specific contaminants. A literature review was conducted to identify available biological markers that could provide a warning of the environmental impact and help to improve the regulation of discharges by regulatory authorities. Biomarkers, that are already integrated in international monitoring programs such as OSPAR/CEMP; ICES/OSPAR and HELCOM were listed and compared with further promising biological effect tools with regards to their potential use as indicators for the contaminants identified in this project considered significant in the meaning of the MSRL and the "Water Framework Directive (WFD)". The information provides a comprehensive overview of selected pollutants and their biological effects and will help to improve the use of biomarker as part of an integrated assessment of the marine ecosystem.

Within the subchapter 5.4 the effects of selected substances on marine mammals are described. Harbour seals, grey seals and harbour porpoises occur in German waters. As top-predators, they accumulate hazardous substances in their bodies. Depending on concentrations and contaminants, these substances can cause acute and chronic effects.

To evaluate the current load situation of local marine mammals, a literature search of known contaminants in animals of the North and Baltic Sea was undertaken, focusing on organic contaminants such as PCB, DDT, PFOS, PBDEs, TBT, Dioxin, and heavy metals (mercury, cadmium, lead and arsenic).

Previously, other studies about these groups of substances have been conducted, but in different regions of Europe. The sample material (e.g. blood, liver, muscle, fat), and the time of sampling vary, so that it is difficult to judge a comparison between regions and periods of time. Nevertheless, different pathological changes can be associated with particular contaminants/groups.

Harbour porpoises from less polluted waters are healthier than animals from higher PCB-polluted areas. High burdens of organic contaminants have a negative influence on fertility. Also, the immune system and the endocrine system in cetacean can be weakened through substances like PCB and PBDE. This leads to a higher susceptibility of the animals for chronic diseases and parasite infestation. In grey seals, changes in the female reproduction tract, adrenal hyperplasia, intestinal ulcer, arteriosclerosis, renal glomerulopathy, hyperkeratosis and claw lesions are summarized as "Baltic seal disease complex". They are probably induced by organochlorines like PCB and DDT.

To evaluate the effects of hazardous substances through the health of marine mammals, different methods for data collection are introduced. It is important that the current evaluation and research of contaminant burden in marine mammals is performed consistently, so that these data can be compared later on. Actual data are rare and should be collected in order to fulfil the requirement of the Marine Strategy Framework Directive.

Work package 5: Marine Litter (Descriptor 10)

Amongst others within work package 5, the Institute of Environmental Systems Research (USF) of the University of Osnabrück performed a literature study focused on literature on marine litter abundance in the North and Baltic Sea and on methods to measure its abundance. Therefore, articles in scientific journals, PhD and Master Theses, and reports of various institutions and organizations have been reviewed.

Additionally, the USF has been commissioned to analyze beach litter monitoring data for temporal and spatial trends, and to derive an evaluation system for beach litter applicable to European marine waters. For this purpose, beach litter monitoring data were acquired in close cooperation with David Fleet (Regional Agency of Coastal and Environmental Protection Schleswig-Holstein (LKN-SH), Tönning) from the OSPAR Intersessional Correspondence Group Marine Litter (OSPAR ICG ML) (OSPAR, 2009) and from German non-governmental environmental organizations, which have performed beach litter monitoring at a variety of beaches for partly more than 20 years (Clemens et al., 2011). These data were analyzed applying univariate and multivariate statistical techniques, as well as artificial neural networks, and were used to derive a classification of beaches according to descriptor 10 of the Marine Strategy Framework Directive (MSFD). Based on this classification in a next step, a multi-criteria evaluation system was derived.

Beyond the work with beach litter data, the USF had the task to identify further sources of long-term data on marine litter at the sea surface and in the water column, to analyze these data, and to suggest further methods to measure the abundance of marine litter. Some publications regarding the identification of litter at the sea surface and the vertical distribution of marine litter in the water column, which have been found during the literature study, are summarized. Spatial distributions of floating litter and litter at the sea bottom were acquired, analyzed and correlated to each other, as well as to beach litter monitoring data.

Moreover, methodological work has been done to optimize and standardize analytics of microplastics in beach samples. First counts and analytical results of microplastics are given and related to results from previous works on microplastics.

Trends in the amount and composition of litter ingested by marine organisms (MSFD indicator 10.2.1) were examined by analyses of stomach contents to monitor the impact of litter on marine life. In this context, the German contribution to the established OSPAR Fulmar Litter EcoQO approach was continued within the current project.

The Northern Fulmar qualified as a suitable indicator as it feeds exclusively at sea, ingests a wide variety of plastic items and accumulates these in its stomach. Moreover, Northern Fulmars are abundant in the North Sea, North Atlantic and Pacific and are often found in beached bird surveys. The stomach contents of Fulmars have been examined for plastic contents in the entire North Sea since 2002 following an initial Dutch pilot study. The international investigation is led by the Dutch IMARES institute and has resulted in the definition of the Fulmar Litter EcoQO (Ecological Quality Objective), which was implemented by OSPAR. The Fulmar EcoQO is reached when less than 10% of Fulmars exceed the level of 0.1g of plastic per stomach. This target level refers to a pollution level in a reference area where the environmental quality is considered to be acceptable.

Since 2002 the German contribution is conducted by the FTZ in collaboration with a network of professional and voluntary supporters who collect the beached birds. The methodologies on dissections and analysis of stomach contents are internationally standardised. Within the current project 37 stomachs of Fulmars found in 2011, 52 of 2012 and several additional samples from earlier years were processed and analysed in combination with data from earlier study years.

In the current situation, i.e. the most recent 5-year period (2008-2012), 96% of the 235 Fulmars found in Germany had plastic litter in their stomach. This comprised on average 22.6 plastic pieces totaling 0.34g per bird. The two main plastic categories, industrial plastics and user plastics, could be found in 53% and in 96% of Fulmars, respectively. User plastics consisted predominantly of fragments which were found in 91% of birds accounting for almost half of the overall plastic mass. Foamed plastics (mostly polystyrene) were another dominant component of the user plastic category. Since the early 2000s the average mass of user plastics increased from 0.29g per bird in Germany (2003-2007) to 0.36g (2007-2011) but dropped back to the initial value of 0.29g in the current period (2008-2012). In contrast, the pollution level of industrial plastics stayed almost stable with around 0.05g per bird for the entire period. Compared with the Dutch samples the composition of litter shows very similar patterns. The observed high incidence of plastic litter unavoidably has mechanical and chemical consequences affecting the body condition of birds.

The Fulmar monitoring tool using the OSPAR Fulmar Litter EcoQO approach is the only fully mature MSFD indicator on ingested litter so far. It comprises the most comprehensive data set in the EU focusing on the trends of ingested litter or its impacts. It is applicable to most of the North East Atlantic and can be used to assess temporal and regional trends of different litter categories. In consequence, it has been adopted as an indicator for GES in the European MSFD.

In the sub-project ‘Trends in macroscopic litter on the seafloor and ghost nets’, litter present on the seafloor valued as an anthropogenic stressor was used as an index for the environmental status. In addition, the spatial distribution and abundance of fisheries nets trapped and entangled in ship wrecks was recorded in order to investigate the harmful effects of the potential stressor ‘ghost net’.

The Alfred Wegener Institute Helmholtz-Centre for Polar and Marine Research has regularly conducted beam trawl and otter trawl catches in the German EEZ and in the Wadden Sea of the North Sea between 1998 and 2010 in order to monitor the development of the benthic faunal communities over a longer spatial period. In the reports of these scientific fishery hauls, the amounts of litter caught in the nets have been recorded as well. In the present study these data were used to estimate the amount and spatial distribution of litter on the seafloor in the German Bight and to describe the development of the amounts of litter over time. According to the results, areas were selected and methods recommended for a long-term monitoring on the development of the amounts of litter on the seafloor and a suggestion was made on the degree of pollution and the quality of the litter that should represent an achieved ‘good state’ of the seafloor sensu MSRL. Carefully averaged over all beam trawl catches, a mean litter contamination of 10 kg km^{-2} can be calculated for the German EEZ (North Sea). Transferred to the area of the German EEZ covering 28,539 km^2, this refers to a total amount of 285,390 kg of inorganic litter. In four areas a temporal comparison was possible.

The Federal Maritime and Hydrographic Agency (BSH) conducts inspection dives at the ship wrecks in the German EEZ. By video inspections it was possible to roughly estimate the amount of ‘ghost nets’ on wrecks. In the present study, the inspection video recordings of 64 wrecks were used to:

- estimate the percentage coverage of wrecks through nets in the German EEZ and to determine the types of nets caught in the wrecks,
- depict the spatial distribution of the ‘wreck ghost nets’ in the German EEZ,
- estimate the total surface area of ‘ghost nets’ on ship wrecks in the German Bight and
- identify selected wrecks at which harmful impacts directly effected through the ‘ghost nets’ could be investigated in the future.

The degree of coverage was identified for each wreck, but not the actual number of nets covering a respective wreck. The average surface of a wreck covered with nets was 37.4±65.5 m^2. Transferred and extrapolated to the number of about 1,300 wrecks in the southern North Sea this amounts to 48,600 m² of stationary 'ghost nets' in the German Bight. A regular ascertainment of net data at wrecks is highly recommended.

Work package 6: Aggregation of pressures to an overall assessment (Integrated Ecosystem Assessment)

The concept of the Marine Strategy Framework Directive (MSFD) is a holistic approach implying the assessment of the European marine ecosystems. In general, 'Integrated Ecosystem Assessments' can be done in different ways.

- The simplest way of an integrated ecosystem assessment is the 'one-out, all-out' (OOAO) approach. That means that the overall status is determined by the worst status of the components used in the assessment.
- Another well-known and often used method for aggregating data is the averaging and weighting.
- In order to analyse interactions between impacts and ecosystem components a decision tree or a decision matrix can be applied.
- Very often indices are used to assess the overall state of an ecosystem (trophic states, negative impacts on indicator organisms, et al.). For a comprehensive approach to assess the ecosystem, the cumulative effects will have to be regarded.

In addition to these methods, combinations of different methods are feasible. One example is the combination of the OOAO approach and the weighted average method with a decision tree, as applied within the frame of the WFD.

Different approaches for an integrated ecosystem assessment with regard to the implementation of the MSFD have been proposed within the report as follows.

For a particular marine region to be classified, this will result in 12 individual assessments (11 descriptors and the cumulative effects) that have to be aggregated to an integrated overall assessment. In order to achieve that goal, three conceptual approaches can be used. One pre-condition for applying this approach is the development of individual assessments systems for each descriptor.

- The simplest approach for an integrated ecosystem assessment (IEA) would be to calculate just weighted average values for the 12 components. Instead of applying the two-stage classifying system of the MSFD ('Good Environmental Status' (GES) is reached or not), it will be recommended to employ a five-stage classification system for assessing the results in order to get much more detailed information on respective trends within the ecosystem.
- A second approach is the combination of the 'One Out, All Out' principle and the weighted assessment. It should be considered whether the absolute and schematic approach of the OOAO principle could be alleviated in the sense that one or two negative assessment values for the 12 components could be allowed without preventing the overall assessment from reaching the GES.
- An alternatively proposed method for an overall assessment is based on the aggregation of indicators, which are described in the Commission Decision for the MSFD. These indicators will be divided into two groups of variables: status variables such as habitat size, distribution and condition as well as ecosystem structure and function, and biodiversity on one hand, and pressure variables such as impacts of nutrients and pollutants as well as effects of litter and noise on the other hand.

The ecological status is not only defined by the intensity of the anthropogenic pressures, but is also based on interaction effects between different aspects of the ecosystem, which can indicate a possible disturbance of the ecological balance. Such a description of the ecological status can be obtained by applying an integrative matrix system. Such as matrix consists of all species and species groups considered for the assessment as well as of measures for the ecosystem structure, ecosystem function, biodiversity, and eventually socio-economic aspects. Moreover, cumulative effects such as interaction effects, temporal, and spatial effects are closely linked to the results of the indicators and criteria, which define the aspects of the ecosystem the integrative matrix is based on.

Zusammenfassung

Ziel des Projektes

Im Jahr 2008 trat die Meeresstrategie-Rahmenrichtlinie (Marine Strategy Framework Directive - MSFD, 2008/56/EG) in Kraft. Das Ziel dieser Richtlinie ist es, einen guten Zustand der marinen Ökosysteme bis zum Jahr 2020 zu erreichen und/oder zu erhalten. Dieser gute Umweltzustand muss anhand von qualitativen Deskriptoren gemäß Anhang I der Richtlinie definiert und durch entsprechende Kriterien und Indikatoren, die durch die Europäische Kommission formuliert wurden, spezifiziert werden.

Die Anfangsbewertung der Gewässer beinhaltet die Analyse der wichtigsten Merkmale des derzeitigen Umweltzustands und umfasst die jeweiligen physikalischen und chemischen Kenngrößen, die Lebensraumtypen sowie die biologischen Eigenschaften und die Hydromorphologie des jeweiligen Systems. Darüber hinaus müssen jedoch auch die äußeren Einwirkungen (auch die anthropogenen Ursprungs) und deren Effekte auf den Umweltzustand analysiert und deren qualitative und quantitative Aspekte sowie nachweisbare Trends betrachtet werden.

Während sich bisherige Projekte hauptsächlich mit den sogenannten Zustandsdeskriptoren beschäftigt haben, lag der Focus des hier dargestellten Projektes auf der Betrachtung der Belastungsdeskriptoren. Für diese Deskriptoren, beispielsweise D2 (nicht-einheimische, invasive Arten), D6 (Integrität des Meeresbodens), D7 (Hydrographische Bedingungen), D8 (Schadstoffe) and D10 (Meeresmüll) gab es bisher keine Bewertungssysteme. Innerhalb des Projektes wurden bestehende Defizite identifiziert und mögliche Lösungen vorgeschlagen, beispielsweise durch die Erstellung entsprechender Bewertungssysteme. Außerdem wurden mögliche Ansätze für ein übergreifendes Konzept (Gesamtbewertung über alle Deskriptoren) für den guten Umweltzustand unter Berücksichtigung weiterer Projekte zur Umsetzung der Meeresstrategie-Rahmenrichtlinie entwickelt.

Arbeitspaket 1: Nicht-einheimische Arten (Deskriptor 2)

Nicht-einheimische Arten werden innerhalb der Meeresstrategie-Rahmenrichtlinie als separater Einflussfaktor auf die Ökosysteme und Lebensgemeinschaften unter Deskriptor 2 aufgeführt. Über verschiedene natürliche und anthropogen beeinflusste Eintragsvektoren können gebietsfremde Arten in Systeme gelangen, sich dort etablieren und unter Umständen die heimischen Organismengesellschaften nachhaltig über die natürliche Variabilität hinaus beeinflussen. In solchen Fällen bezeichnet man die Arten als invasiv.

Es gibt bereits verschiedene Ansätze, um den Einfluss von nicht-einheimischen Arten auf die heimischen Systeme zu quantifizieren und damit den ökologischen Zustand zu bewerten.

- Der HELCOM-Trendindikator betrachtet über einen Zeitraum von sechs Jahren die Gesamtzahl neu eingetragener Arten. Der gute Umweltzustand ist dabei erreicht, wenn keine neuen gebietsfremden Arten in diesem Zeitraum auftreten.
- Das dreistufige „Schwarze-Listen-System“ ordnet die nicht-einheimischen Arten nach dem Ausmaß der Effekte auf das System in eine schwarze Liste (Arten mit hohem Gefährdungspotential), eine graue Liste (mögliches Gefährdungspotential) und eine weiße Liste (keine Gefahr).
- Der Biopollution Level Index (BPL) berücksichtigt auf verschiedenen Ebenen den Einfluss fremder Arten auf die heimischen Gesellschaften. In die Bewertung gehen jeweils mit einer fünfstufigen Skala die Abundanz und Verbreitung, der Einfluss auf heimische Arten und Gesellschaften, auf die Habitate und die Ökosystemfunktionen ein. Über eine Matrix gelangt man schließlich zur Gesamtbewertung.

- Beim Biocontamination-Index werden die Abundanz der fremden Arten und deren Einfluss auf die Diversität der Gesellschaften über die Richness berücksichtigt.

Neben dem Neueintrag (Trend) von nicht-einheimischen Arten werden die Qualität und der Zustand eines marinen Ökosystems wesentlich aber auch von der Menge bereits vorhandener fremder Arten und deren Effekte auf die heimischen Organismen und Gesellschaften bestimmt. Entsprechend sind die drei zugehörigen Indikatoren für den Deskriptor 2 definiert. Auf dieser Basis wurde der multimetrische „Non-indigenous Species Index“ (NISI) entwickelt.

- In einer fünfstufigen Skala (skalierbar zur zweistufigen Einordnung innerhalb der Meeresstrategie-Rahmenrichtlinie) wird der Prozentsatz nicht-einheimischer Arten an den Gesellschaften je nach Organismengruppe bezogen auf die Abundanzen oder einen Indikator für Biomasse bewertet.
- Die Auswirkungen der fremden Arten werden über ein „Schwarze-Listen-System“ zugeordnet, in dem die Arten unterschiedliche Gewichtungen erhalten, die bei den vorkommenden Organismen mit deren Abundanz oder Biomasse kombiniert werden. Darauf basierend wird eine ebenfalls fünfstufige Skala für die Bewertung erstellt.
- Beide Klassifizierungen werden schließlich gemittelt zum Gesamtindex kombiniert. Der Trendindikator wird für eine zusätzliche Herauf- oder Herabstufung herangezogen oder separat für eine Beurteilung benutzt.

Um auf allen Organismen- und Ökosystemebenen die entsprechenden nicht-einheimischen Arten zu finden und auch benennen zu können, muss ein angepasstes und effektives Monitoringsystem räumlich und zeitlich etabliert werden.

Arbeitspaket 2: Integrität des Meeresgrunds - physische Schäden unter Berücksichtigung der Substrateigenschaften (Deskriptor 6)

Im Rahmen des F+E-Vorhabens “Erfassung und Bewertung ausgewählter anthropogener Belastungen im Kontext der Meeresstrategie-Rahmenrichtlinie“ wurde BioConsult Schuchardt & Scholle GbR mit der Entwicklung eines Bewertungskonzepts für den Indikator 6.1.2 „Ausdehnung des durch menschliche Aktivitäten erheblich beeinträchtigten Meeresbodens in Bezug auf verschiedene Substrattypen“ beauftragt. Der vorgeschlagene Bewertungsansatz basiert auf der Modellierung der Beeinträchtigung, indem die Sensitivität der benthischen Habitate mit Informationen über die zeitliche und räumliche Ausdehnung der physischen Belastungen verknüpft wird. Zur Erfassung der räumlichen Ausdehnung der Belastung wird der Wirkraum für jede menschliche Aktivität definiert. Die zeitliche Komponente der Belastung wird mit Hilfe einer fünfstufigen Skala klassifiziert. Die Sensitivität der Habitate setzt sich zusammen aus der Resistenz als der Fähigkeit einer Störung oder Stress zu widerstehen und der Fähigkeit zur Erholung im Anschluss an eine Störung, jeweils bezogen auf eine spezifische Beeinträchtigung. Resistenz und Erholungszeit werden sowohl für die physikalischen Habitateigenschaften als auch für die charakteristischen Arten bestimmt und in Sensitivitätskategorien eingeordnet. Die Erfassung der physischen Beeinträchtigung erfolgt mit Hilfe einer Matrix, die die Intensität der Belastung hinsichtlich der zeitlichen Ausdehnung mit der Sensitivität der Habitate verbindet. Jeder Belastungskategorie wird ein Prozentwert zugeordnet, der eine Annäherung an die relative Schädigung des Habitats darstellen soll. Die kumulative physische Beeinträchtigung jedes Habitats resultiert aus der Summe der Einzelwerte der relativen physischen Beeinträchtigung. Diese Methode bietet den Vorteil der direkten Vergleichbarkeit unterschiedlicher Auswirkungen von Flächenverlust und physischer Schädigung als Beeinträchtigung des Zustands der Habitate.

Eine erste Anwendung der Bewertungsmethode erfolgt für die deutsche AWZ der Nordsee. Flächenmäßig ist „Abschürfung“ durch die Schleppnetzfischerei die Hauptbelastung, die nahezu die gesamte AWZ betrifft

(98,9 % der Fläche). Der Verlust von Habitatflächen beträgt aktuell weniger als 0,01 % der Gesamtfläche. Kumulative Beeinträchtigungen ergeben sich durch die Überlagerung von Fischerei und Sandabbaugebieten sowie von Fischerei und Rohrleitungen. Andere Nutzungen schließen sich gegenseitig aus, wie z.B. Bautätigkeiten und Schleppnetzfischerei oder Offshore-Windparks, da hier die Fischerei verboten ist. Die berechneten Werte der kumulativen Beeinträchtigung reichen von 20,1 % für Sandbänke im Bereich Borkum Riffgrund / Sylter Außenriff bis zu 47,8 % für Riffe. Die kumulative Beeinträchtigung der vorherrschenden Habitate beträgt etwa 40 % (sublitoraler Sand 37,9 %, sublitoraler Schlick 40,3 %, sublitorales grobes Sediment 43,8 %). Die Werte ergeben sich zumeist aus den hohen Belastungen durch die Schleppnetzfischerei, da weite Teile der benthischen Habitate mehrfach pro Jahr befischt werden. Der vergleichsweise niedrige Wert der kumulativen Beeinträchtigung für die Sandbänke im Bereich Borkum Riffgrund / Sylter Außenriff resultiert vor allem aus dem geringeren Fischereidruck auf dem Borkum Riffgrund. Der hohe Wert für Riffe wird dagegen hauptsächlich durch die relativ hohe Sensitivität des Habitats verursacht.

Arbeitspaket 3: Veränderung der hydrografischen Bedingungen (Deskriptor 7)

Im Rahmen des F+E-Vorhabens “ Erfassung und Bewertung ausgewählter anthropogener Belastungen im Kontext der Meeresstrategie-Rahmenrichtlinie“ wurde BioConsult Schuchardt & Scholle GbR mit der Entwicklung eines Konzepts zur Bewertung des Deskriptors 7 „Hydrographische Bedingungen“ beauftragt. Deskriptor 7 beinhaltet drei Indikatoren zur Erfassung der räumlichen Ausdehnung und der Auswirkungen dauerhafter hydrographischer Veränderungen: Indikator 7.1.1 beschreibt die Ausdehnung der von hydrographischen Veränderungen betroffenen Fläche, Indikator 7.2.1 bezieht sich auf die Ausdehnung der von den Veränderungen betroffenen pelagischen und benthischen Lebensräume und Indikator 7.2.2 erfasst Veränderungen in der Funktion der beeinträchtigten Habitate und Gemeinschaften. Die Bedeutung des Deskriptors 7 wird in der Möglichkeit gesehen, hydrographische Veränderungen großräumiger kumulativer Beeinträchtigungen zu erfassen, die außerhalb der Küstengewässer auftreten und die derzeit nicht ausreichend durch die nationale Gesetzgebung berücksichtigt werden. Es wird vorgeschlagen der Vorgehensweise von OSPAR weitgehend zu folgen, d.h. die Bewertung des Deskriptors fokussiert auf der AWZ und zukünftigen großräumigen Projekten mit dauerhaften hydrografischen Veränderungen. Der Schwerpunkt sollte daher auf Offshore-Windparks liegen, wobei andere menschliche Tätigkeiten mit Auswirkungen auf die Hydrographie mitberücksichtigt werden müssen, um kumulative Beeinträchtigungen erfassen zu können.

Als Basis für ein Bewertungskonzept werden menschliche Aktivitäten mit Auswirkungen auf die hydrographischen Bedingungen und die damit verbundenen Belastungen sowie die derzeit absehbaren Effekte auf das marine Ökosystem beschrieben. Aufgrund großer Wissenslücken hinsichtlich der Belastungen und Auswirkungen auf die Lebensräume und Gemeinschaften ist es gegenwärtig nicht möglich, ein detailliertes Konzept zur Bewertung des Deskriptors 7 zu erstellen. Stattdessen wird in einem ersten Entwurf ein Konzept skizziert, das als grober Rahmen dienen und für zukünftige Erkenntnisse der Meeresforschung offen und anpassbar sein soll. Ein auf der Modellierung der Auswirkungen basierendes Bewertungskonzept als pragmatischen und kostengünstigen Ansatz wird vorgeschlagen. Als erster Schritt sollte die räumliche Ausdehnung der hydrographischen Belastungen modelliert werden, d.h. Veränderungen von Temperatur, Salinität, Strömungen und Wellengang, und hinsichtlich der Intensität klassifiziert werden. Ergeben sich daraus erhebliche und dauerhafte Veränderungen der hydrographischen Bedingungen, wird als nächster Schritt die Auswirkung auf die Lebensräume betrachtet. Hierfür wird die modellierte Belastungskarte mit einer Habitatkarte der Nord- und Ostsee verknüpft, um die betroffenen benthischen und pelagischen Habitate zu identifizieren. Anschließend werden die Folgen für die Struktur und Funktion der benthischen und pelagischen Gemeinschaften untersucht. Der Schwerpunkt sollte dabei zunächst auf dem Pelagial und den Phyto- und Zooplanktongemeinschaften liegen, da hier die ersten Effekte vermutet werden. Lassen sich innerhalb dieser Gemeinschaften erhebliche Veränderungen nachweisen, sollte zusätzlich die Abundanz und Verbreitung von ausgewählten Indikatorarten des Makrozoobenthos und der Fischfauna betrachtet werden.

Abschließend wird auf weitere erforderliche Arbeitsschritte für die Entwicklung des Konzepts verwiesen und die Bedeutung des Deskriptors insbesondere im Hinblick auf den weiteren Ausbau der Offshore-Windparks in der Nordsee und die möglicherweise damit verbundenen grundlegenden Veränderungen des marinen Ökosystems hervorgehoben.

Arbeitspaket 4: Schadstoffe in der Meeresumwelt (Deskriptor 8)

Einer der großen Stressoren in der Meeresumwelt ist die chemische Verschmutzung. Arbeitspaket 4 beschäftigt sich mit Schadstoffen in der Meeresumwelt (Deskriptor 8), wobei die Schwerpunkte auf der Relevanz der Schadstoffe (Kapitel 5.1), Umweltqualitätszielen (Kapitel 5.2), biologischen Effekten (Kapitel 5.3) und den Auswirkungen auf Meeressäuger (Kapitel 5.4) liegen.

Im Unterkapitel 5.1 zielt das Projekt auf eine Eingrenzung der relevanten Substanzen, die unter der WFSD überwacht werden sollen.

Angesichts der Vielzahl von Stoffen und der Komplexität der Auswirkungen wird in dieser Arbeit ein pragmatischer Ansatz gewählt. Die Clusterung der Substanzen erfolgt auf Basis ihrer Verwendungsmuster und die Zusammenstellung der Kriterien für ihre Relevanz auf Basis der verfügbaren Daten. Die relevanten Stofflisten wurden zusammengefasst und konsolidiert. Die resultierende Tabelle bildet die Grundgesamtheit aller betrachteten Substanzen. Die direkt in die Meereseumwelt emittierten Stoffe wurden in die Betrachtungen einbezogen.

Das PEC/PNEC-Verhältnis sollte – wenn vorhanden - als das primäre Kriterium für die Beurteilung einer Substanz herangezogen werden. Zuverlässige Werte für PNEC und PEC in der Meeresumwelt sind jedoch kaum vorhanden. Die weiteren Kriterien sind: Persisitenz und Mobilität eines Stoffes, das Bioakkumulationspotenzial, Toxizität und Potenzial für Transoport über lange Distanzen, kontaminierte Sedimente als sekundäre Quelle, große Produktionsmengen, diffuse Quellen sowie der Status der gesetzlichen Regelungen.

Im Überblick werden als Ergebnis die Gruppen Schwermetalle und zinnorganische Verbindungen, bromierte Flammschutzmittel (BFR), Polyzyklische aromatische Kohlenwasserstoffe (PAK) und die große Gruppe von Industrie- Chemikalien mit Polyflourated Chemicals PFC, Chloralkane und Phenole als prioritär relevant eingestuft, gefolgt von Dioxinen und Bioziden, gefolgt von Pharmazeutika.

Mit dem „CHASE-Tool“ wurde von HELCOM ein Bewertungswerkzeug entwickelt, das der Erreichung der HELCOM-Ziele einen Kontaminierungsstatus (sehr schlecht, schlecht, mittel, gut, hoch) zuweist. Durch den Berechnungsmodus der „contamination ratio“ (CR), mit Division durch die Quadratwurzel, ist der Umweltstatus nicht nur durch die Anzahl der Indikatoren determiniert, sondern es wird die kumulative Wirkung mehrerer kleinerer Belastungen berücksichtigt. Dieses CHASE-Tool, ergänzt durch OSPAR-Bewertungskriterien und geografisch strukturiert, scheint in seiner erweiterten Form als „CHASE 2.0“ ein gutes und praktikables Assessment-Tool zu sein.

Das Unterkapitel 5.2 behandelt mögliche Umweltqualitätsnormen (UQN) von Substanzen mit Relevanz für die marine Umwelt unter besonderer Berücksichtigung der flussgebietsspezifischen Schadstofffe nach WRRL. 28 relevanter Substzanzen im Sinne der MRSL wurden, wie im Unterkapitel 5.1 beschrieben, ausgewählt und vom Umweltbundedsamt um 31 flussgebietsspezifische Schadstofffe ergänzt.

Die Vorgehensweise zur Ableitung mariner UQN für Stoffe, die nach der WRRL reguliert werden sollen, ist im “Technical Guidance for Deriving Environmental Quality Standards” (Guidance Document No. 27, 2011) beschrieben. Je nach Anzahl der Ökotoxizitätsdaten für verschiedene taxonomische Gruppen werden die niedrigsten Süß- und/oder Salzwasser Effektkonzentrationen mit Bewertungsfaktoren verrechnet, um die akzeptable UQN zu erhalten. Für marine Systeme wird ein zusätzlicher Sicherheitsfaktor von 10 veranschlagt, es sei denn, es liegen Informationen für zwei über den Basisdatensatz hinausausgehende marine taxonomische Gruppen vor. Im Unterkapitel 5.2 wird die Frage behandelt, ob für die Ableitung einer marinen UQN die Prozedur des Guidance Document No. 27 angewendet werden kann, oder ob eine Ableitung basierend auf rein marinen Daten zu bevorzugen ist.

Da eine intensive, abgesicherte UQN Ableitung für die 59 Substanzen im Rahmen dieses Projektes nicht möglich war, wurden zu Übersichtszwecken Daten der ECOTOX Datenbank (USEPA, Aquire) verwendet, um eine UQN_{Marin} Ableitung basierend auf Salzwasserorganismen-Daten im Vergleich zu vereinten Süß- und Salzwasserorganismen-Daten vorzunehmen. Generell läst sich sagen, dass nur wenige marine Daten für die ausgewählten Substanzen vorlagen und deshalb eine UQN Ableitung nur mit marinen Daten nicht möglich war. Die Süßwasserorganismen wiesen meist eine höhere Empfindlichkeit auf als die marinen Organismen, möglicherweise bedingt durch die höhere ANzahl an Süßwassereffektdaten. Durch die limiterte Anzahl von Effektdaten für marine Organismen bzw. zusätzlicher taxonomischer mariner Gruppen erscheint das Verfahren nach dem Guidance Document No. 27 unter Verwendung der vereinten Süß- und Salzwasser Effektdaten als die zurzeit am Besten geeignete Methode zu Ableitung von UQN für die Meeresumwelt.

Im Unterkapitel 5.2 werden als Zusatzinformation UQN_{marin} präsentiert, die zurzeit in einem parallellaufenden UBA Projekt (FKZ 3712 28 232) abgeleitet werden.

Im Unterkapitel 5.3 werden die biologischen Effekte der ausgewählten Schadstoffe hinsichtlich deren Eignung für ein Biomonitoring diskutiert. Die Überwachung biologischer Effekte in der Nord- und Ostsee stellt eine große Herausforderung für die Umsetzung der europäischen Meeresstrategie-Rahmenrichtlinie (MSRL) dar. Die Richtlinie erklärt, dass die in der Umwelt vorhandenen Schadstoffkonzentrationen keinen schädigenden Einfluss auf aquatische Organismen haben dürfen. Bioindikatoren sind daher erforderlich, um negative Auswirkungen zu identifizieren und zu verifizieren und gegebenenfalls spezifischen Schadstoffen zuzuordnen. Eine Literaturrecherche wurde durchgeführt, um verfügbare Biomarker zu identifizieren, die vor einer Schädigung der Umwelt warnen und dadurch helfen die Regelungen zur Einleitung von Chemikalien in die Umwelt zu verbessern. Biomarker, die bereits in internationalen Überwachungsprogrammen wie OSPAR/CEMP; ICES/OSPAR und HELCOM zum Einsatz kommen, wurden aufgeführt und mit weiteren potentiellen Markern zur Bestimmung biologischer Effekte verglichen, die sich als Indikatoren für die im Rahmen dieses Projekts identifizierten Umweltschadstoffe anbieten. Die Informationen liefern einen umfassenden Überblick über ausgewählte Kontaminanten und deren biologische Wirkung und werden helfen den Einsatz von Biomarkern im Rahmen der integrierten Bewertung von marinen Ökosystemen gemäß MSRL und Wasserrahmenrichtlinie zu verbessern.

Im Unterkapitel 5.4 werden die Effekte ausgesuchter Schadsubstanzen auf marine Säuger behandelt. In Deutschland sind der Seehund, die Kegelrobbe und der Schweinswal beheimatet. Als Topprädator akkumulieren diese Tiere Schadstoffe in ihrem Körper. In Abhängigkeit der Konzentrationen und der Stoffe können diese akute oder chronische Effekte hervorrufen.

Um die aktuelle Belastungssituation der heimischen marinen Säugetiere besser beurteilen zu können, wird eine Literaturrecherche über bekannte Schadstoffe und deren Effekte bei marinen Säugetieren in der Nord- und Ostsee angefertigt. Im Fokus stehen Organische Substanzen wie PCB, DDT, PFOS, PBDEs, TBT und Dioxin so wie Schwermetalle wie Quecksilber, Cadmium, Arsen und Eisen.

In der Vergangenheit sind einige Studien über diese Stoffgruppen durchgeführt worden, allerdings in verschiedenen Regionen Europas. Das Untersuchungsmaterial (z.B. Blut, Leber, Muskel, Fett) sowie der Untersuchungszeitraum variierten, so dass ein Vergleich zwischen den Regionen und der zeitliche Verlauf schwer zu beurteilen ist. Dennoch können jetzt schon verschiedene pathologische Veränderungen bestimmten Schadstoffen/-gruppen zugeordnet werden.

So sind Schweinswale aus weniger verschmutzen Gewässern gesünder als Tiere aus PCB-belasteten Gebieten. Hohe Belastungen an Organischen Substanzen haben einen negativen Einfluss auf die Fertilität der Tiere. Auch das Immunsystem und Endokrinium bei Walen kann durch Substanzen wie PCB und PBDE geschwächt werden und macht die Tiere dadurch anfälliger für chronische Erkrankungen und Parasitenbefall. Bei Kegelrobben wurden diverse Veränderungen im weiblichen Geschlechtstrakt, Nebennierenhyperplasie, intestinale Ulzera, Arteriosklerose, Glomerulopathie, Hyperkeratose und Krallenverletzungen als „Baltic Seal Disease Complex" zusammengefasst und stehen im Verdacht durch Organochloride wie PCB und DDT ausgelöst zu werden.

Um die Auswirkungen der Schadstoffe auf die Gesundheit der marinen Säugetiere besser bewerten zu können, werden verschiedene Methoden zur Datenerhebung vorgestellt. Wichtig ist hierbei, dass bei der aktuellen Bestimmung der Schadstoffbelastung eine einheitliche Datenerhebung erfolgt, um diese später vergleichen und bewerten zu können. Gerade aktuelle Daten sind nur vereinzelt vorhanden und sollten unbedingt erhoben werden. Vorher ist es nicht möglich die Anforderung der Meeresstrategie-Rahmenrichtlinie zu erfüllen.

Arbeitspaket 5: Abfälle im Meer (Deskriptor 10)

Innerhalb von Arbeitspaket 5 war das Institut für Umweltsystemforschung (USF) der Universität Osnabrück damit befasst, eine Literaturstudie durchzuführen, die auf Literatur über Meeresmüll in der Nord- und Ostsee sowie auf Methoden zur Erfassung seiner Abundanzen fokussierte. Zu diesem Zweck wurden Artikel in wissenschaftlichen Zeitschriften, Doktor- und Masterarbeiten sowie Berichte verschiedener Institutionen und Organisationen ausgewertet.

Weiterhin war das USF beauftragt, Monitoringdaten von Strandmüll auf zeitliche und räumliche Trends hin zu analysieren und ein Bewertungssystem für Strandmüll zu entwickeln, das auf europäische Meeresgewässer anwendbar ist. Dazu wurden Strandmüllmonitoringdaten in enger Kooperation mit David Fleet (Landesamt für Küsten- und Naturschutz Schleswig-Holstein (LKN-SH) Tönning) von der OSPAR Intersessional Correspondence Group Marine Litter (OSPAR ICG ML) (OSPAR, 2009) und von deutschen gemeinnützigen Umweltverbänden akquiriert. Letztere haben an mehreren Stränden ein Strandmüllmonitoring über teilweise mehr als 20 Jahre durchgeführt (Clemens et al., 2011). Die Daten wurden mit univariaten und multivariaten statistischen Methoden analysiert. Die Analysenergebnisse wurden dazu verwendet, eine Klassifikation von Stränden gemäß Deskriptor 10 der Meeresstrategie-Rahmenrichtlinie (MSRL) vorzunehmen. Basierend auf dieser Klassifikation wurde in einem nächsten Schritt ein multikriterielles Bewertungssystem erstellt. Weiterhin wurden neuronale Netze zur Simulation von Strandmüllzeitreihen entwickelt.

Neben der Auswertung von Strandmülldaten hatte das USF die Aufgabe, weitere Langzeitdaten von Meeresmüll auf der Wasseroberfläche und in der Wassersäule zu eruieren, diese Daten zu analysieren und weitere Methoden vorzuschlagen, wie Meeresmülldaten erhoben werden können. Einige Publikationen aus der diesbezüglichen Literaturstudie, welche die Beobachtung von Meeresmüll auf der Wasseroberfläche und seine vertikale Verteilung in der Wassersäule thematisieren, werden zusammengefasst. Daten zu räumlichen Verteilungen von treibendem Meeresmüll und Müll am Meeresgrund wurden akquiriert, analysiert sowie miteinander und mit Strandmülldaten korreliert.

Außerdem erfolgten methodische Arbeiten zur Optimierung und Standardisierung der Analytik von Mikroplastik in Strandproben. Erste Analysenergebnisse und Zählungen von Mikroplastik werden vorgestellt und in Beziehung zu früheren Studien über Mikroplastik gesetzt.

Zur Untersuchung der Auswirkungen von Müll auf marines Leben wurden Trends der Mengen und der Zusammensetzung von Müll, der von Meerestieren verschluckt wird (Indikator 10.2.1), mittels Mageninhaltsanalysen untersucht. In diesem Zusammenhang wurde der deutsche Beitrag zum laufenden OSPAR Fulmar Litter EcoQO-Ansatz im Rahmen des aktuellen Vorhabens weitergeführt.

Der Eissturmvogel stellt eine besonders geeignete Indikatorart dar, da er sich ausschließlich auf See ernährt und dabei verschiedenste Plastikobjekte aufnimmt, die sich in seinem Magen ansammeln. Darüber hinaus ist der Eissturmvogel eine häufige Art in der Nordsee, im Nordatlantik und Nordpazifik und wird häufig im Rahmen von Totfundsammlungen entlang der Küsten gefunden. Im Nordseebereich werden die Mageninhalte von Eissturmvögeln seit 2002 auf Plastikbelastung hin untersucht. Die Koordination der internationalen Arbeiten sowie der vorangegangenen Pilotstudie liegt beim niederländischen IMARES Institut. Aus diesen Untersuchungen ging die Definition des Fulmar Litter EcoQO (Ökologisches Qualitätsziel) hervor, das von OSPAR implementiert wurde. Das Fulmar Litter EcoQO wird erreicht, wenn weniger als 10% der gefundenen Eissturmvögel eine Plastikbelastung von über 0,1 g im Magen aufweisen. Dieses vorgegebene Niveau orientiert sich an der Müllbelastung eines Referenzgebiets, dessen Umweltzustand als akzeptabel angesehen wird.

In Deutschland werden die Arbeiten seit 2002 vom FTZ in Kooperation mit einem Netzwerk von haupt- und ehrenamtlichen Unterstützern durchgeführt, die die Strandfunde einsammeln. Die Methoden der Sektionen und Mageninhaltsanalysen sind international standardisiert. Im Rahmen des aktuellen Vorhabens wurden die Mägen von 37 im Jahr 2011 und 52 in 2012 an deutschen Küsten gefundenen Eissturmvögeln zusammen mit verschiedenen zusätzlichen Proben aufbereitet und in Kombination mit Daten früherer Untersuchungen analysiert.

Aktuell, d.h. über die Periode der letzten 5 Jahre (2008-2012) gesehen, sind die Mägen von 96% der 235 in Deutschland gefundenen Eissturmvögeln mit Plastikmüll belastet. Durchschnittlich ist jeder Vogel mit 22,6 Plastikobjekten belastet, die zusammen 0,34 g wiegen. Die zwei Haupt-Plastikkategorien, Industrieplastik und Verbraucherplastik, waren in 53% bzw. 96% der Eissturmvögel vertreten. Das Verbraucherplastik setzte sich hauptsächlich aus Fragmenten zusammen, die in 91% der Vögel gefunden wurden und für knapp die Hälfte der Gesamtplastikmasse verantwortlich zeichneten. Daneben zählten Schaumstoffe (v.a. Polystyrol) zu den vorherrschenden Bestandteilen des Verbraucherplastiks. Seit Studienbeginn stieg die durchschnittliche Masse des Verbraucherplastiks von 0,29 g pro Vogel (2003-2007) in Deutschland auf 0,36 g (2007-2011) an, ging in der aktuellen Periode jedoch wieder auf den Anfangswert von 0,29 g zurück (2008-2012). Im Gegensatz dazu blieb das Belastungsniveau durch Industrieplastik über die gesamte Zeit mit ca. 0,05 g stabil. Die Zusammensetzung des gefundenen Plastiks ähnelt der von niederländischen Tieren. Die beobachtete hohe Plastikmüllbelastung beeinträchtigt die Körperkondition der Vögel zwangsläufig sowohl mechanisch als auch chemisch.

Das Fulmar Monitoring Tool, das den OSPAR Fulmar Litter EcoQO Ansatz nutzt, stellt bisher den einzigen vollständig entwickelten MSRL Indikator für verschluckten Müll dar. Es umfasst den umfangreichsten Datensatz der EU mit Fokus auf Trends von verschlucktem Müll und dessen Auswirkungen. Es kann im Großteil des Nordostatlantiks angewendet und zur Untersuchung von zeitlichen und regionalen Trends verschiedener Müllkategorien verwendet werden. Folglich wurde es als Indikator für den GES („Good Environmental Status“ = guter Umweltzustand) der europäischen MSRL übernommen.

Im Teilprojekt „Trends des makroskopischen Mülls auf dem Meeresboden und ghost nets“ wurde der Indikator menschliche Abfälle auf dem Meeresboden genutzt um den Umweltstatus zu beschreiben. Zusätzlich wurden die räumliche Verteilung und die Abundanz von verfangenen Fischereinetzen an Schiffwracks untersucht um die schädlichen Effekte des potentiellen Stressfaktors „Geisternetze“ zu untersuchen.

Für die Abschätzung der Müllmengen auf dem Meeresgrund der Nordsee wurden Fischereihols des AWIs ausgewertet. Das Alfred-Wegner-Institut in der Helmholtz-Gemeinschaft für Polar- und Meeresforschung hat zwischen den Jahren 1998 und 2010 in der deutschen AWZ und dem Wattenmeer der Nordsee regelmäßig Baumkurren- und/oder Grundschleppnetzfänge durchgeführt, um die Entwicklung des Benthos über einen langen Zeitraum hinweg beschreiben zu können. In den Protokollen dieser Forschungs-Fischereihols wurden auch die gefangenen Müllmengen erfasst. In der vorliegenden Studie wurden diese Daten dazu genutzt, um die Menge und die räumliche Verteilung des Mülls am Meeresboden in der Deutschen Bucht abzuschätzen und die Entwicklung der Müllmengen über die Zeit zu beschreiben. Anhand der Ergebnisse wurden Methoden und Gebiete für ein langfristiges Monitoring zur Entwicklung der Müllmengen am Meeresboden empfohlen und ein Vorschlag dazu unterbreitet welcher Grad der Müllverschmutzung mit welchen Abfallarten als „Guter (zu erreichender) Zustand“ des Meeresbodens im Sinne der MSRL angesehen werden sollte. Vorsichtige Schätzungen über alle Baumkurrenfänge ergeben für die Fläche der deutschen AWZ eine durchschnittliche Müllbelastung von 10 kg km^{-2}. Für die komplette Fläche der deutschen AWZ von 28.539 km^2 ergibt sich somit eine durch Menschen verursachte Gesamtmüllmenge von 285.390 kg. Vier Untersuchungsgebiete erlaubten eine zeitliche Auflösung des Beprobungszeitraumes.

Das BSH führt an den Schiffswracks in der AWZ Inspektionstauchgänge durch um den Zustand der Wracks zu dokumentieren. Die Videoaufzeichnungen der Inspektionstauchgänge bieten die Möglichkeit, eine große Anzahl Wracks hinsichtlich der Abundanz von „Ghost nets“ ohne den Einsatz eigener Forschungstaucher zu untersuchen.

In der vorliegenden Studie wurde anhand dieser Inspektionsvideos an 64 Wracks

- die Prozentuale Bedeckung der Wracks in der AWZ und die Art der darn verfangenen Netze an Schiffswracks bestimmt,
- die räumliche Verteilung der „Wrack ghost nets“ in der AWZ dargestellt
- die gesamt an den Schiffswracks in der Deutschen Bucht vorhandene Fläche der „Ghost nets“ abgeschätzt und
- eine Auswahl von Wracks erarbeitet, an denen die direkte Schadwirkung von „Ghost nets“ untersucht werden könnte.

Die durchschnittliche Fläche der an einem Wrack vorhandenen Netze betrug 37,4±65,5 m^2. Extrapoliert auf die Anzahl von rund 1.300 Wracks in der Nordsee bedeutet dies, dass 48.600 m^2 stationäre „Ghost nets“ in der deutschen Bucht vorhanden sind. Eine regelmäßige Auswertung von Netzbedeckungen an Wracks wird empfohlen.

Arbeitspaket 6: Zusammenfassung der Belastungen zur Gesamtbewertung (Integrierte Ökosystembewertung)

Das Konzept der MSRL ist ein ganzheitlicher Ansatz, der die Bewertung der europäischen marinen Ökosysteme umfasst. Allgemein kann eine solche integrierte Gesamtbewertung auf verschiedene Arten vorgenommen werden.

- Die einfachste Methode stellt der „One-Out, All-Out“ (OOAO, auf Deutsch: „Einer-raus, Alle-raus“) - Ansatz dar. Das bedeutet, dass der Gesamtstatus durch den schlechtesten Status der in der Bewertung verwendeten Komponenten bestimmt wird.
- Eine bekannte und gängige Methode der Aggregation von Daten zur Bewertung ist die Mittelwertbildung und Gewichtung.

- Um Interaktionen zwischen Auswirkungen und Ökosystemkomponenten besser zu erschließen, können diese in übersichtlicher Form in einem Entscheidungsbaum oder in einer Entscheidungs-Matrix dargestellt werden.
- Bei der Bewertung eines Gesamtstatus des Ökosystems werden häufig Indizes verwendet, die bestimmte Zustände des Ökosystems widerspiegeln (trophische Zustände, Belastungen für bestimmte Indikatororganismen u.a.). Für einen umfassenden Ansatz zur Ökosystembewertung müssen allerdings die kumulativen Effekte berücksichtigt werden.

Über die genannten Methoden hinaus sind zahlreiche Kombinationen verschiedener Methoden möglich. Ein Beispiel ist die Kombination des OOAO-Ansatzes und der gewichteten Mittelwertbildung mit einem Entscheidungsbaum, wie sie in der WRRL angewendet wird.

Im vorliegenden Bericht werden verschiedene Möglichkeiten einer Integrierten Ökosystembewertung für die Implementierung im Rahmen der MSRL vorgeschlagen.

Für die Klassifizierung einer bestimmten Meeresregion würde eine integrierte Gesamtbewertung aus 12 verschiedenen Einzelbewertungen (11 Deskriptoren und die kumulativen Effekte) bestehen, die aggregiert werden müssten. Um dieses Ziel zu erreichen, können drei verschiedene konzeptionelle Ansätze verwendet werden. Eine Voraussetzung für die Anwendung des Ansatzes ist die Entwicklung eines individuellen Bewertungssystems für jeden Deskriptor.

- Der einfachste Ansatz für eine Integrierte Ökosystembewertung wäre die Berechnung von gewichteten Mittelwerten für die 12 Komponenten. Anstelle des zweistufigen Klassifizierungssystems der MSRL sollte eine fünfstufige Bewertungsskala für die Ergebnisse angewendet werden, um Informationen bezüglich möglicher Trends im Ökosystem zu erhalten.
- Der zweite Ansatz ist eine Kombination des OOAO-Ansatzes und der gewichteten Bewertung. Hierbei gilt es zu überlegen, ob die absolute und schematische Einstufung durch den OOAO-Ansatz dadurch abgemildert werden könnte, dass ein oder zwei schlechte Bewertungen bei den 12 Komponenten zulässig sein könnten, ohne die mögliche Gesamteinstufung des Ökosystems in einen „Guten Zustand“ zu verlieren.
- Eine alternativ vorgeschlagene Methode basiert auf einer Zusammenfassung der Indikatoren, die im EU-Kommissionsbeschluss der MSRL beschrieben sind. Dabei werden die Indikatoren in zwei Gruppen unterteilt: die Statusvariablen, wie Habitatgröße, -verbreitung, -zustand, Ökosystemstruktur und -funktion, Biodiversität u.a., sowie die Belastungsvariablen, wie Nährstoff- und Schadstoffbelastungen sowie Müll- und Lärmeffekte.

Der ökologische Zustand wird nicht nur durch die Intensität der anthropogenen Belastungen definiert, sondern auch durch die Interaktionseffekte zwischen den verschiedenen Aspekten des Ökosystems, die auf eine mögliche Störung des ökologischen Gleichgewichts hinweisen können. Eine solche Beschreibung des ökologischen Zustandes kann durch die Anwendung eines integrierten Matrixsystems erhalten werden. Eine solche Matrix besteht aus sämtlichen Arten und Artengruppen, die durch die Bewertung berücksichtigt werden sollen, sowie Aspekten wie Ökosystemstruktur, Ökosystemfunktion, Biodiversität und ggf. sozioökonomische Aspekten. Außerdem werden kumulative Effekte wie beispielsweise Interaktionseffekte, zeitliche und räumliche Effekte eng mit dem Ergebnis der Indikatoren und Kriterien verknüpft, die die Aspekte des Ökosystems definieren, auf denen die integrierte Matrix basiert.

1 Introduction

On 17th June 2008, the guideline for establishing a framework for Community action in the field of marine environment (Marine Strategy Framework Directive - MSFD, 2008/56/EG) was published. The overall objective of the guideline is to achieve and/or maintain a good status of the marine environment before the year 2020. The good environmental status has to be defined in accordance with qualitative Descriptors as listed in Annex I and specified through respective Criteria and Indicators given by the European commission. For the definition of the good environmental status, the ecosystem approach will have to be taken into account. Particular marine regions, such as the northeast Atlantic including the North Sea as sub-region, and the Baltic Sea, have been identified as management units. All member states have been asked to establish a national action plan for their marine waters with the overall aim to reach or maintain the good status of the marine environment as well as the contributing environmental objectives.

Central steps to implement the MSFD will be:

- Carrying out an initial assessment in order to map the current environmental status as well as the anthropogenic impact on this status
- Description of the good environmental status
- Definition of environmental objectives being guidelines for reaching the good environmental status, and of respective indicators for each marine region based on the initial assessment
- Establishing suitable monitoring programmes for the running assessment and continuously updating the objectives
- Revisions of the assessment of the national marine regions at regular intervals of 6 years

As far as possible, the member states shall rely on existing programmes and measures being developed with the frame of regional marine conventions such as OSPAR and HELCOM.

The initial assessment includes the analysis of main characteristics of the current environmental status of the assessed waters and covers the respective physical and chemical features, the habitat types, as well as the biological features and the hydromorphology. Moreover, the most important impacts and effects, also of anthropogenic origin, on the environmental state of the targeted waters will be analysed and qualitative and quantitative aspects of the different impacts as well as detectable trends covered. Furthermore, the most important cumulative and synergetic effects on the marine environment will have to be covered. In addition, the initial assessment requires socio-economic analyses of the usage of the marine waters as well as of the costs of a degradation of the system. For identifying the current status of the region/sub-region, the Marine Strategy Framework Directive requires the application of coherent assessment systems for the both the whole marine region and the sub-regions. Also, cross-border effects and characteristics will have to be considered.

The MSFD reports for the initial assessment, for the Good Environmental Status (GES) and for the environmental targets have already been published within the frame of previous projects. For most of the characteristics, impacts and effects listed in Annex III of the MSFD, the German national initial assessment consists mainly of a comprehensive compilation of individual parameters having already been reported within other reporting tasks and being very often spatially restricted. With regard to the description of the good environmental status, existing definitions of environmental targets from other guidelines, e.g. the EG Water Framework Directive, the German Flora-Fauna-Habitat-Richtlinie, OSPAR and HELCOM, have been used. So far, it has not been possible to achieve an integrated assessment for all the Descriptors of the MSFD. The environmental targets stand for specific qualitative and quantitative requests, referring to partial steps in the direction of a good environmental status.

While recent projects have focussed mainly on the so-called ‘state’ Descriptors of the MSFD, the focus of the current project has been on ‘pressure’ and ‘impact’ Descriptors. For these, assessment systems were not yet available, such as D2 (non-indigenous/’invasive’ species), D6 (sea-floor integrity), D7 (hydrographical conditions), D8 (contaminants) and D10 (marine litter). Within in the project, we have been identifed existing deficits and present possible solutions, for example by developing respective assessment systems. Moreover, an overall assessment concept for ’the good environmental status’ according to the MSFD has been developed, with special regard to the results of recent MSFD projects. This work has also been included the examination and specification of the MSFD Indicators of the individual impact Descriptors. Further aim of the project has been the definition of quantitative environmental targets and subsequent operationalization of the respective Indicators.

Based on the recent assessment work and on existing monitoring programmes, we will define a monitoring concept with the objective of effectively ensuring the accomplishment of the future assessment work.

2 Work package 1: Non-indigenous species (Descriptor 2)

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2.1 Introduction

Non-indigenous species are listed as the separate 'impact' Descriptor 2 of the Marine Strategy Framework Directive (MSFD) and have to be individually assessed and included into the overall assessment.

Within this study, existing definitions for non-indigenous species will be reviewed and a feasible definition elaborated. There are several approaches of assessing the status of an ecosystem by the means of non-indigenous species. These classification systems will be analysed for their applicability to the German marine areas with regard to the implementation of the MSFD. For that purpose, different organism groups of the North Sea and Baltic Sea will be evaluated. A concept for adapting, extending and supplementing existing approaches will be elaborated and first quantitative environmental targets for individual indicators to be applied in a suitable classification system be proposed. This will be a further step in the direction of developing a practicable and feasible assessment system within the frame of the MSFD. Finally, a monitoring concept for non-indigenous species within the German marine areas will be set up and checked whether this could also be used as an early warning system, for example in harbour areas.

2.2 Non-indigenous species in marine aquatic systems

All over the world, non-indigenous species are detected in marine aquatic ecosystems. Although species have been transported by human activities all the time, the rate of introducing new species to existing habitats is increasing significantly. Due to a changing climate, species may also establish and spread in environments, which did not offer optimum ecological conditions in the past (Walther et al., 2009; Nehring, 2003 b). Currently, there are more than 90 non-indigenous species counted for the North Sea and Baltic Sea of that 53 and 29 are presumed as established, respectively. Because of their potential effect on native species and the ecosystem structure and functioning, they may influence the ecological status of an ecosystem. Therefore, non-indigenous species have to be implemented into the assessment of the 'Good Environmental Status' (GES) within the Marine Strategy Framework Directive.

2.2.1 Definition

New species that are introduced in native ecosystems by human action and not by natural spreading are known by many different terms as 'non-indigenous species', 'non-native species', 'aliens' or 'alien species', 'neobiota', 'exotic species', 'introduced species', 'invasive species' and possibly more (e.g. Panov et al., 2009; Nehring et al. 2009; Genovesi and Shine, 2003; Secretariat of the Convention on Biological Diversity, 2001; Gollasch, 1997; Nehring, 2005; Walther et al., 2009). The term 'invasive' refers to the intrusion of the species into the new ecosystem and is used especially for the non-indigenous species that have an impact on the native populations, species communities, habitats or ecosystem structure and functioning. In addition to the term 'invasive species', the word 'invasive' is also added to one of the other synonyms, resulting in terms such as 'invasive alien species', to describe a species having an effect on its new ecosystem. However, the terms are used with different definitions. Thus, it is not clear if a species has to reproduce and establish in the new environment, before the mentioned terms can be applied. While reproduction is included in most definitions, the establishment is not always mandatory. When species are released into a new region without significant detectable impact, they are suggested to be called 'inoculations' (Carlton, 2003). When it is not known, whether a species is native or introduced, it will be referred to as 'cryptogenic'. Collected from several publications a variety of terms and definitions which are related to the field of non-indigenous species are presented in Table 2-1.

Table 2-1: Collection of terms and definitions related to the field of non-indigenous species.

Term	Definition
Alien	An organism occurring outside its natural past or present range and dispersal potential, whose presence and dispersal is due to intentional or unintentional human action. (Walther et al., 2009)
Alien species	Refers to a species, subspecies, or lower taxon, introduced outside its natural past or present distribution; includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce. (Panov et al., 2009)
Alien species	A species, including genetically distinct populations, occurring outside of its natural range (past or present) and dispersal potential (i.e. outside the range it occupies naturally or could not occupy without direct or indirect introduction or care by humans); includes any part, gametes or propagules of such species that might survive and subsequently reproduce. (Nehring et al., 2009)
Alien species	A species, subspecies or lower taxon, introduced outside its natural past or present distribution; includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce. (Genovesi and Shine, 2003)
Alien species	see "Introduced species". (GESAMP, 1997)
Alien species	A species, subspecies, or lower taxon introduced outside its normal past or present distribution; includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce. Synonyms: foreign, exotic, introduced, new, non-indigenous, non-native, neophytes, neozoans. (Nehring, 2005)
Alien species	As those "whose introduction does or is likely to cause economic or environmental harm or harm to human health". (Moser and Leffler, 2010)
Alien species	... (non-native, non-indigenous, foreign, exotic) means a species, subspecies, or lower taxon occurring outside of its natural range and dispersal potential (i.e. outside the range it occupies naturally or could not occupy without direct or indirect introduction or care by humans) and includes any part, gametes or propagule of such species that might survive and subsequently reproduce. (Hopkins, 2001)
Alien species	.. (synonyms: non-native, non-indigenous, foreign, exotic) is a species, subspecies, or lower taxon introduced outside its normal past or present distribution; includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce. (Secretariat of the Convention on Biological Diversity, 2001)
Casual	…refers to organisms that do not form self-replacing populations and rely on repeated introductions for their persistence. (Walther et al., 2009)
Casual alien species	Alien species that may flourish and even reproduce occasionally in an area, but which do not form self-replacing populations, and which rely on repeated introductions for their persistence. (Secretariat of the Convention on Biological Diversity, 2001)

Term	Definition
Casual alien species	Alien species that may flourish and even reproduce occasionally in an area, but which do not form self-replacing populations, and which rely on repeated introductions for their persistence. (Nehring, 2005)
Cryptogenic	A term used for species of unknown origin or means of arrival, which cannot be ascribed as being native or alien. (Walther et al, 2009)
Cryptogenic species	…is a species that is not demonstrably native or introduced. (GESAMP, 1997)
Cryptogenic species	A species that is not demonstrably native or introduced. (Nehring, 2005)
Cryptogenic species	…is a species that is not demonstrably native or introduced. (Hopkins, 2001)
Cryptogenic species	…are those of unknown origin which cannot be ascribed as being native or alien. (Olenin et al., 2010)
Established species	Species occurring as a reproducing, self-sustaining population in an open ecosystem, i.e. in waters where the organisms are able to migrate to other waters. (GESAMP, 1997)
Established species	…are species occurring as a reproducing, self-sustaining population in an open ecosystem, i.e. in waters where the organisms are able to migrate to other waters. (Hopkins, 2001)
Establishment	…refers to the process of an alien species in a new habitat successfully producing viable offspring with the likelihood of continued survival. (Panov et al., 2009)
Establishment	The process of an alien species in a new habitat successfully producing viable offspring with the likelihood of continued survival. (Genovesi and Shine, 2003)
Establishment	The process of a species in a new habitat successfully reproducing at a level sufficient to ensure continued survival without infusion of new genetic material from outside the system. (Secretariat of the Convention on Biological Diversity, 2001)
Establishment	The process of a species in a new habitat successfully reproducing at a level sufficient to ensure continued survival without infusion of new genetic material from outside the system. (Nehring, 2005)
Exotic species	see "Introduced species" (GESAMP, 1997)
gebietsfremde Art	Eine wildlebende Tier- oder Pflanzenart, wenn sie in dem betreffenden Gebiet in freier Natur nicht oder seit mehr als 100 Jahren nicht mehr vorkommt. (Probleme bei der Wiederansiedlung von Arten, die vor mehr als 100 Jahren verdrängt wurden). (Hubo et al., 2007)
heimische Art	Eine wild lebende Tier- oder Pflanzenart, die ihr Verbreitungsgebiet oder regelmäßiges Wanderungsgebiet ganz oder teilweise a. im Inland hat oder in geschichtlicher Zeit hatte oder b. auf natürliche Weise in das Inland ausdehnt; als heimisch gilt eine wild lebende Tier- oder Pflanzenart auch, wenn sich verwilderte oder durch menschlichen Einfluss eingebürgerte Tiere oder Pflanzen der betreffenden Art im Inland in freier Natur und ohne menschliche Hilfe über mehrere Generationen als Population erhalten“ besonderer Schutz auch für bestimmte IAS möglich, da sie als heimisch definiert sind! (Hubo et al., 2007)
Incidental species	...are alien species that have been introduced through human agency into a new area, but have not become established in the wild. (Hopkins, 2001)

Term	Definition
Indigenous (= native) species	A species or lower taxon living within its natural range (past or present) including the area which it can reach and occupy using its natural dispersal systems (modified after CBD, GISP). (ICES, 2005)
Inoculation	Species are not considered as being introduced to a region if they are simply re-leased into a new region (such releases are inoculations) and there is no evidence of reproduction or establishment. (Carlton, 2003)
Intentional introduction	Deliberate transfer and/or release by humans of a species or genetically distinct population outside its natural range (past or present) and dispersal potential (such introductions may be authorised or unauthorised); this includes also species which subsequently escape or which are released into the environment. (Nehring et al., 2009)
Intentional introduction	The deliberate movement and/or release by humans of an alien species outside its natural range. (Genovesi and Shine, 2003)
Intentional introduction	The purposeful movement by humans of a species outside its natural range and dis-persal potential (such introductions may be authorized or unauthorized) (IUCN, 2000) (c.f. unintentional introduction). (Secretariat of the Convention on Biological Diversity, 2001)
Intentional introduction	...is a deliberately made introduction by humans, involving the purposeful transport of a species or subspecies (or propagules thereof) outside its natural range. Such in-troductions may be either authorised or unauthorised. (GESAMP, 1997)
Intentional introduction	The purposeful movement by humans of a species outside its natural range and dis-persal potential (such introductions may be authorised or unauthorised). (Nehring, 2005)
Intentional introduction	...means an introduction made deliberately by humans, involving the purposeful movement of a species outside of its natural range and dispersal potential (Such in-troductions may be authorized or unauthorized). (Hopkins, 2001)
Introduced species	(= alien species, = exotic species, non-indigenous species) Any species intentionally or accidentally transported and released by humans into an environment outside its present range. (GESAMP, 1997)
Introduced species (= non-indigenous species, = exotic species)	Any species transported intentionally or accidentally by a human-mediated vector into aquatic habitats outside its native range. Note: Secondary introductions can be transported by human-mediated or natural vectors. (ICES, 2005)
Introduction	Refers to the movement by human agency, indirect or direct, of an alien species outside its natural range (past or present); this movement can be either within a country or between countries or areas beyond national jurisdiction. (Panov et al., 2009)
Introduction	The transfer, by direct or indirect human agency, of a species or genetically distinct population outside of its natural range (past or present) and dispersal potential; this movement can be either within a country or between countries or areas beyond na-tional jurisdiction. Human involvement here does not include habitat changes, global warming, eutrophication, etc. (Nehring et al., 2009)
Introduction	The movement by human agency, indirect or direct, of an alien species outside of its natural range (past or present). This movement can be either within a country or be-tween countries or areas beyond national jurisdiction. (Genovesi and Shine, 2003)

Term	Definition
Introduction	The movement, by human agency, of a species, subspecies, or lower taxon (including any part, gametes, seeds, eggs, or propagule that might survive and subsequently reproduce) outside its natural range (past or present). This movement can be either within a country or between countries (IUCN, 2000). (Secretariat of the Convention on Biological Diversity, 2001)
Introduction	An introduction of an organism is the dispersal, by human agency, of a living organism outside its historically known range. (GESAMP, 1997)
Introduction	The movement, by human agency, of a species, subspecies, or lower taxon (including any part, gametes, seeds, eggs, or propagules that might survive and subsequently reproduce) outside its natural range (past or present). This movement can be either within a country or between countries. (Nehring, 2005)
Introduction	...means the movement, by human agency, of a species, subspecies, or lower taxon (including any part, gametes or propagule that might survive and subsequently reproduce), outside its historically known natural range, within the same country or in another country. (Hopkins, 2001)
Introduction	...refers to a deliberate or accidental transfer or release of organisms into the open environment by human activities across natural barriers of dispersal, refers to the movement of organisms. (Reise et al., 2006)
Introduction / introduced	Direct or indirect movement by human agency, of an organism outside its past or present natural range (Walther et al., 2009)
Invasibility	The probability of establishment of alien species as a complex function of abiotic and biotic resistance by the ecosystem to introductions under a specific level of propagule pressure. (Panov et al., 2009)
Invasion	...is used for any process of colonization and establishment beyond a former range, particularly in which a species plays a conspicuous role in the recipient ecosystems, addresses to the occupation process with ecological interactions and evolutionary changes. (Reise et al., 2006)
Invasion / invasive	...refers to established alien organisms that are rapidly extending their range in the new region. (This is usually associated, although not necessarily for an organism to qualify as invasive, with causing significant harm to biological diversity, ecosystem functioning, socio-economic values and human health in invaded regions). (Walther et al., 2009)
Invasive alien species	An alien species which is known or expected to exert effects on native populations and species, natural habitats and ecosystems beyond of which can be considered to be within the range of average regional conditions. (Nehring et al., 2009)
Invasive alien species	An alien species whose introduction and/or spread threatens biological diversity. (Genovesi and Shine, 2003)
Invasive alien species	An alien species whose establishment and spread threaten ecosystems, habitats or species with economic or environmental harm. These are addressed under Article 8(h) of the CBD. (Secretariat of the Convention on Biological Diversity, 2001)
Invasive alien species	An alien species whose establishment and spread threaten ecosystems, habitats or species with economic or environmental harm. These are addressed under Article 8(h) of the CBD. (Nehring, 2005)

Term	Definition
Invasive alien species	...(IAS) are a subset of established NIS which have spread, are spreading or have demonstrated their potential to spread elsewhere, and have an adverse effect on biological diversity, ecosystem functioning, socio-economic values and/or human health in invaded regions. Species of unknown origin which can not be ascribed as being native or alien are termed cryptogenic species. They also may demonstrate invasive characteristics and should be included in IAS assessments. (Olenin et al., 2010)
Invasive alien species (IAS)	...are a subset of established NIS and/or cryptogenic species which have spread, are spreading or have demonstrated their potential to spread elsewhere, and have an adverse effect on biological diversity, ecosystem functioning, socio-economic values and/or human health in invaded regions. (Olenin et al., 2010)
Invasive species	...means an alien species which becomes established in natural or semi-natural ecosystems or habitat, is an agent of change, and threatens native biological diversity. (Hopkins, 2001)
Invasiveness	The degree to which an organism is able to spread from site of primary introduction, to establish a viable population in the ecosystem, to negatively affect biodiversity on the individual, community, or ecosystem level and cause adverse socioeconomic consequences. (Panov et al., 2009)
Native	An organism that has originated in a given area without human involvement or that has arrived there without intentional or unintentional intervention of humans (Walther et al, 2009)
Native range	Natural limits of geographical distribution of a species. (Nehring, 2005)
Native range	Natural limits of geographical distribution of a species. (ICES, 2005)
Native species	A species, including genetically distinct populations, occurring within its natural range (past or present) and dispersal potential (i.e. within the range it occupies naturally or could occupy without direct or indirect introduction or care by humans). (Nehring et al., 2009)
Native species	...is a species, subspecies or lower taxon, occurring within its natural range and dispersal potential (i.e. within the range it occupies naturally or could occupy without direct or indirect introduction by humans). (GESAMP, 1997)
Native species	A species, subspecies, or lower taxon living within its natural range (past or present), including the area which it can reach and occupy using its own legs, wings, wind/waterborne or other dispersal systems, even if it is seldom found there. Synonym: indigenous. (Nehring, 2005)
Native species	...(indigenous) means a species, subspecies, or lower taxon, occurring within its natural range and dispersal potential (i.e. within the range it occupies naturally or could occupy without direct or indirect introduction or care by humans). (Hopkins, 2001)
Native species (synonym: indigenous species)	...a species, subspecies, or lower taxon living within its natural range (past or present), including the area which it can reach and occupy using its own legs, wings, wind/water-borne or other dispersal systems, even if it is seldom found there. (Secretariat of the Convention on Biological Diversity, 2001)
Naturalization	...refers to aliens that form free-living, self-sustaining (reproducing) and durable populations persisting in the wild (Walther et al, 2009)

Term	Definition
Naturalized species	Alien species that reproduce consistently (cf. casual alien species) and sustain populations over more than one life cycle without direct intervention by humans (or in spite of human intervention); they often reproduce freely, and do not necessarily invade natural, semi-natural or human-made ecosystems. (Secretariat of the Convention on Biological Diversity, 2001)
Naturalized species	Alien species that reproduce consistently (cf. casual alien species) and sustain populations over more than one life cycle without direct intervention by humans (or in spite of human intervention); they often reproduce freely, and do not necessarily invade natural, semi-natural or human-made ecosystems. (Nehring, 2005)
Neozoon	...is an animal species, that reached, after the year 982 AD (first introduction of American organisms in Europe; trans-Atlantic cruise of Eric the Red), under trans-Atlantic cruise of Eric the Red), under direct or indirect anthropogenic involvement, a specific area and has lived there wildly for at least three generations (= established reproduction community) or over a longer period (at least 25 years) up to now. (Nehring and Leuchs, 2000)
Neozoon incertum	In the case of species where direct or indirect anthropogenic involvement for occurrence (e.g. in the area probably always existent), and/ or the current setup of a reproducing population is to be strongly doubted, the term “neozoon incertum” (plural: neozoa incerta) can be used. (Nehring and Leuchs, 2000)
Neozoon simulatum	It is an animal species that appears without recognizable connection with human activities in the appropriate area and also reproduces (= natural expansion of the area). (Nehring and Leuchs, 2000)
New introduction	The human-mediated movement of a species outside its present distribution. (ICES, 2005)
Non-indigenous	see "Introduced species". (GESAMP, 1997)
Non-indigenous species	...(NIS; synonyms: alien, exotic, non-native, allochthonous) are species, subspecies or lower taxa introduced outside of their natural range (past or present) and outside of their natural dispersal potential. This includes any part, gamete or propagule of such species that might survive and subsequently reproduce. Their presence in the given region is due to intentional or unintentional introduction resulting from human activities. Natural shifts in distribution ranges (e.g. due to climate change or dispersal by ocean currents) do not qualify a species as a NIS. However, secondary introductions of NIS from the area(s) of their first arrival could occur without human involvement due to spread by natural means. (Olenin et al., 2010)
Non-indigenous species (NIS)	(synonyms: alien, exotic, non-native, allochthonous) these are species, subspecies or lower taxa introduced outside of their natural range (past or present) and outside of their natural dispersal potential. This includes any part, gamete or propagule of such species that might survive and subsequently reproduce. Their presence in the given region is due to intentional or unintentional introduction resulting from human activities, or they have arrived there without the help of people from an area in which they are alien. (Olenin et al., 2010)
Nonnative, nonindigenous species	Synonyms for ‘‘alien species’’ (Panov et al., 2009)
Non-target species	Any species inadvertently accompanying in, on, or with the species intended for introduction or transfer. (Nehring, 2005)

Term	Definition
Non-target species	Any species inadvertently accompanying in, on, or with the species intended for introduction or transfer. (ICES, 2005)
Pest	"Any species, strain or biotype of plant, animal or pathogenic agent injurious to plants or plant products" (IPPC). (Secretariat of the Convention on Biological Diversity, 2001)
Pest	Any species, strain or bio-type of plant, animal or pathogenic agent (not necessarily alien) injurious to plants or plants products. (Nehring, 2005)
Pests	...are harmful organisms (not necessarily alien) living in places where they are unwanted and have a detectable environmental and/or economic impact or impact on human health. Pests may be native, cryptogenic or alien species. (Olenin et al., 2010)
Re-introduction	...means an attempt to establish a species in an area which was once part of its historical range, but from which it has been extirpated or become extinct. (Hopkins, 2001)
Release	Voluntary or accidental dissemination of an organism, or its gametes, outside its controlled area of confinement. (Nehring, 2005)
Release	Voluntary or accidental dissemination of an organism, or its gametes, outside its controlled area of confinement. (ICES, 2005)
Secondary introduction	...is one that takes place as the result of an intentional or unintentional introduction into a new area and the species disperses from that point of entry to other areas that it could not have reached without the initial (primary) human mediated introduction. (GESAMP, 1997)
Secondary introduction	Takes place as the result of an intentional or unintentional introduction into a new area and the species disperses from that point of entry to other areas that it could not have reached without the initial (primary) human mediated introduction. (Nehring, 2005)
Secondary introduction	...is one that takes place as the result of an intentional or unintentional introduction into a new area, when the species disperses from that point of entry into areas it could not have reached without the initial (primary) human aided introduction. (Hopkins, 2001)
Spread	Expansion of the geographical distribution of an alien species within an area. (Nehring, 2005)
Transferred species	(= transplanted species) Any species intentionally or accidentally transported and released within its present range. (GESAMP, 1997)
Transferred species	Any species intentionally or accidentally transported and released within areas of established populations and continuing genetic flow where it occurs. Synonym: transplanted. (Nehring, 2005)
Transferred species (= transplanted species)	Any species intentionally or accidentally transported and released within areas of established populations, and continuing genetic flow where it occurs. (ICES, 2005)
Translocation	Movement of native or introduced species to habitats outside its historically known range. (GESAMP, 1997)

Term	Definition
Unintentional introduction	All other introductions which are not intentional; this also includes parasites, symbionts etc. of intentionally introduced species. (Nehring et al., 2009)
unintentional introduction	All other introductions which are not intentional. (Genovesi and Shine, 2003)
Unintentional introduction	A species utilising unwitting humans or human delivery systems as vectors to disperse and become established outside its natural range (IUCN, 2000). (Secretariat of the Convention on Biological Diversity, 2001)
Unintentional introduction	A species utilising unwitting humans or human delivery systems as vectors to disperse and become established outside its natural range. (Nehring, 2005)
Unintentional introduction	...means an introduction made as a result of a species utilizing humans or human delivery systems as vectors for dispersal outside its natural range. (The introduction is incidental to the main transaction taking place (often trade), but may have major environmental consequences). (Hopkins, 2001)
Un-Intentional introduction	...is one made as a result of organisms utilising humans or human transport systems as vector for dispersal into new areas. The introduction is incidental to the main transaction taking place (often trade and in the marine environment aquaculture). (GESAMP, 1997)
Vector	Specific human transport or natural carrier that transmits alien species to the recipient ecosystem. (Panov et al., 2009)
Vector	Any living or non-living carrier that transports living organisms intentionally or unintentionally. (Nehring, 2005)
Vector	A vector is a transfer mechanism and is the physical means by which species are transported from one geographic region to another, e.g. ballast water or ship's hull. (Olenin et al., 2010)
Vector	Any living or non-living carrier that transports living organisms intentionally or unintentionally. (ICES, 2005)
Weeds	Plants (not necessarily alien) that grow in sites where they are not wanted and have detectable negative economic or environmental effects; alien weeds are invasive alien species. (Nehring, 2005)
Weeds (synonyms: plant pests, harmful species, problem plants)	Plants (not necessarily alien) that grow in sites where they are not wanted and have detectable negative economic or environmental effects; alien weeds are invasive alien species. (Secretariat of the Convention on Biological Diversity, 2001)

Following the MSFD, it is proposed to use the term 'non-indigenous species' which should be defined as follows. A 'non-indigenous species' is a species, subspecies or lower taxon (including genetically distinct populations) that is able to survive and subsequently to reproduce outside its past and present natural distribution area and dispersal potential due to intentional or unintentional human action. The introduction of a 'non-indigenous species' into a new area can happen with any part of its life cycle including gametes, eggs, seeds or other propagules, that are able to produce autonomous individuals. If the 'non-indigenous species' affects adversely native populations, species communities, habitats, ecosystem structure and functioning, socio-economic values or human health it is called invasive.

Using the term ‘indigenous species’ usually the occurrence in an area is implied over a long period of several centuries, millennia or even historical. That, for example means, a species introduced two or three hundred years ago is called ‘non-indigenous species’. Often the discovery of America by Columbus in 1492 is used to discriminate between Archaeo- and Neobiota. This refers to the increasing exchange of species between America and Europe marked by that date.

For the use of the term concerning the implementation of the MFSD such a temporal demarcation should be reconsidered. For the marine area, both for coastal and open waters, it is very difficult to eliminate non-indigenous species completely by measures if they have already established themselves in the ecosystem. Corresponding activities will have mainly preventive and confining character. Many introduced species fit into the complex interactions of food webs in the new areas after some time without having negative effects to other populations or the structure of the system itself. As an example, the diatom *Odontella sinensis* may be mentioned which was observed in the North Sea in 1908 for the first time (Ostenfeld, 1908). For today’s scientists this species is a regular part of the planctonic system that has no negative effects. For that reason, it is proposed to use only ‘non-indigenous species’ for the classification of the environmental status concerning the implementation of descriptor 2 within the MSFD which have been introduced during the past 50 years. In addition, older 'non-indigenous species' will only be considered for the classification if they have had demonstrably negative impact during the last 20 years, or still have today.

2.2.2 Vectors of introduction and distribution, management for prevention

There are many vectors with that species may be introduced into non-native habitats. One main vector for introduction or distribution is the transport in the ballast water of ships. Ships may also transport the species being directly attached to their hull, in the sediments of the ballast water tanks or in sediments attached to the hull, anchors or chains, and with commercial fishing nets and gear as well (Gollasch and Leppäkoski, 1999). Another important vector is aquaculture (Leung and Dudgeon, 2008). In this case, the intentional introduced target species may escape and spread unintentionally or introductions of species may be unintentionally introduced together with the target species for culture as epi- or endobionts, parasites and diseases (Wolff and Reise, 2002). Additionally, the removal of geographical barriers as by building of canals may contribute to the spread of species to non-native habitats (Nehring, 2002). These three vectors are probably the most important ones, but there might be several more: As examples, Gollasch and Leppäkoski (1999) listed stock enhancement purposes, use of living organisms as bait or packing material for bait, ornamental trade, imports for private or public aquaria, discharging waste material of imported live specimens by fish processing companies potentially containing parasites and diseases, accidental escapes or intentional releases after experiments within research, remaining organ-isms left on or within recreational equipment (e.g. fishing rods and tackle, diving gear), import of live animals for human consumption, accidentally released into the wild before marketing, ocean and coastal currents transporting organisms attached to man-made floating objects, species introductions as fouling organisms on migrating non-indigenous host species (e.g. fish and birds) and transport of sand and gravel as construction material.

Especially in aquatic habitats the eradication or control of non-indigenous species is difficult or impossible. Thus, the main focus lies on the prevention of new introductions. For the vector ballast water many approaches have been made to develop effective treatments or suitable management methods (Matheickal et al., 2004). At the moment, the International Convention for the Control and Management of Ships’ Ballast Water and Sediments adopted by the International Maritime Organization (IMO) in 2004 requires all vessels to have a Ballast Water Management Plan and to carry out ballast water management according to a given standard. Ballast water exchange will continue to be common practice until all vessels have to meet the IMO standard. The exchange should take place 200 nautical miles from land and at sites with water depths > 200

m. For preventing introductions by aquaculture, the ICES Code of Practice on the Introductions and Transfers of Marine Organisms (2005) recommends respective procedures. Against hull fouling, the toxic TBT was successfully used as paint coat until its interdiction. An alternative product has not yet been developed (Nehring, 2001). Thus, although the development of a prevention management is in progress, there is a need for setting up further methods.

In autumn 2012, the European Commission published a proposal for a regulation on the prevention and management of the introduction and spread of invasive alien species (European Commission, 2013). The proposal seeks to address the problem of invasive alien species in a comprehensive manner so as to protect native biodiversity and ecosystem services, as well as to minimize and mitigate the human health or economic impacts that these species can have. With regard to the realisation of measures and the establishment of monitoring programmes, the proposal contains practical instructions, but some of the objectives will have to be revised and clarified.

2.2.3 Impacts within ecosystems

The new arrival of a non-indigenous species does not necessarily cause problems in the new habitat immediately. Nehring (2003 b) described three possible ecological reactions when a non-indigenous species reaches a new habitat: Firstly, it may not even establish. Secondly, it establishes and coexists with the native species without significant interaction. Thirdly, it establishes and alters relationships between species, establishes a new dynamic of competition and predation and or displaces native species directly or indirectly. The reason why some species establish and others not is still in question. However, most establishing species are probably highly competitive opportunists and therefore it can be assumed that they will affect their new habitat. The impact may be positive (Norkko et al. 2012), but often invasive species cause negative effects on communities, habitats and ecosystems (Nehring and Klingenstein, 2007). The effects can be classified into five categories: (i) hybridisation with native and/or other alien species, and then (ii) producing reproductive offspring, (iii) competition, predation and herbivory, (iv) introduction of parasites and disease agents which affect indigenous species, (v) habitat alteration resulting in changes of biological structures and water budget (Nehring and Klingenstein, 2007 modified after Jansson, 1994). Non-indigenous species have been shown to displace native species by competition for food (Kotta and Olafsson, 2003) or space (Nehls et al., 2006). They may change habitats, e.g. by acting as ecosystem engineers (Zaiko et al., 2009; Wallentinus and Nyberg, 2007), and influence ecosystem structure and functioning (Bilio and Niermann, 2004). Non-indigenous species are considered being the main cause for loss in biodiversity world-wide and may cause economic loss and be a threat for human health (Weidema, 2000). The impacts may change with time, related to environmental changes, such as climate warming. As a consequence, some effects of invasive species may decrease, but on the other hand, new impacts may arise from species that seemed to have been harmless before.

2.2.4 Non-indigenous species in the North Sea

The North Sea is a marginal sea of the Atlantic Ocean enclosed at three sites by land with two connections to the ocean. The marine inflow comes mainly through the large funnel-shaped opening at the north and to a lesser extent through the more narrow and shallow English Channel. The North Sea is a structurally uniform shallow shelf sea with larger depths only in the Norwegian channel. In the central North Sea, the salinity is about 35 which decreases in direction to the coastal estuaries. There are strong tides in the North Sea affecting also the Wadden Sea. Thermal stratification develops in spring time in the deeper areas and lasts until complete mixing in autumn occurs.

At least 167 non-indigenous species have been documented for the North Sea. Several studies summarise the findings for different regional coasts or at a larger scale which have been revised by Gollasch et al. (2009 and references therein). They also provide a detailed table of the non-indigenous species with further information about countries, status and vectors of introduction. At the German North Sea coast, 53 species are regarded as established[1] (see Table 2-2). Most of them have been found at the shores and estuaries and the proportion of non-indigenous species to native species increased from offshore towards the coast and have been found to be highest in estuaries (Nehring and Klingenstein, 2005). Wolff (1998) and Nehring (2006) have described reasons for this finding. First of all, the intensive international shipping poses a higher potential infection rate to the estuaries where economically important ports are located. Secondly, brackish water species are physiologically more tolerant with regard to transport in ballast water tanks - this water is often brackish - and therefore, they have a greater chance to be released from tanks being still alive. Thirdly, in brackish waters that exhibit an indigenous species minimum non-indigenous species have a higher potential to establish in the unsaturated ecological niches (Remane, 1934). Additionally, in estuaries the invasion pressure occurs from two sides, via the shipping from the ocean and via inland waters especially due to shipping channel constructions.

Table 2-2: Established non-indigenous species in the North Sea and the Baltic Sea (modified from www.aquatic-aliens.de, last update 18/06/2013).

Group	Taxonomic affiliation	Taxon	North Sea	Baltic Sea
Phytoplankton	Dinophyceae	*Gymnodinium mikimotoi*	x	-
Phytoplankton	Dinophyceae	*Thecadinium yashimaense*	x	-
Phytoplankton	Raphidophyceae	*Chattonella antiqua*	x	-
Phytoplankton	Raphidophyceae	*Chattonella marina*	x	-
Phytoplankton	Raphidophyceae	*Fibrocapsa japonica*	x	-
Phytoplankton	Bacillariophyceae	*Coscinodiscus wailesii*	x	x
Phytoplankton	Bacillariophyceae	*Odontella (Biddulphia) sinensis*	x	x
Phytoplankton	Bacillariophyceae	*Thalassiosira punctigera*	x	x
Macrophytes	Poaceae	*Spartina anglica*	x	-
Macrophytes	Phaeophyceae	*Colpomenia peregrina*	x	-
Macrophytes	Phaeophyceae	*Fucus evanescens*	-	x
Macrophytes	Phaeophyceae	*Sargassum muticum*	x	-
Macrophytes	Rhodophyceae	*Bonnemaisonia hamifera*	x	-
Macrophytes	Rhodophyceae	*Dasya baillouviana*	x	x
Macrophytes	Rhodophyceae	*Gracilaria vermiculophylla*	x	x
Macrophytes	Rhodophyceae	*Polysiphonia harveyi*	x	-
Macrophytes	Chlorophyceae	*Codium fragile* ssp. *tomentosoides*	x	-
Zooplankton	Ctenophora	*Mnemiopsis leidyi*	x	x
Zooplankton	Crustacea	*Acartia tonsa*	x	x
Zooplankton	Crustacea	*Ameira divagans*	-	x
Zooplankton	Crustacea	*Cercopagis pengoi*	-	x
Macrozoobenthos	Hydrozoa	*Bimeria franciscana*	x	-
Macrozoobenthos	Hydrozoa	*Cordylophora caspia*	x	x
Macrozoobenthos	Hydrozoa	*Nemopsis bachei*	x	-
Macrozoobenthos	Anthozoa	*Diadumene cincta*	x	-
Macrozoobenthos	Bivalvia	*Congeria leucophaeta*	x	x
Macrozoobenthos	Bivalvia	*Corbicula fluminalis*	x	-
Macrozoobenthos	Bivalvia	*Crassostrea gigas*	x	-
Macrozoobenthos	Bivalvia	*Dreissena polymorpha*	x	x
Macrozoobenthos	Bivalvia	*Ensis americanus*	x	x

[1] http://www.aquatic-aliens.de

Group	Taxonomic affiliation	Taxon	North Sea	Baltic Sea
Macrozoobenthos	Bivalvia	*Petricola pholadiformis*	x	x
Macrozoobenthos	Bivalvia	*Teredo navalis*	x	x
Macrozoobenthos	Gastropoda	*Crepidula fornicata*	x	-
Macrozoobenthos	Gastropoda	*Potamopyrgus antipodarum*	x	x
Macrozoobenthos	Polychaeta	*Ficopomatus enigmaticus*	x	-
Macrozoobenthos	Polychaeta	*Marenzelleria neglecta*	x	x
Macrozoobenthos	Polychaeta	*Marenzelleria viridis*	x	x
Macrozoobenthos	Crustacea	*Balanus improvisus*	x	x
Macrozoobenthos	Crustacea	*Caprella mutica*	x	-
Macrozoobenthos	Crustacea	*Corophium curvispinum*	x	x
Macrozoobenthos	Crustacea	*Corophium sextonae*	x	-
Macrozoobenthos	Crustacea	*Elminius modestus*	x	-
Macrozoobenthos	Crustacea	*Eriocheir sinensis*	x	x
Macrozoobenthos	Crustacea	*Gammarus tigrinus*	x	x
Macrozoobenthos	Crustacea	*Hemigrapsus sanguineus*	x	-
Macrozoobenthos	Crustacea	*Hemigrapsus* cf. *takanoi*	x	-
Macrozoobenthos	Crustacea	*Melita nitida*	-	x
Macrozoobenthos	Crustacea	*Palaemon macrodactylus*	x	-
Macrozoobenthos	Crustacea	*Pontogammarus robustoides*	-	x
Macrozoobenthos	Crustacea	*Proasellus coxalis*	x	-
Macrozoobenthos	Crustacea	*Rhithropanopeus harrisii*	x	x
Macrozoobenthos	Crustacea	*Sinelobus stanfordi*	x	-
Macrozoobenthos	Bryozoa	*Tricellaria inopinata*	x	-
Macrozoobenthos	Bryozoa	*Victorella pavida*	x	-
Macrozoobenthos	Ascidiacea	*Styela clava*	x	-
Macrozoobenthos	Insecta	*Telmatogeton japonicas*	x	x
Fishes		*Neogobius melanostomus*	-	x
Parasites	Fungi	*Claviceps purpurea*	x	-
Parasites	Nematoda	*Anguillicola crassus*	x	x
Total Number			**53**	**29**

One species that probably has benefited from the new waterways is the Chinese mitten crab *Eriocheir sinensis*. Its life-cycle is characterized by migrations to waters of different salinities (15-32) (Anger et al., 1991). Thus, individuals of this species occur in rivers, estuaries and marine habitats (Rudnick et al., 2000). *E. sinensis* originates from China, Japan and Taiwan (Peters, 1933; Panning, 1938). In Europe it was first recorded in the German river Aller in 1912. The species then spread rapidly throughout northern Europe. It was probably introduced to the Baltic Sea via the North Sea and Baltic canal and reached the German Baltic coast in 1926. The crab is abundant in German estuaries adjacent to the North Sea, i.e Ems, Elbe and Weser rivers. The most likely introduction vector is shipping (ballast water and hull fouling of vessels) or import of living species for aquaria and for human consumption (Marquard, 1926; Peters, 1933). Due to the mass occurrence of the crab the economic damage in German waters is estimated to approximately 80 million Euros since 1912 (Gollasch, 2006). Impacts derive mainly from the burrowing, migratory and feeding behaviours of the crabs (Dittel and Epifano, 2009) resulting in increased erosion of dikes, river and lake embankments. Furthermore, they can also clog up industrial water intake filters during mass occurrences. *E. sinensis* also damages nets by feeding on fish caught in traps and nets and is a competitor for place and food (Gollasch, 2006). In some European countries, crabs are imported for human consumption.

The largest fraction of non-indigenous species in the North Sea is found among macrozoobenthos species followed by phytoplankton and phytobenthos (Nehring, 2003c). The most invertebrate non-indigenous species have their origin at the Atlantic coast of America and have been introduced most probably by shipping, while many algal species originate from Pacific regions and unintentionally have arrived with imported seed oysters (Nehring and Klingenstein, 2005). One of the first non–indigenous phytoplankton species being identified in the North Sea is the Pacific algae *Odontella (Biddulphia) sinensis*. A mass occurrence was reported in 1908 (Ostenfeld, 1908). Already at that time, ballast water has been suggested to be the introduction vector. The introduction vectors are reflected as well by the types of life forms within the non-indigenous species. Most of them are epibenthic or vagile epifauna (Nehring, 2003 c). The probability of being transported in ballast water or attached to ship hulls or mariculture products is higher for mobile species than for species that live mainly endobenthic. Faubel and Gollasch (1996), Gollasch (1999) and Gollasch and Riemann-Zürneck (1996) have described specimens of a plathelminth, a decapod and a sea anemone, respectively, that have arrived alive in German ports on ships hulls. These species are just some examples found by a German study of the Environmental Protection Agency in which samples of ballast water, tank sediment and hulls were taken from 186 ships (Gollasch, 1996; Lenz et al. 2000). In about 75% of the ballast water and sediment samples and in 99% of the hull samples, organisms have been detected. 58% of them were determined as non-indigenous. The global shipping traffic is immense and further increasing. The complex global network is shown in a study by Kaludza et al. (2007). They have found regional clusters and different patterns for different ship types which may have implications for the spread of species. It seems that the Netherlands have more records of non-indigenous species than other parts of the North Sea. This may be due to the intensive shipping and aquaculture in this region (Gollasch et al., 2009). Nehring et al. (2009) have identified several hotspots of non-indigenous species in the Wadden Sea which may be important to survey for early detection. The rate of introduction is still increasing. Most of the established species have broad tolerances for ecological factors. Many have good dispersal mechanisms as mass development of swimming larvae, thus giving them the possibility to spread rapidly. However, some species show higher abundances only locally. In some cases, non-indigenous species also seem to have facilitated the development of other non-indigenous species (Nehring et al., 2009).

With the climate warming, species may expand their ranges and more warm water species might establish in the North Sea (Walther et al., 2009; Nehring, 1998). Also, indirect effects of climate change may facilitate the establishment of new species in the habitat (Reid et al., 2009 and references therein). First signs have been already noted. On the one hand, species increase their range moving further northwards with natural drifts from the Atlantic and establishing in the German bight (Gollasch et al., 2009). On the other hand, more and more thermophile species introduced by human activity can establish there (Nehring, 2003c). One species that may be favoured by the increasing temperatures is the toxic algae *Pfiesteria psicicida*. In the North Sea, this species is living at the edge of its temperature tolerance range (Gollasch, 2003). If the toxic variant of these algae were able to establish in the North Sea this could lead to fish mortalities, as for example at the Atlantic coast of the USA.

However, the number of non-indigenous species may be underestimated. Reise et al. (1999) have discussed several reasons for that. The identification often depends on the interest in an organism group and the knowledge about it. Especially less conspicuous and less studied groups may be underrepresented (Carlton, 2003) such as the small planktonic fraction. *Karenia mikimotoi* is one example for a small planktonic species which has proved being problematic with regard to its proper identification. This and the features and effects of *K. mikimotoi* are described in more detail in chapter 2.3.2.3.1. Often the identification is difficult or even impossible (Nehring, 2003 c). Consequently, the introduction of a species may have occurred before it was recognised taxonomically. In addition, also indigenous species may have been overlooked (Nehring, 1998). Thus, if it is not possible to decide whether a species is indigenous or not, then it will be called cryptogenic. The question remains after what time period an introduced species may not be addressed as 'exotic' any longer. One good example is the naval shipworm *Teredo navalis*, which is not a worm but a mussel from the family of Teredinidae. The historic native distribution of *T. navalis* is uncertain[2]. This is due to the fact that wooden ships have unwittingly transported this mussel all over the world for many centuries. The first record of *T. navalis* in northern European waters was in 1731 in a Dutch dike (Kramp, 1945). It appeared between the 1930s and 1950s in the Baltic Sea and since 1931 it has been abundant along the German Mecklenburg-Pomeranian coast (Bönsch and Gosselck, 1994; Sordyl et al., 1998). The main vectors of introduction have been floating wooden objects, such as wooden ship hulls. Moreover, currents and ballast water act as a carrier, in especially for the larvae of *T. navalis*[3]. There is no evidence of how *T. navalis* reached the Baltic Sea. One possibility is ballast water; the other is by currents coming from the North Sea (Didžiulis, 2011). However, the mussel has an enormous economic impact although there is no known negative ecological effect on habitats and indigenous organisms. There are several examples for the destruction of wooden constructions by *T. navalis*, for example wooden ships, dikes, pears and wharfs (Hubschman, 1979; Thompson et al., 2005). In the Baltic Sea, the economic damage due to the activity of *T. navalis* since 1933 is estimated to be around 50 million Euros (Wichman, 2005).

Another early invader has been *Balanus improvisus*. It has probably been introduced by hull fouling of ships from the Atlantic coast of America. The barnacle has probably been abundant in European waters for more than 200 years. The first record from the German North Sea coast stems from 1858 (Gollasch and Nehring, 2006), and the species was first recorded in the Baltic Sea in 1944 at Königsberg (Leppäkoski and Olenin, 2000). There is some discussion with regard to the earliest records. *Balanus improvisus* can dominate the community by competing for space and food, but it does not have a negative effect on community diversity in the Baltic (Dürr and Wahl, 2004[4]). This species fouls communities such as mussels, but also ships hulls and other underwater constructions. The barnacle changes the habitat through the construction of dense crusts on hard surfaces and secondary hard substrates, inhibiting water flow, attracting associated fauna and producing organic debris (Leppäkoski, 1999).

Although many species so far have a more additive character with regard to the diversity of the North Sea (Reise et al., 1999), there are further examples for possible ecological or economic effects, either positive or negative. Additionally, the impacts may change over time and are not easy to predict. For example, *Coscinodiscus wailesii* was first recognised in 1977 (Boalch and Harbour, 1977). The species has probably been introduced via oyster cultures (Rincé and Plaumier, 1986). During the following years it had spread over the whole North Sea and become successfully established, sometimes making up to 90 % of the phytoplankton biomass (Dürselen and Rick, 1999). During dense blooms, it had produced high amounts of mucilage that had clogged fishing nets or aquaculture cages and covered the seabed. This situation seems to have changed: During the past decade, *C. wailesii* has been found regularly in phytoplankton samples, but no dense blooms have been reported any more.

Reise et al. (2005) identified six species which already have permanent effects on biota or altered the habitat. The pacific oyster *Crassostrea gigas* is one example and described in detail in chapter 2.3.2.3.1, a second example is the polychaete *Marenzelleria* cf. *wireni* discussed in chapter 2.3.2.3.2. Another species is the cord-grass *Spartina anglica*. This species is a fertile hybrid originating from the native Britain *Spartina maritima*

[2] http://www.europe-aliens.org/pdf/teredo_navalis.pdf

[3] http://www.norsas.eu/species/teredo-navalis

[4] http://www.europe-aliens.org

and the North-American *Spartina alterniflora*, introduced before 1870 to Great Britain (Eno et al., 1997). The first record was made in 1892 in Britain (Gray et al. 1991). Its ability to accrete sediments results in an enormous export of *S. anglica* to Europe, China and western USA for coastal protection purposes. The first export was in 1924 to the Netherlands in the Westerschelde estuary (Wolff, 2005). The first specimens of *S. anglica* in Germany were introduced to the northern Wadden Sea near Husum in 1927 (Kolumbe, 1931). Today, it occurs along the entire German Wadden Sea coast (Nehring and Hesse, 2006). It can be assumed that most introductions around the world were intentional. Not all effects have to be negative, for *S. anglica* both positive and negative effects have been described. *S. anglica* has a negative impact on indigenous organisms by replacement of native plants and animals, such as *Salicornia stricta*, *Zostera noltii*, *Arenicola marina*, *Nereis diversicolor* and *Corophium volutator* (König, 1948; Reise, 1994; Loebl, 2002; Reise et al., 2005; Gribsholt and Kristensen, 2003; Nehring and Adsersen, 2006). As a consequence, this will affect habitats for endobenthic invertebrates, migrating shorebirds and waterfowl as well as rearing habitat for fish. It also has an effect on the biogeochemical turnover in the sediment (Gribsholt and Kristensen, 2002). In its function for coastal protection and stabilization it influences water circulation. This could lead to decreased flow and increased flooding. Economically *S. anglica* affects oyster fisheries and activities like fishing, boating, bird watching etc. (Nehring and Adsersen, 2006). Positive effects are the mentioned protection and stabilisation of the coast. Furthermore, it serves as a food source for cattle and goat (Ranwell, 1967) and is used as manure (NWCB, 2005).

Another macrophyte affecting the habitat is the Japanese seaweed *Sargassum muticum*. It originates in the Northwest Pacific and is found in coastal waters of Japan, Russia, Korea and China, where it lives in shallow waters in the lower intertidal and upper subtidal zones. It was accidentally introduced to European waters in the early 1970s and was first recorded in the English Channel in 1973 (Farnham et al., 1973). Most introduced populations probably originate from imports of oysters (*Crassostrea gigas*) and mussels for aquacultures to France (Farnham, 1994). This seaweed rapidly spread north- and southward along the European coasts, from Portugal to Norway, as well as into the North Sea and Baltic Sea (Wallentinus, 1999; Staehr et al., 2000). *Sargassum muticum* was first observed in Germany in 1988 and reached the island of Sylt in 1993 (Kornmann and Sahling, 1994). *S. muticum* has under favourable conditions a rapid growth rate (2–4 cm/day), high fecundity and a long-life span. It has the ability to reproduce from drifting fertile individuals or branches, which are able to drift long distances due to the buoyancy provided by the air vesicles, and rapidly colonize new areas. *S. muticum* is a strong competitor with native macrophytes for space and light through its fast growth, high fertility, high biomass, and high densities which may prevent settlement and development of other algae (Critchley et al., 1986; Staehr et al., 2000). But some investigations have indicated that the introduction of the Japanese seaweed has not caused major changes in the community structure of the epibiota and that the abundance of epibiota has increased instead (Buschbaum et al., 2006; Wernberg et al., 2004). Dense stands of *S. muticum* may provide protection and serve as living and feeding ground for invertebrates and fish (Polte and Buschmann, 2008; Wallentinus, 1999).

A further non-indigenous species that has a significant impact in the North Sea is the slipper limpet *Crepidula fornicata*. This species is often found in a chain with the oldest female individual at the base and several male individuals attached. The males transform to females over time and further males attach. In this way, chains of up to 12 individuals can be built up. Males that do not find a female after their planktonic stage may self-fertilize (Cole, 1952). This reproduction strategy and the ability to colonize a wide range of habitats is a key to the invasion success of this gastropod. *C. fornicata* probably has been introduced with clam imports from North America to Europe. The first known occurrence was in 1872 in Liverpool Bay, but at that time the species could not establish (Minchin et al., 1995). Further introductions with the American oyster *Crassostrea virginica* to Essex about 15 years later seemed to have been more successful and since then the species has spread rapidly. It has considered a 'pest' on oyster banks and the first individuals in Germany were found in 1934 on an oyster bank at Sylt (Ankel, 1935). There is evidence that colder winters reduce the populations (Thieltges et al., 2004), but with the global warming the slipper limpet may increase its abundances and spread further north. Dispersal occurs via the planktonic larval stage with natural drifts or in ballast water but also with hull fouling (Blanchard, 1997). The occurrences of *C. fornicata* have been often associated with oysters (Thieltges at al., 2003). The slipper limpet might compete for food and space with several filter feeding invertebrates as mussels (Thieltges, 2005a) but also provide shelter against star fish predation (Thieltges 2005b). High abundances can change the sediment structure by accumulation of faeces and pseudofaeces (Barnes et al., 1973).

Another invertebrate has become important in the North Sea: The American razor clam *Ensis americanus* was probably introduced as larvae with ballast water from the Atlantic coast of North America in 1979 (Cosel et al., 1982). It spread very rapidly to the north and the west (Nehring and Leuchs, 2000 and references therein) via natural larval dispersal. *E. americanus* may be the most abundant large bivalve in the shallow subtidal (Nehring and Leuchs, 2000). In spite of these observed high abundances, no direct impacts on species and communities have been found so far (Jensen and Kathe, 2010). On the contrary, *E. americanus* could be an example for a beneficial role in ecosystem functioning by e.g. complementary resource use compared to native species. The suspension feeder could strengthen the coastal biofilter and increase benthic biomass production (Reise et al., 2006; Armonies and Reise, 1998).

Although the North Sea is a region with high invasion rates of non-indigenous species, so far no severe impacts on species, habitats and ecosystems have been detected. However, several species have influenced the systems by habitat alteration, competition or facilitation. In some cases, economic damage had occurred. Due to changing conditions as by climate warming or further introductions of species that could lead to shifts in the food webs, the future is not easy to predict and may bring more dangerous impacts.

2.2.5 Non-indigenous species in the Baltic Sea

The Baltic Sea is an intra-continental sea and one of the world’s largest brackish water bodies. From the geological origin, it is a young sea that had formed after the last glaciation period. The shallow basin has an irregular shape with several deeper basins and gulfs as the Gulf of Bothnia, the Gulf of Finland and the Gulf of Riga. The semi-enclosed Baltic Sea is connected with the marine North Sea by various basins, shallow areas and sills; the Belt Sea, the Sound, and the Kattegat Strait. Due to the shallow areas of the sills and sounds, the water exchange with the North Sea is limited. In addition to temporary salt-water inputs, many rivers bring freshwater into the Baltic Sea. This leads to a strong salinity gradient from about 30 near the North Sea to 20–24 in the Kattegat, 6–8 in the Baltic Proper and 2–3 in the inner parts of the large gulfs. The Baltic Sea has only a minimal tidal action and a strong and permanent halocline/pycnocline. In summer, a distinct thermocline can develop and separate cold intermediate winter waters from the warm surface layer. Below the thermo- and pycnocline, anoxic conditions occur regularly. The temperature of the surface layer also forms a gradient from north to south with colder climatic conditions and ice cover over 170 to 190 days in the north and irregular ice cover and maximum water temperatures of 14–16/19°C in summer in the south. These horizontal and vertical gradients in addition to the complicated bottom relief and coastal topography provide many habitats for organisms in a wide salinity and temperature range.

In general, the indigenous species richness of the Baltic Sea is low. Most species are postglacial immigrants and mainly euryhaline (Segerstråle, 1957). The low species richness offers a good habitat for invasion with less competition and more empty niches (Paavola et al., 2005). The same conditions as discussed for the North Sea estuaries (chapter 2.2.4) may also apply to the brackish Baltic Sea with its broad salinity range. Paavola et al. (2005) have indeed found that most of the established non-indigenous species are within the salinity range of their native habitat but might tolerate a much broader range. Altogether at least 116 non-indigenous species have been introduced to the Baltic Sea (Olenin et al., 2010; Gollasch and Leppäkoski, 2007), so far 88 species are considered to be established (Baltic Sea Alien Species Database, 2010[5], see Table 2-3). For the German part of the Baltic Sea, 34 species have been counted as non-native and 29 as established (Gollasch and Nehring, 2006; Nehring, 2010[6], see Table 2-2). The origin of the non-indigenous species is reflecting the probable transport vectors. Many species are transoceanic and seem to have been transported mainly via ballast water and hull fouling, but also via sediments of ballast water tanks and aquaculture. Another part originates from the Ponto-Caspian region to which the Baltic is connected by a river and channel system. The removal of natural barriers by building the channels seemed to have facilitated their active or passive dispersal which can be also supported by shipping. Due to their history, the Ponto-Caspian fauna is highly euryhalin and has broad environmental tolerances (Reid and Orlova, 2002). In combination with the comparable environmental conditions of Baltic Sea and Ponto-Caspian Sea, this may explain their success in the Baltic Sea.

Table 2-3: Non-indigenous species (independent from status) in the Baltic Sea (modified from 'AquaNIS', http://www.corpi.ku.lt/databases/index.php/aquanis, last update 11/12/2013).

Group	Taxonomic affiliation	Taxon
Phytoplankton	Bacillariophyceae	*Coscinodiscus wailesii*
Phytoplankton	Bacillariophyceae	*Odontella sinensis*
Phytoplankton	Bacillariophyceae	*Thalassiosira punctigera*
Phytoplankton	Dinophyceae	*Karenia mikimotoi*
Phytoplankton	Dinophyceae	*Prorocentrum minimum*
Macrophytes	Characea	*Chara connivens*
Macrophytes	Chlorophycea	*Codium fragile fragile*
Macrophytes	Chlorophycea	*Protomonostroma undulatum*
Macrophytes	Hydrocharitacea	*Elodea canadensis*
Macrophytes	Phaeophyceae	*Colpomenia peregrina*
Macrophytes	Phaeophyceae	*Fucus evanescens*
Macrophytes	Phaeophyceae	*Sargassum muticum*
Macrophytes	Poacea	*Spartina townsendii var. anglica*
Macrophytes	Poacea	*Spartina x townsendii*
Macrophytes	Rhodophyceae	*Bonnemaisonia hamifera*
Macrophytes	Rhodophyceae	*Dasya baillouviana*
Macrophytes	Rhodophyceae	*Gracilaria vermiculophylla*
Macrophytes	Rhodophyceae	*Neosiphonia harveyi*
Zooplankton	Crustacea	*Acartia (Acanthacartia) tonsa*
Zooplankton	Crustacea	*Ameira divagans divagans*
Zooplankton	Crustacea	*Cercopagis (Cercopagis) pengoi*
Zooplankton	Crustacea	*Cornigerius maeoticus*
Zooplankton	Crustacea	*Evadne anonyx*
Zooplankton	Crustacea	*Platorchestia platensis*
Zooplankton	Ctenophora	*Mnemiopsis leidyi*
Zooplankton	Hydrozoa	*Maeotias marginata*
Macrozoobenthos	Ascidiacea	*Styela clava*
Macrozoobenthos	Bivalvia	*Crassostrea gigas*

[5] http://www.corpi.ku.lt/nemo

[6] http://www.aquatic-aliens.de

Group	Taxonomic affiliation	Taxon
Macrozoobenthos	Bivalvia	*Crassostrea virginica*
Macrozoobenthos	Bivalvia	*Dreissena bugensis*
Macrozoobenthos	Bivalvia	*Dreissena polymorpha*
Macrozoobenthos	Bivalvia	*Ensis directus*
Macrozoobenthos	Bivalvia	*Mya arenaria*
Macrozoobenthos	Bivalvia	*Mytilopsis leucophaeata*
Macrozoobenthos	Bivalvia	*Petricolaria pholadiformis*
Macrozoobenthos	Bivalvia	*Rangia cuneata*
Macrozoobenthos	Bivalvia	*Teredo navalis*
Macrozoobenthos	Bryozoa	*Victorella pavida*
Macrozoobenthos	Crustacea	*Amphibalanus improvisus*
Macrozoobenthos	Crustacea	*Callinectes sapidus*
Macrozoobenthos	Crustacea	*Chaetogammarus ischnus*
Macrozoobenthos	Crustacea	*Chaetogammarus warpachowskyi*
Macrozoobenthos	Crustacea	*Chelicorophium curvispinum*
Macrozoobenthos	Crustacea	*Dikerogammarus haemobaphes*
Macrozoobenthos	Crustacea	*Dikerogammarus villosus*
Macrozoobenthos	Crustacea	*Eriocheir sinensis*
Macrozoobenthos	Crustacea	*Gammarus tigrinus*
Macrozoobenthos	Crustacea	*Gmelinoides fasciatus*
Macrozoobenthos	Crustacea	*Hemimysis anomala*
Macrozoobenthos	Crustacea	*Homarus americanus*
Macrozoobenthos	Crustacea	*Limnomysis benedeni*
Macrozoobenthos	Crustacea	*Limulus polyphemus*
Macrozoobenthos	Crustacea	*Obesogammarus crassus*
Macrozoobenthos	Crustacea	*Orchestia cavimana*
Macrozoobenthos	Crustacea	*Orconectes limosus*
Macrozoobenthos	Crustacea	*Pacifastacus leniusculus*
Macrozoobenthos	Crustacea	*Palaemon elegans*
Macrozoobenthos	Crustacea	*Paramysis (Mesomysis) intermedia*
Macrozoobenthos	Crustacea	*Paramysis (Serrapalpisis) lacustris*
Macrozoobenthos	Crustacea	*Pontogammarus robustoides*
Macrozoobenthos	Crustacea	*Pseudocuma (Stenocuma) graciloides*
Macrozoobenthos	Crustacea	*Rhithropanopeus harrisii*
Macrozoobenthos	Gastropoda	*Crepidula fornicata*
Macrozoobenthos	Gastropoda	*Lithoglyphus naticoides*
Macrozoobenthos	Gastropoda	*Potamopyrgus antipodarum*
Macrozoobenthos	Hydrozoa	*Bougainvillia rugosa*
Macrozoobenthos	Hydrozoa	*Cordylophora caspia*
Macrozoobenthos	Hydrozoa	*Gonionemus vertens*
Macrozoobenthos	Hydrozoa	*Pachycordyle navis*
Macrozoobenthos	Insecta	*Telmatogeton japonicus*
Macrozoobenthos	Oligochaeta	*Branchiura sowerbyi*
Macrozoobenthos	Oligochaeta	*Paranais frici*
Macrozoobenthos	Oligochaeta	*Potamothrix bedoti*
Macrozoobenthos	Oligochaeta	*Potamothrix heuscheri*
Macrozoobenthos	Oligochaeta	*Potamothrix vejdovskyi*
Macrozoobenthos	Oligochaeta	*Tubificoides pseudogaster*
Macrozoobenthos	Polychaeta	*Alitta succinea*
Macrozoobenthos	Polychaeta	*Alkmaria romijni*
Macrozoobenthos	Polychaeta	*Boccardiella ligerica*
Macrozoobenthos	Polychaeta	*Ficopomatus enigmaticus*
Macrozoobenthos	Polychaeta	*Marenzelleria arctia*

Group	Taxonomic affiliation	Taxon
Macrozoobenthos	Polychaeta	*Marenzelleria neglecta*
Macrozoobenthos	Polychaeta	*Marenzelleria viridis*
Fishes		*Acipenser baeri*
Fishes		*Acipenser gueldenstaedtii*
Fishes		*Acipenser oxyrinchus*
Fishes		*Acipenser ruthenus*
Fishes		*Acipenser stellatus*
Fishes		*Carassius gibelio*
Fishes		*Catostomus catostomus*
Fishes		*Coregonus autumnalis*
Fishes		*Coregonus muksun*
Fishes		*Coregonus nasus*
Fishes		*Coregonus peled*
Fishes		*Ctenopharyngodon idella*
Fishes		*Cyprinus carpio*
Fishes		*Huso huso*
Fishes		*Hypophthalmichthys molitrix*
Fishes		*Hypophthalmichthys nobilis*
Fishes		*Lepomis gibbosus*
Fishes		*Micropterus dolomieu*
Fishes		*Micropterus salmoides*
Fishes		*Neogobius melanostomus*
Fishes		*Oncorhynchus gorbuscha*
Fishes		*Oncorhynchus keta*
Fishes		*Oncorhynchus kisutch*
Fishes		*Oncorhynchus mykiss*
Fishes		*Oncorhynchus nerka*
Fishes		*Oncorhynchus tshawytscha*
Fishes		*Perccottus glenii*
Fishes		*Salvelinus fontinalis*
Fishes		*Salvelinus namaycush*
Birds		*Branta canadensis*
Parasites	Monogenoidea	*Pseudodactylogyrus anguillae*
Parasites	Monogenoidea	*Pseudodactylogyrus bini*
Parasites	Nematoda	*Anguillicoloides crassus*
Total Number		**116**

One of the first species in the Baltic Sea of Ponto-Caspian origin has been the zebra mussel *Dreissena polymorpha* (Gollasch and Leppäkoski, 1999). The most likely introduction vector is shipping (ballast water and hull fouling of vessels), the transfer of animals (including crayfish) for stocking in farms (e.g. Thienemann, 1950) and the introduction into lakes of mussels attached to ship hulls (e.g. Jungbluth, 1996). This species requires suitable substrate, like hard substrates for attachment and colonises lakes, rivers and brackish lagoons. *D. polymorpha* tolerates salinities up to 7[7]. The high reproduction rate and ability to extend their planktonic stage enable *D. polymorpha* to disperse rapidly (Gollasch and Leppäkoski, 1999). Mass occurrences of *D. polymorpha* cause ecologic and economic impacts. The species competes for space and food with the native epifauna, but it serves also as an important food component for birds and fish. The zebra

[7] http://www.europe-aliens.org

mussel has a high filtration capacity and consumes high amounts of phytoplankton, which slows down the eutrophication processes, increases water transparency and improves the light conditions for benthic macro-vegetation. There might be a risk of hybridisation with other species of the genus *Dreissena* (Mills et al., 1996). Negative economic impacts caused by *D. polymorpha* are fouling of intake pipes, ship hulls, navigational constructions, cages of aquaculture and reduced angling catches (Gollasch and Leppäkoski, 1999). *D. polymorpha* was presumably introduced to Germany in the course of the extension of the inland waterway network between eastern and central Europe at the beginning of the 1800s (Gollasch and Nehring, 2006; Nehring and Leuchs, 1999; Orlova, 2002; Reinhold and Tittizer, 1997). The mussel then reached the Netherlands by 1826, being found in the Rhine at Rotterdam, most probably carried with timber imports from the Baltic Sea (Minchin et al., 2002).

This shows that the Baltic Sea is not only recipient but also donor area and plays an important role in global shipping traffic. Gollasch and Leppäkoski (2007) have conducted a risk assessment for the import of non-indigenous species via ballast water and investigated the shipping patterns in the Baltic Sea. They have identified the highly frequented shipping routes connecting the Baltic with the North Sea via the Kiel Canal, then via the British Channel and around the Iberian Peninsula into the Mediterranean Sea to the Suez Canal. As the busiest port, St. Petersburg has been identified with more than 15,500 ships visits followed by Gothenburg, Riga and Copenhagen. In total, the authors have estimated the number of ships operations at 150,000 per year and this number is expected to increase further. The amount of ballast water that is discharged in the Baltic Sea is not trivial to calculate, but may be in total about 118 million tonnes per year (Leppäkoski and Gollasch, 2006). The risk assessment for a selection of ports in the Baltic Sea has revealed that all selected ports have at least one donor port in the highest risk category (Gollasch and Leppäkoski, 2007). The risk has been determined by salinity, temperature, voyage duration and the location of the donor part. The environmental fit of donor and recipient region, as well as the time window of the year, is an important criterion for the risk assessment. However, matching temperatures may not be necessarily due to the ability of species to tolerate or adapt to new temperatures (Gollasch and Leppäkoski, 2007). Many of the invaders come from warmer regions and global warming might increase the risk of invasions, their pattern and the population dynamics. Thus, *Mnemiopsis leidyi* might expand its distribution as described in chapter 2.3.2.3.2.

Another example is the warm water species *Cercopagis pengoi*, which is of Ponto-Caspian origin and remarkably has increased the distribution area in the Baltic Sea during warm years (Leppäkoski et al., 2002). *C. pengoi* is a predaceous cladoceran, which has been first found in the Gulf of Riga and the open Gulf of Finland in 1992 (Ojaveer and Lumberg, 1995; Ojaveer et al., 2000). Since then it has spread rapidly and is now widely distributed in the Baltic Sea (Leppäkoski and Olenin, 2000). The most likely transport vector is ballast water but also attached to hulls or fishing gear the cladoceran may have been introduced unintentionally (Leppäkoski and Olenin, 2000). *C pengoi* is a brackish water species, which usually appears when temperatures reach 13.5-17°C (Birnbaum, 2006). Thus, it becomes abundant in late summer and may form mass occurrences and then clog fishing nets and fishing gear. At a fish farm at the lower Newa Estuary (Primorsk), economic losses of about 50,000 USD have been calculated between 1996 and 1998 (Panov et al., 1999) due to reduced fish catches and biofouling. Decreases of the preferred prey *Bosmina coregoni maritima* have been observed (Ojaveer et al., 2000). *C. pengoi* competes with young fish for food, but is also an important component in the nutrition of fish as the Baltic herring (Gorokhova et al. 2004; Antsulevich and Välipakka, 2000; Ojaveer and Lumberg, 1995; Ojaveer et al., 1998).

A further example may be the freshwater hydroid *Cordylophora caspia*. This species is a brackish colonial hydroid, which originates from the Black and Caspian Sea (Bij de Vaate et al., 2002). The colonies grow on hard surfaces like rocks, pilings and mussel shells. The optimum growth conditions are around salinities of 16 and temperatures of 20°C, but the species can live in salinity ranges from 0 to 30 and therefore occur in fresh and marine waters (Folino-Rorem and Indelicato, 2005). *C. caspia* can survive unsuitable living conditions and cold temperatures by creating so-called menonts (Roos, 1979). The worldwide spread of *C. caspia* has been caused by fouling on ship hulls or macrophytes (Roos, 1979) and also by ballast water, a path primarily used by the larvae. An exception is Lake Erie, USA, where the species has been introduced from aquarium releases. Since 1924, it has been distributed all over the world in temperate and tropical coastal regions (Roch 1924; Arndt 1989), which include coastal waters and estuaries of northern Europe. In the Baltic Sea, the invasion of this Ponto-Caspian species took place in the early 1800s. It occurs presently in all German rivers and channels connected to the Baltic Sea (Tittizer, 1996[8]). *C. caspia* is a competitor for food and space, especially with regard to mussels (*Mytilus edulis*, *Dreissena polymorpha*) which attach to hard substrata and whose larvae are eaten by *C. caspia*[9, 10]. By building dense colonies, it is creating new habitats for other organisms, and is furthermore responsible for a restructuring of benthic and pelagic communities (Olenin and Leppäkoski, 1999; Fuller et al., 2013). There is also a large economic impact due to biofouling, not only on ship hulls, but in particular for water intakes (Folino-Rorem and Indelicato, 2005).

The largest fracture of the non-indigenous species occurs in the coastal inlets, lagoons and gulfs of the Baltic Sea (Paavola et al., 2005). Many studies have been conducted in estuaries, which seem to be more susceptible for invasions and provide diverse environments concerning bottom conditions, temperature, salinity and anthropogenic influence. Olenin and Leppäkoski (1999) have inspected several coastal lagoons in the Southern and Northern Baltic Sea especially for effects of the non-indigenous species. They have found a variety of species that have modified the habitat, such as *Balanus improvisus* or *Dreissena polymorpha*. Zaiko et al. (2006) have identified habitats modified by shell deposits of *D. polymorpha* to be one of the most invaded and furthermore, have stated a facilitative effect of one invader for further invaders. Another important habitat modifier is the polychaete *Marenzelleria,* which is changing the habitat by its burrowing activities (described further in chapter 2.3.2.3.2). Non-indigenous species also could fill a gap of the functional groups within the habitats. One example is the New Zealand mudsnail *Potamopyrgus antipodarum* that took the role of surface deposit feeding on extremely soft bottoms in the habitats that lack *Hydrobia* spp.. The small (usually 5-6 mm) gastropod is extremely tolerant against salinities up to 16, but prefers the lowest salinities and is also able to live in freshwater. It has colonised rivers, estuarine habitats, lakes and water courses and lives in soft bottoms, but may occur on rocky bottoms, too (Jagnow and Gosselck, 1987). In Germany, *P. antipodarum* was first observed in the western Baltic Sea in 1887 (Lassen, 1978). Today it is found in freshwater habitats of most European countries. *P. antipodarum* has probably been introduced to Europe by ship ballast water (Alonso and Castro-Diez, 2008) or been transported in drinking water barrels on board of ships (Ponder, 1988). Also, transportation by birds, introduced by importing aquaculture organisms and fish stocking could have been possible vectors (Fretter and Graham, 1994; Loo et al., 2007). The New Zealand mudsnail can establish extremely dense populations of tens to hundreds of thousands of individuals per square meter in certain environments (Heywood and Edwards, 1962). It alters primary production, has a negative impact on natural habitats and competes with or displaces native snails and macroinvertebrates (Alonso and Castro-Diaz, 2008; Kerans et al., 2005).

In addition to unintentional introductions, some species have been introduced intentionally. For example, several fish species have been released into the Gulf of Riga from the 1940s until the 1960s (Leppäkoski et

[8] http://www.europe-aliens.org/pdf/cordylophora_caspia.pdf

[9] http://www.framman dearter.se/0/2english/pdf/cordylophora_caspia.pdf

[10] http://www.europe-aliens.org/pdf/cordylophora_caspia.pdf

al., 2002). Furthermore, some crustacean species that now spread in coastal lagoons and inshore waters were being introduced during the 1960s by Lithuanian hydrobiologists (Olenin and Leppäkoski, 1999). Another example for an intentional introduction is the crustacean *Gammarus tigrinus*. It was introduced to Germany in 1957 in order to replace disappeared gammarids in the river Weser and Werra (Schmitz, 1960). In the Baltic Sea, it appeared in 1975 in the Schlei Fjord using the Kiel Canal (Bulnheim, 1976). Today it is abundant in the inner bays, fjords and estuaries all along the German Baltic coastline (Zettler, 1995). From North America *G. tigrinus* arrived in Europe in 1931 in British waters by ballast water (Sexton, 1939). There is some information that it has already been introduced to Europe during World War I, but there is no distinct evidence for this. Since *G. tigrinus* has arrived in the Baltic Sea, it outnumbered and eliminated many native gammarids, like *Gammarus zaddachi* and *Gammarus duebeni* at certain places, e. g. in Vistula Lagoon (Frisches Haff) (Grabowski et al., 2006). Its superior competitiveness is explained by its well adapted life cycle to the Baltic Sea conditions such as early maturation and short generation time, and its tolerance against environmental changes. *G. tigrinus* probably served as a vector for other Neobiota, the eel parasite *Paratenuisentis ambiguous* (Gollasch and Zander 1995). Furthermore, it can damage fishing gears and injure trapped fish when occurring in massive populations (Pinkster et al., 1977). In contrast, *G. tigrinus* serves as prey for many fish species like *Perca fluviatilis* (Daunys and Zettler, 2006).

A very small planktonic species established itself in the Baltic Sea. The potentially toxic dinoflagellate *Prorocentrum minimum* has been found first during the late 70ies of the last century (Tangen, 1980; Olenina, 2004). It has spread itself from the Skagerrak into the environmentally different Baltic Sea and has been able to form several blooms with densities up to 350 million cells l^{-1} and a relative biomass of 98% (Hajdu et al., 2000; Olenina et al, 2009). Within 15 years, it could be found in almost all regions of the Baltic Sea. In laboratory experiments, it was shown that *P. minimum* can grow in a variety of salinities and especially when adapted to low salinities it was able to grow even below 5 (Hajdu et al., 2000). This shows the potential this species has in shifting phytoplankton communities, which then could have further impact on the whole food web.

In conclusion, as seen for the North Sea, also for the Baltic Sea only some impacts of non-indigenous species have been detected that are less severe than in other regions or habitats. However, a broad variety of effects on species, habitats and ecosystems have been observed and the potential of further shifts will have to be taken into account and watched carefully in the future.

2.3 Assessment-systems on the basis of non-indigenous species

Within this chapter, existing approaches will be described and critically evaluated. Furthermore, the applicability of the proposed MSFD Indicator will be analysed and a new concept for classification will be presented.

2.3.1 Existing approaches

The establishment of non-indigenous species (NIS) and their impacts on biodiversity and ecosystem functioning are difficult to include into the determination of the Good Environmental Status (GES). A questionnaire by the Joint Research Centre (JRC) of the European Commission on the Water Framework Directive revealed that most countries tend to calculate the GES without regarding non-indigenous species, for the reason that major impacts are detected and captured by other descriptors (Vandekerkhove and Cardoso, 2010). In Germany, non-indigenous species are included in an index for large rivers but with lower scorings than most native species. A study by the Joint Research Centre (JRC) of the European Commission showed that no direct correlation between the number of non-indigenous species and the ecological status existed (JRC internal protocol). Nevertheless, there are a lot of approaches for a (risk) assessment on the basis of non-indigenous species within scientific research or in national or international political programmes. This is in accordance with the possible impacts of non-indigenous species on ecosystems which can be important for the ecological status. If the presence or the impacts of non-indigenous species shall be included in the assessment, the respective method will have to be practical, cost effective, meaningful and useful for the management. In the following chapter a selection of different approaches that have been made so far will be presented and discussed.

2.3.1.1 Trend-Indicator HELCOM

Within the HELCOM Coreset, the countries bordering the Baltic Sea decided to use a Trend Indicator as core indicator for the descriptor 'non-indigenous species' (NIS, HELCOM, 2012). However, the definition is still in progress. Currently, the indicator is called 'Trend in the arrival of new non-indigenous species' and defined as 'number of new arrivals against a baseline per six years assessment period'. The Good Environmental Status (GES) is defined as 'no new introductions of NIS per assessment unit during a six-year assessment period' reflecting the goal to minimize anthropogenic introductions of non-indigenous species to zero. Every arrival of a new non-indigenous species shall be counted during the assessment period of six years and at the end of this period the numbers will be summed up for each assessment unit. The numbers will be reported to the HELCOM Secretariat yearly but they will be reassessed every six years. Since a baseline study of the species being already introduced had to be done before, the first assessment period will begin in 2012. Consequently, the first assessment reports will be compiled by the end of 2017, and until that date, only the current and past status will be reported in the HELCOM Coreset Reports (HELCOM factsheet, 2012). For determining the baseline, study reviews (e.g. Gollasch and Nehring, 2006) and national databases[11] were taken as the basis for estimation. At the start of every new assessment period, the counter for non-indigenous species will be set to zero. All new arrivals will be counted independently, irrespective of whether the species is able to establish or not, because each new arrival can be regarded as an issue of failed management. Sometimes, it will not be possible to distinguish between species introduced by human activities and species spread naturally. For that reason, all new species will be included in the assessment. When a natural spreading can be proved, the respective species will be removed from the list. The assessment units will comprise coastal and offshore waters which are divided into sub-basins. The data from conventional biodiversity monitoring studies can be used for the assessment, but additional information will be necessary in special areas, such as harbours, main shipping lanes, where water ballast exchange takes place, or in the proximity of aquacultures. For coastal and estuarine waters, the spatial resolution of many monitoring programmes is not sufficient to reliably indicate the distribution and abundance of all non-indigenous species. Consequently, a sub-basin scale has to be used in many cases. The used data set shall be based on at least two samplings per year.

In addition to the introduction rate of species reflecting the status of the water management, the direct and indirect impacts of non-indigenous species on the ecosystems are very important. Not all newly entered species will become invasive and have a negative effect on communities, habitats or ecosystems or cause harm for humans. But for an ecosystem approach especially the invasive species and their impacts have to be taken into account. It is considered to use the Biopollution Level Index (BPL) (Olenin et al., 2007; see chapter 2.3.1.3) as additional information in combination with the trend of arrivals: Within this scheme, the trend will be calculated first, and then the new arrivals will be checked for effects.

The HELCOM trend indicator is relatively easy to calculate and well reflects the management success with regard to the arrival of new species. It has been developed for the Baltic Sea, where non-indigenous species

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[11] Poland: http://www.iop.krakow.pl/ias/Baza.aspx
Sweden: http://www.frammandearter.se/index.html
BINPAS: http://www.corpi.ku.lt/databases/index.php/binpas

are subject of various scientific studies, and a shared database compiles the information from scientific publications and national reports. The results cannot necessarily be transferred to other regions. The trend indicator avoids the problem that once a species is introduced, it will not be possible to eradicate it. As a consequence, the GES can never be reached when defined by returning to the historical status without the existence of this non-indigenous species. However, the mere existence of non-indigenous species is not necessarily reducing the GES (JRC), and even positive impacts could arise from this. But it's not only the trend that accounts for the impacts of the species and thus, it should be complemented with further information: Especially the environmental effects of the new species are important for the ecosystem and its structure and functioning. In order to gather a maximum of information, the monitoring needs to be intensified in certain areas and national data sets have to be harmonised. According to the official timetable, the GES will be calculated every six years, and the first results will be published in 2017 only.

This means that a full assessment cannot be conducted while this 6-years period is running. Alternatively, sub periods might be analysed, but this could give a distorted picture of the real situation. For example, if no non-indigenous species occurred within the first year of the assessment period, this would lead to the conclusion that the environmental status with regard to this Indicator is good and will gradually deteriorate within the subsequent years as soon as new species are introduced.

A much more objective system would be established if always the current year was taken as starting point and then the past 6 years were cumulated for the analysis. This would have the effect that the base datasets for the assessment were always compatible and furthermore, would permit the continuous detection of trends. Assessment systems being employed in the Water Framework Directive (WFD) follow this procedure.

Generally, monitoring programmes will have to be adapted according to the ecology and life strategies of organism groups. For example, a proposed sampling of phytoplankton organisms twice a year will absolutely not be sufficient, due to the seasonality of the plankton development.

2.3.1.2 Black-List System

The 'black list' system used in Germany and Austria allows the assessment of invasive species regarding their effects on the ecosystem and being combined with direct application for the management. The lists for the non-indigenous species with proved or potential negative impact are based on clear and strict criteria. Therefore, the system is verifiable and comprehensible. The criteria are simple and practicable as well (Nehring et al., 2010). The system was developed and tested with fishes and vascular plants. However, the concept of the method is defined in a way that it should be applicable for all organism groups.

The system consists mainly of three lists, the black, the grey and the white list (Figure 2-1). In the white list the alien species without evidence of a measurable impact on the ecosystem are included. The black list treats the invasive species and is divided into three parts: (i) the black list-'warning list' where all species that are not yet to find in the referred region but have a proved impact in other climatically and ecologically similar regions; (ii) the black list-'action list' with the species being invasive on a small scale in the referred region and for that an immediate management procedure is available, and (iii) the black list-'management list' containing the invasive species being existent in the studied region just on a small scale, for which no explicit management procedure exists, or that are spread in such a wide area that action is not reasonable. In the grey list, these species are pooled for that the evidence for their invasiveness is not proven. This list is also divided in two compartments: (i) the grey list-'activity list' includes the non-indigenous species for that a reasonable assumption exists that they endanger native species directly or indirectly via ecosystem change. The evidence for the negative impact has to be large enough to justify a management procedure. (ii) The grey list-'watch list' consists of the non-indigenous species for that hints exists that they endanger native species directly or indirectly via ecosystem change. Monitoring and research are the main activities for analysing these species, whereas management procedures do not seem to be indicated (Essl et al., 2008).

The main criterion to include a species into the black list or the grey list-activity list is the negative impact of this species on the biodiversity. This could be by interspecific competition, predation or herbivory, hybridisation, propagation of diseases or parasites and negative effects on ecosystem functioning and processes (Essl et al., 2008). The differentiation into the different compartments of the black list is based on the distribution of the non-indigenous species and the existence of explicit management procedures. Criteria for the grey list-watch list are characteristics of a species with regard to its habitat, its potential for reproduction, its progress in spreading, its life-form and manner and its way of coping with climate change.

Nehring et al. (2010) stated that the method for the black list system should fulfil the following requirements: It should be based on objective criteria, well documented and reproducible. The classification should be applicable with only a small number of criteria. It should be applicable for all organism groups. The concept should be simple and enabling a simple categorisation. Furthermore, the classification should be based on easily and universally available data and knowledge. This will make the assessment to a simple but reliable instrument with high acceptance and serving as a good base for communication. The system will not be an alternative for further scientific research, but in a first approach help to recognize and eliminate information gaps.

Similar approaches have been internationally developed and reviewed in Nehring et al. (2010). In Europe, several methods exist which are similar in their main features. Some are working on an international level, but up to now there is no European standardisation existent. The aggregation of the criteria is done in different ways: Many systems use the ‘one out - all out’ option and base the overall outcome on the lowest rating. Others use a scoring system where the sum of all points given for each criterion determines the classification. To a lesser extent, an alternative system is applied, such as a dichotomous key that leads to the result category.

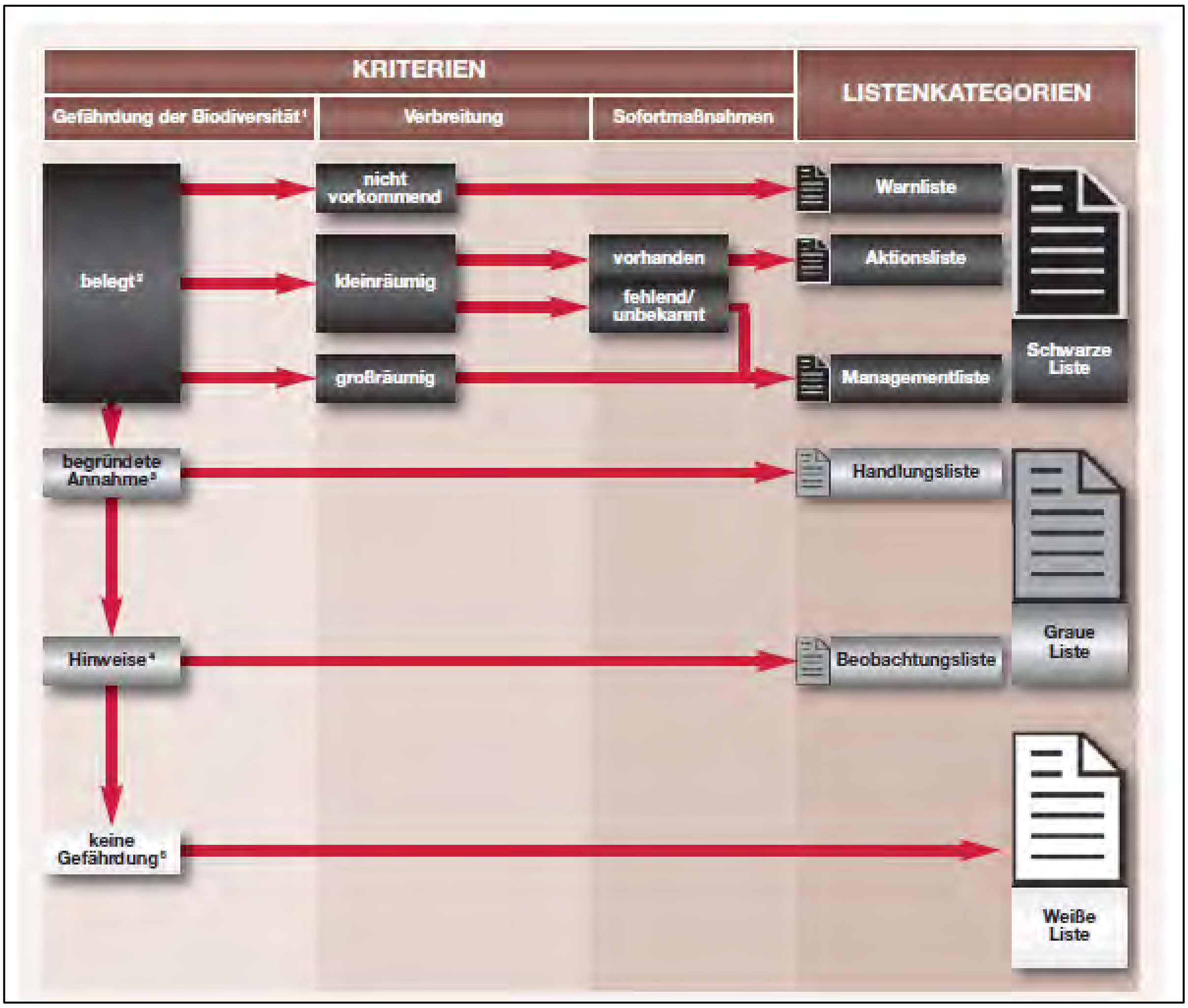

Figure 2-1: Method of classification of non-indigenous species into list categories (from Essl et al., 2008).

For aquatic ecosystems, it will not make sense to use the management procedure as criterion for a differentiation of the lists. Although Schories and Selig (2006) mentioned that the problem of non-indigenous species has been studied within several European research programs, there are not many methods to inhibit or control the spread of non-indigenous species in aquatic systems so far. However, the black list system accounts for species with a high risk to successfully invade. The classification is achieved by deducing their potential from their ecological features and from systems they have already invaded. This could be used as an early warning system and be an important base for the management to avoid the invasion of new species. In general, this system is strongly related to management procedures and facilitates easy communication. Nevertheless, the data density and/or quality from routine monitoring programmes very often do not meet the requirements for the respective classification of species. At the moment, the definition of boundary conditions is mainly qualitative and based on expert judgement or literature data.

2.3.1.3 Biopollution Level Index

The biopollution level index (BPL) has been developed by Olenin et al. (2010) in order to create a measure for the influence of non-indigenous species in an ecosystem. The system allows comparison of the different impacts in space and time and is useful for developing and applying management procedures. It evaluates the impact of non-indigenous species on all levels of an ecosystem. The approach is using information on abundance and distribution of the non-indigenous species and its impacts on species, community, habitat and ecosystem functioning. The assessment is defined for a given assessment unit and assessment time. In a first step, the abundance data of a non-indigenous species is used to calculate its share in the whole community. Three assessment levels are defined, from 'low' (only minor contribution) to 'moderate' (less than 50%) and 'high' (Proportion more than 50%). The distribution of the species is described by the terms 'one locality', 'several localities', 'many localities' and 'all localities'. The combinations of both characters are narrowed down to five classes (Table 2-4) which reflect the different phases of the invasion: arrival (Class A), establishment (B), expansion (C, D and in extreme cases E) and adjustment.

Table 2-4: Five classes representing the abundance and distribution range of alien species (AS) (from Olenin et al., 2007).

Code	Description
A	An AS occurs in low numbers in one or several localities.
B	An AS occurs in low numbers in many localities or in moderate numbers in one or several localities or in high numbers in one locality.
C	An AS occurs in low numbers in all localities, or in moderate numbers in many localities, or in high numbers in several localities.
D	An AS occurs in moderate numbers in all localities, or in high numbers in many localities.
E	An AS occurs in high numbers in all localities.

Table 2-5: Classification of alien species (AS) impact on native species and communities (from Olenin et al., 2007).

Code	Impact	Description
C0	None	No displacement of native species, although AS may be present. Ranking of native species according to quantitative parameters in the community remains unchanged. Type-specific communities are present.
C1	Weak	Local displacement of native species, but no extinction. Change in ranking of native species, but dominant species remain the same. Type-specific communities are present.
C2	Moderate	Large scale displacement of native species causes decline in abundance and reduction of their distribution range within the assessment unit; and/or type-specific communities are changed noticeably due to shifts in community dominant species.
C3	Strong	Population extinctions within the ecosystem. Former community dominant species still present but their relative abundance is severely reduced; alien species are dominant. Loss of type-specific community within an ecological group.
C4	Massive	Population extinction of native keystone species. Extinction of type-specific communities occurs within more than one ecological group.

Then the magnitude of impacts will then independently classified by using five classes for each level (Table 2-5 to Table 2-7) and coding the level and the strength of impact ('none' = Community C0, Habitat H0, Ecosystem E0, 'weak' =C1, H1, E1, 'moderate' = C2, H2, E2, 'strong' = C3, H3, E3, 'massive' = C4, H4, E4).

Table 2-6: Classification of alien species (AS) impact on habitats (from Olenin et al., 2007).

Code	Impact	Description
H0	None	No habitat alteration.

Code	Impact	Description
H1	Weak	Alteration of a habitat(s), but no reduction of spatial extent of a habitat(s).
H2	Moderate	Alteration and reduction of spatial extent of a habitat(s).
H3	Strong	Alteration of a key habitat, severe reduction of spatial extent of habitat(s); loss of habitat(s) within a small area of the assessment unit.
H4	Massive	Loss of habitats in most or the entire assessment unit, loss of a key habitat.

Table 2-7: Classification of alien species (AS) impact on ecosystem functioning (from Olenin et al., 2007).

Code	Impact	Description
E0	None	No measurable effect.
E1	Weak	Measurable, but weak changes with no loss or addition of new ecosystem function(s).
E2	Moderate	Moderate modification of ecosystem performance and/or addition of a new, or reduction of existing, functional group(s) in part of the assessment unit.
E3	Strong	Severe shifts in ecosystem functioning in part of the assessment unit. Reorganisation of the food web as a result of addition or reduction of functional groups within trophic levels.
E4	Massive	Extreme, ecosystem-wide shift in the food web and/or loss of the role of a functional group(s).

Additionally, a confidence level (‘high’, ‘moderate’, ‘low’) is defined for each step of the assessment. The biopollution level can be assessed by listing all possible combinations of the abundances and distributions according to their impact codes for each class (Figure 2-2). Because some combinations are unlikely to occur, the matrix can be reduced and the authors provide a decision support scheme to determine the level of biopollution in one of five classes from ‘none’ to ‘massive’ impact.

These classes reflect the five ecological quality classes sensu the Common Implementation for the EG Water Framework Directive (European Community 2000). The decision will be based on the highest impact; the species has reached within one level. The assessment should be conducted for each non-indigenous species in the assessment unit and the overall biopollution level for this unit will then be determined according to the greatest impact level for at least one species (‘one out, all out’ rule).

Thus, the species with a lower level will not influence the final assessment. Knowledge about the relative abundance of the non-indigenous species, their distribution and their impacts is needed for calculating this index. If this information is missing, data from the species with the strongest impact should be used. In most cases, this species will be easily identified. Similarly, the division of the assessment of impacts into the three categories may facilitate the calculation of the index even if knowledge about the impacts is limited.

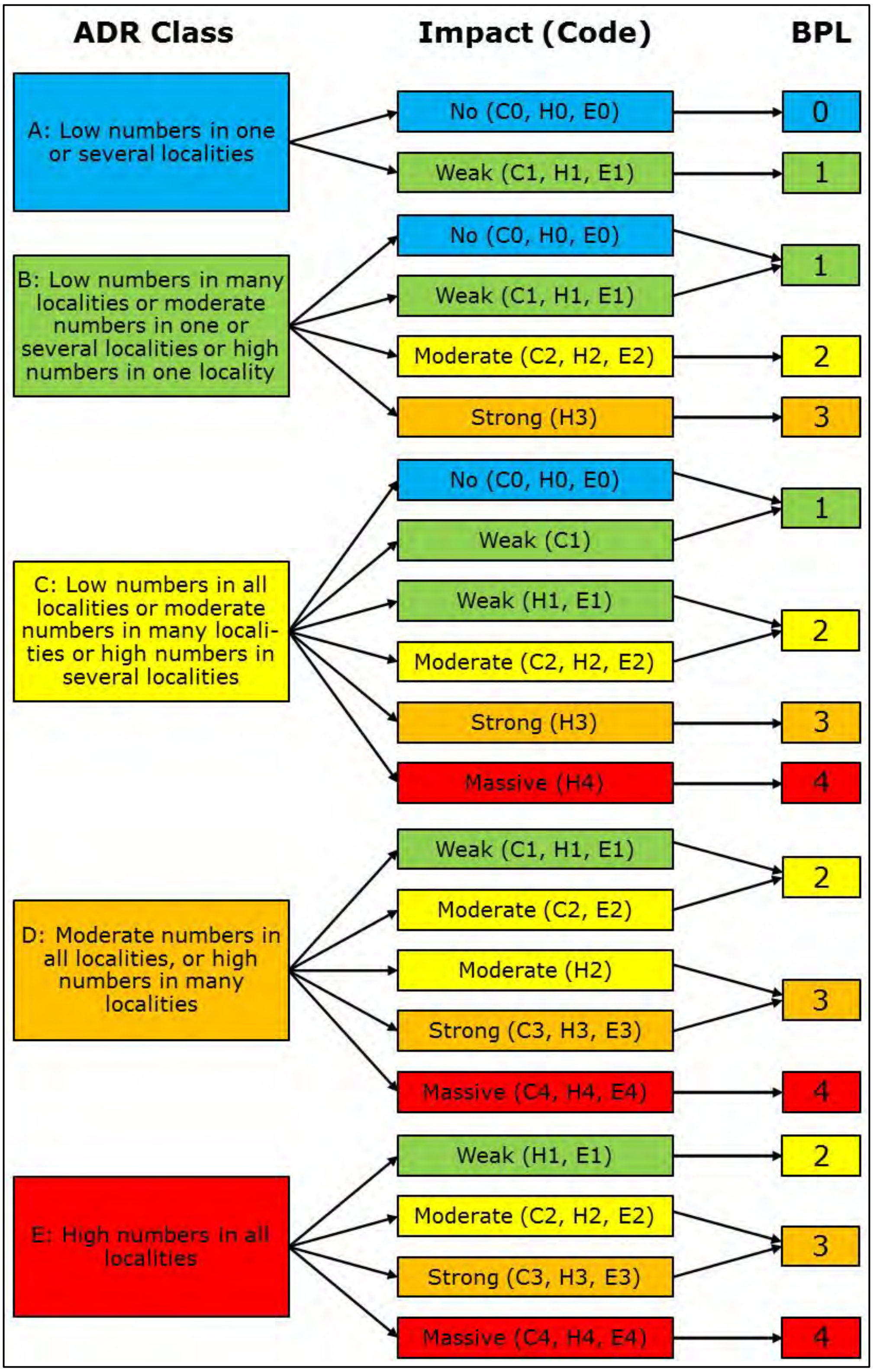

Figure 2-2: The decision support scheme for assessment of biopollution level (from Olenin et al., 2007).

However, compared to other approaches the BPL needs a lot of data and information and extended ecosystem research. Boundaries between classes are not based on discrete and quantitative data. The assessment relies on expert judgement and is based on qualitative descriptions sometimes leading to biased results.

Although it will be possible to have assessment units with low numbers of alien species with a good environmental status if not having any impact on the system, it will be highly questionable how to achieve a good environmental status without eradicating the non-indigenous species, the latter being practically impossible.

Another weakness of the system is that it will make no difference for the status whether just one or more species with significant impacts on the system have been identified. In general, integration of all possible effects into an overall assessment should be preferable, compared to a system where only the worst assessment feature determines the result and where the information on the impact of other species gets lost.

2.3.1.4 Biocontamination Index

Arbačiauskas et al. (2008) developed the biocontamination index which to their opinion is sufficient to provide an integrated estimation for the status of water bodies concerning non-indigenous species. This system will not require much research or expert knowledge and routine monitoring data can be used. Two metrics are needed for the calculation, the abundance contamination index (ACI) and the richness contamination index (RCI). The indices are calculated as the proportions of the non-indigenous species abundance or number of orders of the total abundance or orders, respectively. Then the site-specific biocontamination index (SBCI) can be derived from a matrix of the values of these indices. Five classes of biocontamination ranging from 0 (‘no’ contamination) to 4 (‘severe’ contamination) are defined (Table 2-8).

Table 2-8: Assessment of site-specific and integrated biocontamination indices (SBCI and IBCI, correspondingly) based on abundance contamination index (ACI) and ordinal richness contamination index (RCI). SBCI and IBCI classes: 0 (no biocontamination, ‘high’ ecological status, blue cell), 1 (low biocontamination, ‘good’ ecological status, green cell), 2 (moderate biocontamination, ‘moderate’ ecological status, yellow cells), 3 (high biocontamination, ‘poor’ ecological status, orange cells), 4 (severe biocontamination, ‘bad’ ecological status, red cells), (from Arbačiauskas et al., 2008).

RCI	ACI				
	none	0.01 – 0.10	0.11 – 0.20	0.21 – 0.50	>0.50
none	0				
0.01 – 0.10		1	2	3	4
0.11 – 0.20		2	2	3	4
0.21 – 0.50		3	3	3	4
>0.50		4	4	4	4

These five classes reflect the five ecological quality classes sensu the Common Implementation for the EG Water Framework Directive (European Community, 2000). The lowest quality limit is reached when more than half of the orders are represented by non-indigenous species or when their abundance exceeds 50 % of the total abundance. When multiple ACI and RCI for the same ecosystem or assessment unit are available, the integrated biocontamination index (IBCI) can be derived on the basis of their means.

In this study (Arbačiauskas et al., 2008) the investigation of benthic macroinvertebrates of several European inland waterways showed temporal and spatial trends of biocontamination, indicating that richness contamination precedes abundance contamination. Additionally, the separation between moderate, high and severe biocontamination was mainly depending on abundance contamination values, while the good and moderate environmental status was distinguished primarily by richness contamination. Therefore, the authors advise that both measures which characterize different aspects of community structural organisation should be taken into account when assessing biocontamination. The relative abundance of individuals of non-indigenous species reflects their dominance in the community, while their contribution to community disparity is measured by richness contamination.

As for the Trend Indicator of the HELCOM CORESET, the effects of the non-indigenous species are not assessed with this index. The calculation is easy and can be done with data from routine monitoring. The boundaries between classes are defined by discrete values. The system was developed with benthic macroinvertebrates and according to the authors is assignable to other taxonomic groups as fish or macrophytes. It remains to be tested whether it can be applied in general. It will also remain questionable how to deal with order richness and if this is always practicable. Orendt et al. (2010) criticised that the accuracy and robustness of the index is reduced if non-indigenous species belong to the same order as native species.

2.3.1.5 Risk Assessment Toolkit

Panov et al. (2009) developed a conceptual risk assessment model for invasive species introductions via European inland waterways. They recommend their approach for application as part of the Common Implementation Strategy of the European Commission Water Framework Directive for integrated river basin management in Europe. The system includes risk assessment protocols and water quality indicators for non-indigenous species. Panov et al. (2009) developed their model within the socio-economic context of the driving forces-pressures-state-impact-response (DPSIR) framework used by the European Environment Agency (EEA), which structures the relations between the environment and socio-economic activities[12]. By presenting a qualitative model, the problem of numerical data gaps is avoided as well as the problem of dealing with scientific uncertainties. By adapting the DPSIR model for assessing the risk of non-indigenous species, the pathways introducing such non-indigenous species can be considered as driving forces for each assessment unit. The pathways can be divided into three groups and are separately assessed for each unit. If there is no or low certainty about the existence of this pathway in the assessment unit, it will be defined as a 'low-risk pathway'. If there is a high level of certainty about its existence but no evidence about the introduction of non-indigenous species for the assessment unit, it will be a 'high-risk pathway'. With evidence for both, existence of the pathway and introduction of species, the pathway will be considered as an 'extreme-risk pathway'. Combining existing pathways with environmental matches of donor areas additionally gives information about 'high risk donor areas' for the assessment unit. The last three give three environmental indicators for 'driving forces': For the environmental indicators 'pressures', the pathway-specific biological contamination rate (PBCR) can be calculated, which is determined by the number of recorded non-indigenous species per time and assessment unit introduced by a certain pathway. It is also useful to specify the pathways into high-risk pathways (PBCR is 0) and extreme-risk pathways (PBCR>0).

Several indices can be used to describe the environmental indicator 'state'. These include the SBCI and IBCI described above (Arbačiauskas et al., 2008), and the biological contamination level BCL. The last corresponds to the number of established non-indigenous species since 1900 and reflects the invasibility of the ecosystem.

'Impacts' can be reflected by an index such as the biopollution level index, which has been described above. Because of the intensive research needed to obtain the data for this index, Panov et al. (2009) suggested a risk-based assessment of invasiveness of the established alien species and developed a species-specific biological pollution risk (SBPR) index. It is based on three descriptors that estimate the invasiveness of a species.

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[12] http://ia2dec.ew.eea.europa.eu/knowledge_base/Frameworks/doc101182

These are the potential to spread, the potential for establishment in a new environment and the potential to cause negative ecological and socioeconomic impacts. As useful for this categorisation the authors suggested a modification of the grey, white and black listing system (chapter 2.3.1.2). When no information on all three descriptors is available, the species should be assigned to the grey list and its SBPR level, marked as unidentified ('N/A'). If the only information on a species is about either its rapid dispersal or its establishment, the species can be attributed to the white list and its SBPR would be 1 (low level of invasiveness). A species with rapid dispersal and establishment would be assigned to the white list with an SBPR of 2 (moderate level of invasiveness). Information about adverse impacts of a non-indigenous species would qualify it for the black list regardless of the information on dispersal and establishment, and its SBPR would be 3 (high level of invasiveness). In combination with relative abundance data, the SBPR values can be used for estimating the integrated biological pollution risk (IBPR) of an assessment unit. The value would be 0 if no non-indigenous species was present in the assessment unit. This would correspond to the 'high' ecological status sensu the Common Implementation Strategy of the European Commission Water Framework Directive (European Community 2000). Low abundances (less than 20% of total abundance) of species from the white or grey list would match an IBPR index of 1 and reflect a 'good' environmental status. If abundances of non-indigenous species from the white or grey list exceeded 20% of total abundance, the IBPR index would be 2 and the environmental status 'moderate'. When species from the black list were present in relatively low or high abundances (threshold of 20%) the IBPR index would have values of 3 and 4, respectively and the status 'poor' and 'bad' would be assigned. The status estimated by the IBPR index is generally lower than that one calculated with SBC and IBC.

For the 'response' part of this framework, the development of an online risk assessment toolkit with an early warning service is proposed. It should be a user-friendly platform for reporting environmental indicators and recommendations for risk management to stakeholders. It shall include risk assessment protocols, supporting information systems, the electronic journal 'Aquatic invasions' and transmit the information to the level of end users.

The described system is a very complex approach. Some of the used tools are discussed above; further mentioned indices have similar advantages and disadvantages as discussed before. Interesting is the combination of different approaches and their integration in the DPSIR system. The authors suggest the environmental indicators for non-indigenous species as a useful tool for the risk management including preventive actions, and mention the BCR and PBCR as indicators for preventive management measures. They recommend the SBC index, the IBC index, and in particular, with regard to the precautionary approach, the IBPR index as cost effective elements for the assessment of the environmental status. The toolkit includes an early warning system and online tools for assessment and communication.

Since the system has been developed for European waterways as pathways for introducing new species, it is questionable whether it might be possible to transfer it to more complex ecosystems such as the open sea. For example, transport processes in the North Sea and Baltic Sea are very complex and take place in a three-dimensional spatial structure, while the DPSIR system focusses on two-dimensional transport processes in river systems. For an application to the marine environment, the DPSIR system will have to focus on improvement of the transport processes in order to evaluate the different marine pathways. Furthermore, stronger connections and feedbacks between the single elements will have to be established. Individual tools and indices may be used as described before. However, from the study results it remains unclear how the single components could work together in a completed assessment system and not only function as isolated elements.

2.3.1.6 Evaluation, Résumé, Conclusion

The current approaches described so far are mostly in different developmental stages. Very often, verification is still pending, and all systems have several advantages and disadvantages at the same time. Some models promise easy assessment with standard monitoring data, others need more information and/or expert judgement. In particular, the effects of non-indigenous species on the ecosystem are very difficult to assess and depend on expert judgement. Here, further intensive scientific research will be required. Especially if effects are included in the assessment, it will be difficult to reach the GES, because management measures are mainly reduced to the avoidance of new introductions. Most approaches have a connection to management measures; however, the practicability and applicability of the systems differs. Using the 'one-out, all-out' rule might ease the assessment process, but on the other hand, the information might be significantly reduced when different situations are merged into one status. Boundaries between classes are more often defined by qualitative information then by objective data.

Consequently, an applicable and ease-to-use assessment approach should fulfil the following requirements: It should be based mainly on data collectable by routine monitoring. It should be possible to formulate quantitative and objective criteria for the boundaries of the status classes. The assessment should be applicable in a way that results can be presented every year and not only by the end of a six-year assessment period. The JRC suggests including non-indigenous species into the assessment only if they have an impact on the structure or functioning of the ecosystem (JRC internal protocol). However, the control of preventive management measures requires at least the continuous survey of newly appearing species. In contrast, simple counts of invader numbers or comparison of non-indigenous to indigenous species numbers will yield no information on the impacts of the non-indigenous species, which certainly will reduce the environmental status of the ecosystem. Thus, a functional assessment system should combine both the (relative) abundance and the effects of a non-indigenous species. Instead of using the 'one-out, all-out' option, the assessment system should follow an integrative approach in order to register and assess as many accumulative effects as possible. Within this context, it will be very important that the good environmental status will be attainable without eradicating non-indigenous species being already present in the marine system. And last but not least, an early warning system should be included in the assessment system in order to apply preventive management actions whenever needed.

2.3.2 Discussion of practicability and applicability of the given MSFD Indicators

As noted before, the Annex I of the MSFD defines 11 Descriptors for characterising the GES. In the 'Commission decision on criteria and methodological standards on good environmental status of marine waters' (1st September 2010, 2010/477/EU), these Descriptors are specified in detail and broken down into 29 Criteria and 56 Indicators. The latter refer to particular parameters of the features to be analysed and shall enable a simplified assessment of the complex ecosystem.

For the Descriptor 2 in question here ('Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems'), 2 Criteria and 3 Indicators have been defined by the commission decision:

Criterion 2.1: Abundance and state characterisation of non-indigenous species, in particular invasive species

> **Indicator 2.1.1:** Trends in abundance, temporal occurrence and spatial distribution in the wild of non-indigenous species, particularly invasive non-indigenous species, notably in risk areas, in relation to the main vectors and pathways of spreading of such species

Criterion 2.2: Environmental impact of invasive non-indigenous species

Indicator 2.2.1: Ratio between invasive non-indigenous species and native species in some well-studied taxonomic groups (e.g. fish, macroalgae, molluscs) that may provide a measure of change in species composition (e.g. further to the displacement of native species)

Indicator 2.2.2: Impacts of non-indigenous invasive species at the level of species, habitats and ecosystem, where feasible

In the following, the practicability and applicability of these three Indicators will be critically evaluated.

2.3.2.1 Indicator 'trends in abundance'

Using this Indicator, the density of individuals of non-indigenous species as a function of their temporal and spatial distribution within a specified area will be determined and used for the description of the good environmental status. In particular those areas will be considered that are highly endangered by certain impact vectors, for example harbours and surrounding areas, where the exchange of ballast water plays a major role, or regions with intense aquacultures.

Generally, the determination of abundances of non-indigenous species is reasonable and necessary. In addition to the simple information on the occurrence, the degree of distribution as well as possible impacts can be traced within the ecosystem. However, the abundances of non-indigenous species and respective developments and trends should not be considered in isolation, but also in relation to the number of organisms of the whole community. This is in accordance with the 'abundance contamination index' (ACI, see Arbačiauskas et al. 2008) described in chapter 2.3.1.4.

The current monitoring programmes established in the North Sea and Baltic Sea already record the abundances of different organism groups on the taxonomic level; as a consequence, no further work will be necessary. Provided that the non-indigenous species can be clearly identified without ambiguity, these recordings also include the densities of these species. With regard to the specific requests of the MSFD, some adjustments of sampling frequencies and sampling locations will be necessary, in order to also include the risk areas. For determining the abundances, no new monitoring programmes will be required, and existing data series could and should be included for the analysis of trends.

HELCOM is using the number of newly determined non-indigenous species within a specified time period and in a confined area as a Core indicator (see chapter 2.3.1.1). This will not allow drawing any conclusions with regard to the ecological status of the system, since the mere occurrence of a (new) non-indigenous species in low numbers and in a confined area does not necessarily mean a deterioration in the status. On the other hand, a correct and careful interpretation of this parameter might provide valuable information on the success of management measures. By the existing, possibly modified monitoring programmes and related taxonomic analyses, the new species will automatically be recorded.

2.3.2.2 Impact Indicator 'species composition'

From the textual description of this Indicator, it is not obvious whether it stands only for the relation of species numbers of non-indigenous species to the number of indigenous species, or for the respective abundance and/or biomass relations. If the former was valid, then this relation would refer to the 'richness contamination index' (RCI) described in chapter 2.3.1.4 (Arbačiauskas et al. 2008). A critical point of this approach is the fact that this RCI is based on the taxonomical level of orders. A 'real' richness factor should be determined on the species level. On the other hand, it is very often not possible to identify the organisms down to the species level within the normal routine monitoring programmes. Furthermore, it is strongly dependent on the monitoring strategy whether or not all species can be detected in an assessment area.

The mere number of non-indigenous species in relation to the number of indigenous species will not necessarily provide a good measure for the ecological status of a system. Within a defined area, such a parameter, especially if given as a trend, might well function as a mark for the displacement of indigenous species, especially when the quotient is high. But in most cases, the ratio will result in low values being not directly correlated with the status of the system. Consequently, the abundance or biomass ratios of non-indigenous to indigenous species could be a much better measure (see chapter 2.3.2.1).

While it will be a lot of additional work to analyse the overall species richness in a system, based on existing monitoring programmes, it should be possible to determine the order-based RCI from the data of running samplings. To calculate ratios of abundances or biomasses will also mean no extra effort. Generally, the non-indigenous species will almost completely be identified and classified. For the indigenous species it will not be so important to analyse down to the species level as long as the overall abundances and/or biomasses are correctly determined.

2.3.2.3 Impact Indicator 'system effects'

By using this Indicator, the impacts of non-indigenous species on indigenous species, on habitats and on the overall structure of the ecosystem shall be assessed. The effects can be of a various nature and impact the system on different levels. Chapter 2.2.3 gives an overview. It is not always possible to directly assess the impact of new species on the different and often complex components of the ecosystem on the basis of data from existing monitoring programmes. In order to get an idea of possible effects, it could become necessary to carry out extensive and comprehensive investigations. Such studies have been conducted, but it is often not possible to verify the results within the routine monitoring.

Nevertheless, non-indigenous species can have great – mostly negative – impacts on the biodiversity, structure and function of aquatic ecosystems with long-lasting effects on the status. This problem will be highlighted by two examples from the North Sea and Baltic Sea. For that purpose, a representative plankton and benthos organism has been chosen for each region.

2.3.2.3.1 North Sea

Karenia mikimotoi

Karenia mikimotoi (Miyake & Kominami ex Oda) G. Hansen & Ø. Moestrup is a toxic planktonic dinoflagellate that forms world-wide blooms, so called 'red tides' with up to several million cells per litre. The first occurrence was reported in Japan in 1935 (Oda, 1935). Further incidences were recorded at the east coast of the USA in 1957 (Hulbert, 1957) and in the North Sea at the northwest coast of Norway in 1966 (Tangen, 1977). Since then, mass occurrences have been reported from France over Ireland to Norway (Pingree et al., 1975; Ottway et al., 1979; Silke et al, 2005; Vanhoutte-Brunier et al., 2008). The world-wide expansion coincided probably with the increasing usage of ballast water by large ship vessels.

However, there exists some taxonomic confusion about this species group. One reason for this may be inaccurate taxonomic determination. Gomez (2008) gives some examples from the North-European coasts (*Gymnodinium punctatum* Pouchet, *Gymnodinium minus* Lebour, *Gyrodinium lingulifera* Lebour, *Gymnodinium pygmaeum* Lebour). Furthermore, revisions of the taxonomy led to aggregations of descriptions and to renaming. Takayama and Matsuoka (1991) described *Gymnodinium nagasakiense* as being conspecific to *Gymnodinium mikimotoi*. Currently, both names are synonyms for *Karenia mikimotoi* after a revision of the taxonomy by Daugbjerg et al. (2000). For *Gyrodinium aureolum* the picture is not clear. Two morphotypes from North America and Europe have been described with the same name. It seems that in contrast to the European morphotype the occurrences from North America do not correspond to *Gymnodinium mikimotoi* and thus *Karenia mikimotoi* (Hansen, 2000). However, they are closely related. The confusion by diverse synonyms and especially the difficulties in identification could possibly be solved or at least minimised by genetic analytical methods (Daugbjerg et al., 2000; Al-Kandari et al., 2011) or by methods of spectral absorption (Staer and Cullen, 2003). These methods could probably be used also for identification in monitoring (Zhang et al., 2009; Staer and Cullen, 2003).

World-wide impacts on vertebrates and invertebrates have been observed in correlation to the blooms of *K. mikimotoi* (Lu and Hodgkiss, 2004; Honjo 1994; Silke et al., 2005). First reports from Europe referred to massive impact on benthic communities with mortalities especially of lugworms (*Arenicola marina*) but also with effects on other benthic invertebrates and several fish species (Ottway, 1976; Graneli et al., 1989). Additionally, fish farms were affected and it occurred that complete fish stocks died in their cages (Jones et al., 1982; Jenkinson and Connors, 1979). Lu and Hodgkiss (2004) estimated the commercial loss of a bloom in Hong Kong in 1998 of about USD 40,000,000 for the fishing industry. Two thirds of the companies were concerned and almost all fishes in the affected cages died during this record bloom. At the coasts of Ireland several blooms with observed fish mortalities had occurred, before in 2005 an especially intense bloom brought mass mortalities of sea urchins (*Echinocardium chordatum*) and lugworms (*Arenicola marina*) and had a strong impact on the shellfish farms along the coast. The spatial and temporal extend of the bloom had been extraordinary as well: Several km^2 had been affected and the bloom persisted longer than two months (Silke et al., 2005). Apart from showing a strong lethal effect, several studies produced evidence for further effects, such as reduced feeding rates in juvenile oysters and other bivalves and abnormal behaviour, e.g. valve closure (Cassis, 2005; Matsuyama et al., 1999, Widdows et al., 1979).

The exact causes are still in question. Besides toxins also oxygen depletion is held responsible for the lethal effect. This is supported by the observed behavior of fishes and other organisms (Jenkinson and Connors, 1979). Massive occurrence of dinoflagellates may lead to anoxic conditions due to respiration during night and to degradation of senescent cells by bacteria. Especially, the benthic fauna in shallow waters may be affected by such effects; the stronger the more the water is stratified by temperature or the intrusion of freshwater. Furthermore, respiratory organs as the fish gills could be clogged by the dinoflagellates (Parker, 1982). Additionally, an increase in viscosity of water was observed during such blooms, probably caused by exudations of the cells as mucopolysaccharides (Jenkinson and Connors, 1979; Potts and Edwards, 1987). In this case, more energy will be needed to pump the viscous fluid through the gills. This leads to increased oxygen demand while the viscosity decreases the diffusion of oxygen from surrounding areas in general. Jenkinson and Arzul (1998) had shown in an experiment that rheological effects alone could lead to the death of the fishes. Also, fluctuations between oxygen depletion and super-saturation may contribute to the mortality of the organisms (Silke et al., 2005).

Furthermore, the impact of toxins had been considered as a possible reason for the effects. Cultures of *Gyrodinium aureolum* were shown to be lethal for rainbow trout (*Oncorhynchus mykiss*, formerly *Salmo gairdneri*) within 24 hours without clogging of gills by the cells (Roberts et al., 1983). At further histological examination the fish showed necrotic degeneration and disintegration of the lamellar epithelium. Jones et al. (1982) found cellular damage to gills and guts of fishes of a salmon farm as well. The characteristics suggested necrosis causing toxins as the cause. However, it seemed to be a toxin being different from the toxin isolated from the related dinoflagellate *Gymnodinium breve*. The histological examinations by Mitchell and Roger (2007) showed necrosis as well in gills, guts and the liver of fishes that died on a fish farm during the massive bloom at the Irish coast in 2005. The toxic effects seemed to have based on damage of the tissues. Further histological inspections were done by Smolowitz and Shumway (1997) who found effects on juveniles of several commercial important bivalve species. However, the effects were species-specific and ranged from high mortality rates to no visible histological alterations. Often the toxic effect could not directly be detected in experiments with filtrates of the dinoflagellate cultures (Widdows et al., 1979; Matsuyama et al., 1999). Matsuyama et al. (1999) showed that filtrates had different effects on several bivalve species and that sometimes the mussels did not react to the treatment. Some hours after the filtration, no effects could be detected at all. This gave evidence for the volatility of the excreted substances of *K. mikimotoi* and was confirmed by Satake et al. (2002; 2005) by isolating two cell-damaging polyethers Gymnocin-A and Gymnocin-B from *K. mikimotoi,* which were hardly soluble in water. These toxins resemble very much the structurally similar brevetoxin of *Gymnodinium breve* but in experiments proved to be less toxic on fishes. During blooms of *K. mikimotoi*, fishes with dinoflagellates in their gills were found (Mitchell and Rodger, 2007). Widdows et al. (1979) demonstrated a reduced clearance rate when *Mytilus edulis* was exposed to the cells directly and found bivalves with cells around their gills and in their stomach. Thus, the effect possibly is expressed only through direct contact of the dinoflagellate with the tissues of the organisms. Polyunsaturated fatty acids seem to play an important role in the toxins (Yasumoto et al., 1990; Bodennec et al., 1995). Experiments demonstrated their impact on the embryonic development of sea urchins (Sellem et al., 2000), the deformation of bivalve shells (Erard-Le Denn et al., 1990) and the mortality of fishes and other organisms (Sola et al., 1999). Although the polyunsaturated fatty acids may be allopathic on competitors (Gentien and Arzul, 1990; Arzul et al., 1993), they also might be toxic for *K. mikimotoi* itself (Gentien et al., 2007).

Despite *K. mikimotoi* can be found in waters with a broad temperature and salinity range (Nielsen and Tønseth, 1991), the blooms are promoted by high temperatures and stratification of the water by surface warming and/or intrusion of fresh water in estuaries. Dinoflagellates will have an advantage when the water is stratified because they are able to migrate in deeper and nutrient rich water layers. Koizumi et al. (1996) reported daily migrations of *K. mikimotoi* of more than 20 meters with a speed of about 2.2 meters per hour. Thus, this species probably may uptake nutrients in deeper layers during night and do photosynthesis at the surface during day. Since it is also able to photosynthesise at lower light intensities than other algae groups (Yamaguchi, 1994), it often has abundance maxima near the pycnocline where it profits from the nutrients of the bottom layer (Kimura et al., 1999). For the same reason, blooms often develop near frontal systems (e.g. Ushal Front System in the English Channel) at the border between stratified and mixed waters (Pingree et al., 1975; Morin et al., 1989; Chang and Carpenter, 1979; Holligan, 1984; Vanhoutte-Brunier et al., 2008). These blooms then may be transported to the coasts with the frontal system (O'Boyle and Rain, 2007; Ottway et al., 1979) and affect the benthic communities and fish and shellfish farms.

Crassostrea gigas

The Pacific oyster (*Crassostrea gigas*) is a benthic filter feeder that ingests bacteria, protozoa, many diatoms, larvae of invertebrates and detritus. It originates from Japanese and south-east Asian estuaries and marine coastal waters, where it lives in the intertidal and shallow sub tidal zones. *C. gigas* has a salinity range from 10-42, but can also survive in salinities of 5. The optimal temperature ranges from 4 to 35°C. For reproduction a salinity range from 23 - 36 and a temperature of more than 18°C is required.

Since the 19th century American and Portuguese oysters (*Crassostrea virginica* and *Crassostrea angulata*) were imported in European waters to revive exploited stocks of European oyster (*Ostrea edulis*), but without success.

To counter the decline of *O. edulis*, as a result of overfishing, *C. gigas* had been introduced by humans in European waters to create an efficient oyster aquaculture. The first import occurred in 1964 in the Oosterschelde, Netherlands, with spat from British Columbia, Canada. It was assumed that the oyster would not reproduce at these latitudes, because of her temperature requirements. In 1986 *C. gigas* was also imported for aquaculture activities in Germany. Commercial farming activities take place near the island of Sylt, in the northern Wadden Sea, with spat taken from British and Irish hatcheries (Reise, 1998; Nehring, 1999).

In 1975, triggered by a warm summer, the first natural spatfall occurred in the Oosterschelde and in the beginning 1980's *C. gigas* spread in other Dutch estuaries (Wolff and Reise, 2002; Smaal and al., 2009). The first records of *C. gigas* in the Wadden Sea were noticed in 1983 near the island of Texel, Netherlands (Bruins, 1983). In Germany, the first settlement of *C. gigas* was observed in 1984 in the western Wadden Sea, probably widespread from the Netherlands by natural means (Meixner, 1984). The species entered the East Frisian Wadden Sea in 1998 (Wehrmann et al., 2000). The first natural spatfall leading to successful settlement of *C. gigas* in the northern Wadden Sea occurred in 1991. These specimens came from an aquaculture at Sylt (Reise, 1998) and spread up to the North Frisian Wadden Sea. (Nehring, 2003 a; Diederich et al., 2005). Since then *C. gigas* has been found in the entire Wadden Sea (Reise et al., 2005; Nehring et al., 2009). In 2009, the first free living specimens of *C. gigas* were found on the German Baltic Sea coast, which may have been dispersed from the Kattegat by natural transport processes (Wrange et al., 2010).

Human activity had been the main pathway for the introduction of *C. gigas* in European waters. *C. gigas* has been expected to replace *O. edulis* for commercial aquaculture (Nehring, 1999; Wolff and Reise, 2002). As described above, the further dispersion has taken place by natural spatfall, which were dispersed by currents (Wehrmann et al., 2007). The combination of rapid growth, resistance to highly turbid areas and environmental stress and the capacity to adapt to various environmental conditions is one of the main reasons why *C. gigas* is considered being an invasive alien species (CABI, 2010).

In the Wadden Sea *C. gigas* competes against *Mytilus edulis* predominantly for space. The larvae of *C. gigas* depend on a hard surface to settle. In the Wadden Sea, mostly dead shell material and epibenthic *Mytilus* beds provide a secondary hard substrate. This has led to an overgrowth of mussel beds and consequently, to the development of massive oyster beds (Nehls et al., 2006; Nehring et al., 2009). However, *C. gigas* has a preference to settle on conspecifics, which results in an increased forming of oyster reefs (Diederich, 2005b; Wehrmann et al., 2006). In succession, they themselves provide substrate for *M. edulis* and thus, combined mussel reefs have formed (Fey et al., 2010). Nevertheless, there are still mussel beds in the Dutch Wadden Sea which are unaffected by spatfall from *C. gigas*. On the other hand, all mussel beds in the German Wadden Sea have been transformed into combined mussel-oyster reefs and *M. edulis* has lost his capacity as a 'habitat-engineer' (Nehls et al., 2006; Wehrmann et al., 2007). Also, other species live in and on the oyster reefs. This biocoenosis differs from those in *Mytilus* beds (Kochmann et al., 2008) and therefore could provide a shift in the species composition in the Wadden Sea. The oyster reefs seem to form a suitable substrate for other alien species (*Sargassum muticum* (Lang and Buschbaum, 2010), *Crepidula fornicata*, *Stylea clava*, *Austrominius modestus* and *Hemigrapsus* (Liebich &etal., 2007; Thieltges et al., 2009; Wolff et al., 2010).

In addition to the competition for space, there is probably also a competition for food, resulting from identical feeding behaviour (filter feeder). This could be an explanation for the decline of *M. edulis*, since *C. gigas* has appeared in the Wadden Sea. However, the decline of *M. edulis* has also been caused by changes in climate conditions as shown by experiments carried out by Nehls et al., (2006). *C. gigas* is unaffected by those changes, which results in different reproduction of either mussel.

It is to be noted that *C. gigas* is not restricted only to mussel beds, as in recent times it also settled on snails (*Littorina littorea*) (Eschweiler and Buschbaum, 2011) with negative consequences. The body dry weight of overgrown periwinkle had been half of that of snails without oyster and the crawling speed of overgrown snails had been significantly slowed down. Furthermore, a negative effect on the reproduction had been observed.

In addition, *C. gigas* has an impact on benthopelagic interactions. As a consequence of the higher filtration capacity of *C. gigas* in comparison to *M. edulis* (based on total biomass), there has been a shift from pelagic to benthic consumers as a result of food depletion in the water column (Diederich, 2005a). In contrast to that, a higher pelagic and benthic productivity could possibly occur as *C. gigas* releases nutrients and pseudofaeces into the water column.

Not only the food depletion in the water column is a problem, but also birds may run into a shortage of food as an effect of increasing oyster beds. The shells of *C. gigas* are too large and much stronger than those of *M. edulis* (Blew and Südbeck, 2005; Wehrmann and Schmidt; 2005, Cadee, 2008a, b).

The negative effect of *C. gigas* on the German Blue mussel fishery has become noticeable in the meantime. As described before, the decline of *M. edulis* has not only been due to an increase in *C. gigas*, but has also been a result of climate change. The overgrowth of mussel beds by oysters results in the formation of solid calcareous reef structures, of which harvesting is unlikely to be effective and profitable. In consequence, the economic loss for Germany amounts to about 25 Million Euro per annum.

Until now, 80 alien species have been imported into the Wadden Sea (Wolff, 1992). It can be assumed that 32 of these species have been imported with *C. gigas* of which many have a negative impact on the native ecosystem (Drinkwaard, 1999; Leppäkoski et al., 2002).

Currently, no further spreading is recorded in the North Sea. But warm summers and mild winters will facilitate a further increase in oysters. Due to climate change, a spreading into more northern waters by natural processes in the future will be possible. There are records of large populations of *C. gigas* in Scandinavia whose widespread appearance in this region correlates with warming water temperatures.

Despite all negative impacts, it should be kept in mind that before the entrance of C. gigas into the Wadden Sea O. edulis has already been abundant in the North Sea and formed large oyster reefs there. Even though it’s another species, the introduction of C. gigas might lead to a (re-)formation of plentiful oyster reefs as a habitat in the Wadden Sea.

2.3.2.3.2Baltic Sea

Mnemiopsis leidyi

The lobate planktonic comb jelly *Mnemiosis leidyi* A. Agassiz, 1865 is a voracious carnivore on zooplankton and pelagic fish eggs and larvae (Purcell, 1985; Monteleone and Duguay, 1988; Purcell et al., 1994). The native habitats are the Atlantic coasts of North and South America in temperate and subtropical estuaries (Purcell et al.; 2001). *M. leidyi* tolerates a broad spectrum of salinity (<2 to 38 PSU) and temperature (0 to 32 °C). Reproduction starts at 12°C, but highest rates were found between 24 to 28°C. In the early 80s, the ctenophore was introduced unintentionally into the Black Sea with the ballast water of ships (Vinogradov et al., 1989). From there it spread (probably again in ballast water) to the Caspian Sea where it was first detected in 1999 (Ivanov et al., 2000). By the end of the 1980s, it also colonised the Sea of Azov (Vinogradov et al., 1989) and spread via the Sea of Marmara and the Aegean Sea into the Mediterranean Sea (Shiganova et al., 2001a). In 2009, *M. leidyi* was found in the western Mediterranean in the Ligurian, Tyrrhenian and Ionian Seas (Boero et al., 2009), at the Spanish coast (Fuentes et al., 2009) and the coast of Israel (Galil et al., 2009).

The first appearance of *M. leidyi* in Northern Europe was reported from the Oslofjord at the end of the year 2005 (Oliveira, 2007). Furthermore, it was found at the coast of Sweden in 2006 (Hansson, 2006), the Netherlands (Faasse and Baya, 2006) and the North Sea near Helgoland (Boersma et al., 2007). Tendal et al. (2007) reported other probable sightings since 2005. In the Baltic Sea, the first recordings were made in the Kiel Bight in 2006 (Javidpour, 2006). In the following year, the ctenophore spread to the central Baltic (Kube et al., 2007; Huwer et al., 2007) up to the Gulf of Finland, Åland and the Bothnian Seas (Lehtiniemi et al., 2007). Both invasions to the southern and northern regions of Europe probably occurred independently, since the species originated from different regions of the native habitat. Modeling results seem to support the respective genetic analyses (Reusch, 2010; Seebens pers. Comm.).

The spread and establishment of *M. leidyi* in the northern waters is now observed critically, because a comparable impact of the invader as regarded from the Black and Caspian Seas is suspected. In the Black Sea, the populations of *M. leidyi* almost exploded and the ctenophore was hold responsible for the breakdown of the fish stocks, especially of the anchovies (*Engraulis encrasicolus*), because of its intense predation on fish eggs and juvenile fish and the competition with planktivorous fish (Kideys 1994; Shiganova, 2003). Although the impact of the invader on zooplankton and fish is immense, some contexts were not considered for the first evaluations (Bilio and Niermann, 2004). Between the first record 1982 and the explosion of the population 1989 was a time lag of 7 years. Additionally, the collapse in anchovies' stocks and outbreak of *M. leidyi* happened at the same time. If direct predation would have been the cause, the massive decrease in fish catches should have been expected 1 to 2 years after the affected cohort had reached the size classes exploited by fishery (Bilio and Niermann, 2004). Furthermore, food limitation by competition had not been the case at the first outbreak (Bilio and Niermann, 2004). On the one hand, the biomass of *M. leidyi* could have been overestimated (Weisse et al., 2002), on the other hand model results (Gücü and Oguz, 1998; Gücü, 2002) as well as long-term data (Shiganova and Bulgakova, 2000) have given evidence that enough food had been available for the planktivores (Bilio and Niermann, 2004). Moreover, changes in the Black Sea ecosystem occurred even before the arrival of *M. leidyi*. Since the 1960s, inflow by rivers have brought many nutrients into the ecosystem (Shiganova and Bulgakova, 2000), and this eutrophication led to a shift in the phytoplankton communities including red tides and increase in anoxic areas (Kideys, 1994). Consequently, also biomass and composition of zooplankton changed. In addition to that, changes in fishery occurred. After it had reduced the large pelagic fishes heavily during the 1960s, fishery concentrated on the increasing small pelagic planktivores. The fishing fleets increased and more and more fish were landed by improved equipment and methods (Gücü, 2002). When *M. leidyi* arrived the fish stocks had already been overfished.

Shortly before the outbreak of *M. leidyi*, the overfishing was recognisable by the reduced size of the caught fishes. To hold the same catch level, more newly recruited small-sized fishes were landed (Gücü, 2002; Kideys, 1994) and the stock collapsed thereafter due to recruitment failure. Reduction of feeding pressure led to increasing zooplankton populations. First the native jellyfish *Aurelia aurita* profited of this increased food amount, reaching high biomasses in the 1980s. *M. leidyi* is advantageous over the Scyphozoa *A. aurita* probably because the comb jelly needs no asexually reproducing polyp stage and releases the eggs directly into the water for fertilization. As simultaneous hermaphrodite with high feeding and growth rates, it is able to proliferate rapidly (reviewed in Purcell, 2001). In contrast to that, *A. aurita* needs coastal waters in the oxygenated zone with the suitable substrate for settling with the polyp stage. As strong competitor of *A. aurita* simultaneously reducing other competitors by feeding the offspring of the planktivore fishes, *M. leidyi* profited from the released resources and led to the complete breakdown of fish stocks. Furthermore, zooplankton biomasses had been reduced and some species vanished completely (Shiganova and Bulgakova, 2000). In addition, climate conditions seem to have promoted the success of *M. leidyi* (Bilio and Nierman, 2004; Purcell, 2005). Cold winters in the 1980s probably prevented the outbreak of the invader, but simultaneously made the system vulnerable (Oguz et al., 2008). As the conditions returned to warm spring temperatures, *M. leidyi* made use of this advantage in combination with the available food resources for the massive growth. Since the first outbreak in 1989, *M. leidyi* has constituted a substantial part of the biomass in the Black Sea and had shown additional peaks in 1992 and 1995. Between the peak years, cold winters reduced the growth, resulting in lower biomasses (Oguz et al., 2008). When in 1997 the second exotic ctenophore Beroe ovata appeared as invader in the Black Sea (Konsulov and Kamburska, 1998), the biomasses of *M. leidyi* were reduced. *B. ovata* is a natural predator of *M. leidyi* and has been suggested to be introduced as biological control by the expert group GESAMP (GESAMP, 1997). However, *B. ovata* invaded independently from the Mediterranean where it was introduced with ballast water from the North Atlantic (Shiganova, 2001b). *B. ovata* had heavily predated on *M. leidyi* such that the fish stocks and zooplankton recovered thereafter (Shiganova, 2001b, Shiganova, 2003). Moreover, the reduction of fishing contributed to the recovery because the fishing fleets had either been extremely reduced or migrated to other fish grounds (Gücü, 2002).

In the Sea of Azov and the Caspian Sea, also all trophic levels of the ecosystem and the fishery had been affected by the invasion of *M. leidyi* (Shiganova,2001 a; Roohi et al., 2008; Roohi et al., 2010). In the Caspian Sea, the ecosystem had been affected more rapidly and stronger than in the Black Sea. A subsequent reduction in zooplankton had a negative effect on the commercially most important fish, the kilka (*Clupeonella* spp.), and led to shifts in the composition and biomass of phytoplankton. The benthic communities showed changes as well which had an effect on the bentho-pelagic fishes (Roohi et al., 2010).

The example of the Black Sea clearly demonstrates how the interaction of different factors can contribute to the success of an invader. By the regime shift in the ecosystem, by eutrophication and by overfishing a status had been reached where *M. leidyi* could profit from the increasing zooplankton biomass, which had been released from predation. As a strong competitor with high potential and broad environmental tolerances, *M. leidyi* could have a similar effect in the Baltic Sea in future. Eutrophication and overfishing are well known for the Baltic Sea as well (UNEP, 2005). Just one year after the first recordings of *M. leidyi*, Javidpour et al. (2009a) found evidence for a successful reproduction in the Kiel Bight. Temperature seemed to have played a central role in the successful establishment and reproduction of *M. leidyi*. In its native habitats the ctenophore survives temperatures lower than 4 °C, but in combination with higher salinities than present in parts of the Baltic Sea. In the Black Sea, the Caspian Sea and the Sea of Azov, the species cannot overwinter with low salinities (Purcell et al., 2001). However, it has spread rapidly in the Baltic Sea and overwintered in the south (Kube et al., 2007; Javidpour et al., 2009a) as well as in the north (Vitasaalo et al., 2008; Lehtiniemi et al., 2007) and has also been able to reproduce. It seems that *M. leidyi* generally stays in water layers below the halocline where temperature does not fall below 4 to 5 °C (Lehtiniemi et al., 2007; Haslob et al. 2007; Huwer at al., 2007; Storr-Paulsen and Huwer, 2008). To some extent, it reached very high abundances and showed potential of fast reproduction (Lehtiniemi et al., 2007). In their native range reproduction takes place over 12°C (Shiganova et al., 2001a) but Lehtiniemi et al. (2007) found largest densities of eggs within 4.5 to 5 °C around the halocline in the Baltic Sea. This could be evidence for adaptation of the invader to the new environment. With increasing temperatures *M. leidyi* was found also in upper water layers (Javidpour et al., 2009a). Especially important is the possible establishment of *M. leidyi* in the central Bornholm basin, the probable most important spawning ground for cod (*Gadus morhua*) and sprat (*Sprattus sprattus*) in the Baltic Sea (Köster et al., 2005; Aro, 1989). In May, Haslob et al. (2007) found *M. leidyi* in high density overlapping with the eggs of both fishes consuming them as well. Half a year later, densities had been much lower and most individuals remained at the edge of the basin (Huwer et al., 2008). However, a study with a longer time period by Schaber et al. (2011) showed that an establishment in the Bornholm basin is unlikely and that the occurrences in this area probably is a consequence of yearly transports from the Western Baltic Sea. This would be in accordance with observations in other regions, such as the Sea of Azov which is recolonized each year (Purcell 2001). The spatial as well as the temporal distribution gives evidence that the ctenophore reaches the central basin driven by water currents in autumn and winter. It further seems that the species is not able to reproduce there. Probably it is trapped in a cold-water layer above the halocline where it had encountered most frequently (Schaber et al., 2011). In contrast to most native and invaded habitats (Purcell et al., 2001; Javidpour et al., 2009a), *M. leidyi* mostly vanishes from the Bornholm basin during summer, but is abundant in November and March (Schaber et al., 2011). Responsible for the absence could be food limitation, but also predation by adult cod, which is to be found in these water layers (Schaber et al., 2011; Huwer et al., 2008).

For *M. leidyi* it is not clear yet what the food preferences and competitors are in the Baltic Sea. In contrast to Haslob et al. (2007), Hamer et al. (2008) concluded from laboratory experiments that fish eggs are not within the food range and copepods are preferred. In the same way, Mutlu (1999) demonstrated for the Black Sea that relatively low numbers of fish eggs and mainly copepods serve as food for the ctenophore. Javidpour et al. (2009a) showed that a temporal overlap existed to a greater extent with ciliates than with mesozooplankton which coincided with the occurrence of fish larvae. High numbers of larval stages gave evidence that they preferably had consumed the ciliates (Javidpour et al., 2009a). Further studies revealed a broad food spectrum of *M. leidyi* with seasonal shifts from small, slow swimming organisms as barnacle nauplii at low temperatures to actively swimming larger copepods in summer (Javidpour et al., 2009b). Possibly, the food preferences are related to the size of the individuals and their lobes that they use to catch the prey. Presumably associated with salinity, the individuals in the Baltic and the Caspian Sea (Finenko et al., 2006) are smaller than in their native range or the Black Sea, possibly reducing their efficiency in catching their prey (Javidpour et al, 2009b).

Thus, it remains still questionable which impact *M. leidyi* has on the fish stocks and on the ecological system of the Baltic Sea. Additionally, there may be further shifts by adaptation of the very tolerant invader to the ambient conditions or by altered conditions due to climate change. Especially the last may have consequences, because the temperature is one of the most important factors for *M. leidyi* to successfully establish in a new environment (Kremer, 1994). In the Baltic Sea, the invader lives at the edge of its temperature tolerance limits.

Marenzellaria spp.

There are two species of the benthic red gilled mud worm in North Sea and Baltic Sea, *M. viridis* and *M. neglecta,* which can hardly be distinguished (Sikorski and Bick, 2004). The separation was carried out by genetic analyses, which also identified the taxonomic descent (Bastrop et al., 1995, 1997, 1998). In older documents, the names *Marenzelleria wireni* or mainly *Marenzelleria viridis* had often been used. Both species have only been reliably identified since 2004 by genetic analyses or the examination of the length of nuchal organs in adult specimens.

In their native environment, both species may co-occur but prefer different salinities. For *M. viridis* a minimum of 16 is necessary, while *M. neglecta* prefers salinities from 0.5 to 10. This is the reason, why *M. viridis* and *M neglecta* disperse in the North Sea and Baltic Sea, but can cohabit in the Elbe estuary.

Marenzelleria primarily originates from the North-American Atlantic coast and is limited to the northern hemisphere (Gollasch et al., 1999; Sikorski and Bick, 2004). By detailed analyses, it had been possible to determine where *M. viridis* and *M. neglecta* originally came from. *M. viridis* derives from the coast between New Scotland (Canada) and Cape Henlopen (Delaware, USA). *M. neglecta* has his origin in the Arctic (Tuktoyaktuk, Northwest Territories, Canada), near New Hampshire and between Chesapeake Bay and Ogeechee River in Georgia.

The first evidence for the existence in Europe was found in 1979 in Forth Estuary (Scotland) (Elliot and Kingston, 1987). From there, the genus spread over the rivers Tay (Atkins et al., 1987), Humber, Ems (first German evidence, 1983), Weser and Elbe to the Ringkøbing Fjord (Essink and Kleef, 1988, 1993; Zettler, 1997). According to latest findings, there is only the species *Marenzelleria neglecta* abundant in the Baltic Sea. The first appearance in the Darß-Zingst-Bodden-Chain dated back to 1985 (Bick and Burkhardt, 1989). From there, the species spread over the eastern Baltic to the Gulf of Finland (Maximov and Panov, 2002).

Two causes are seen as possible distribution vectors. On the one hand, the introduction of ballast water seems to be a possible cause. On the other hand, the distribution by planktonic larvae (Didžiulis, 2006), which also can be transported by ballast water, could have been a source. But up to now it is not clear what the main or potentially the only vector has been. Nevertheless, the introduction of *M. viridis* (*Marenzelleria* Type I) into the North Sea and *M. neglecta* (*Marenzelleria* Typ II) into the Baltic Sea (Essink, 1999) took place independently.

Marenzelleria is a deposit feeder and buries deep channels penetrating up to 35 cm into the sediment. Therefore, this genus represents a new functional group in the northern Baltic Sea as it burrows much deeper and more actively than the native polychaetes (Didžiulis, 2006). Its digging behaviour leads to an increased water exchange between bottom water and sediment and hence to an improved oxygen circulation in the sediment (HELCOM, 1996; Didžiulis, 2006). Additionally, it leads to accelerated remineralisation and transformation processes in the sediment, whereof microalgae can profit and the chlorophyll-a concentration in the sediment becomes higher (Kotta et al., 2001).

Marenzelleria is a competitor for food and has negative impacts on abundance and growth of the ragworm *Hediste diversicolor* and the amphipods *Monoporeia affinis* and *Corophium volutator* (Kotta et al., 2001; Kotta et al., 2004; Kotta and Olafsson, 2003; Essink and Kleef, 1993; Essink et al., 2005). Many experiments come to the conclusion that there is a negative correlation between the densities of *Marenzelleria* and *H. diversicolor*. *Marenzelleria* not only reduces the survival of *H. diversicolor*, but also the survival and growth of *Marenzelleria* will increase in presence of *H. diversicolor* which also serves as additional food source after degradation. Overall, the occurrence of *H. diversicolor* will facilitate the settlement of *Marenzelleria*. In contrast, the presence of the mussel *Macoma balthica* causes a dieback of *Marenzelleria*, due to a more efficient food intake (Kotta et al., 2004), and the presence of *Marenzelleria* increases the growth of *M. balthica*. Therefore, *Marenzelleria* escapes to deeper waters where *M. balthica* is almost completely absent due to predation by amphipods, e.g. *M. affinis*. *Marenzelleria* competes for food with this amphipod and negatively influences the growth of *M. affinis*. All these interactions may explain why *Marenzelleria* successfully settles in the deeper waters of the Baltic Sea.

Furthermore, *Marenzelleria* is a food source for other benthic organisms (*Carcinus maenas*) and fish (Essink and Kleef, 1993; Didžiulis, 2006).

As there are many more polychaets with a similar mode of life still living in the North Sea, no definitely negative competition effects have yet been determined (Reise et al., 2005).

2.3.2.3.3Résumé

As the examples have shown, new species will have more or less great impacts on the structure and the function of the system. This can also yield considerable economic consequences. The effects may occur temporary and seasonally (as for *Karenia mikimotoi*), they may lead to irreversible negative changes (as for *Crassostrea gigas*), they may be neutral or partly positive (as for *Marenzellaria* spp.), or pose a potential threat, if there have already been significant changes in other marine areas under certain conditions (as for *Mnemiopsis leidyi*).

The 'Biopollution Level Index' (Olenin et al. 2010) being developed for the Baltic Sea and described in chapter 2.3.1.3 is a way of directly assessing the impact of non-indigenous species. In order to be able to set the classification at three levels (community, habitat and ecosystem), a lot of information on the assessed area and the respective ecosystem is required. Once this information is available, it will be possible to continuously work with the assessment system, being aware of course, that regular checks on the settings are mandatory. For areas where any information on the impact of non-indigenous species is missing, comprehensive investigation has to be carried out prior to the assessment. For the same species, the small-scale impacts might be assessed differently, depending on the boundary conditions. Compiling the necessary information and calculating the index require major efforts. Furthermore, the application of the 'one out – all out' principle, where the lowest classification level determines the overall assessment, leads to a significant loss of information.

Another possible system to assess the impacts is the 'black list' system (Essl et al. 2008; Nehring et al. 2010) that has been described in chapter 2.3.1.2. For allocating the non-indigenous species to differently graduated lists, information on the species impact on the system will be required as well. In this case, expert judgement can be used to assign information being available only for other regions to the region of interest and thus, classifying the species, respectively.

For assessing an ecosystem through the Descriptor non-indigenous species, it is recommended not only to consider the abundance or biomass, but also the impacts on the system or on parts of the system. As has been shown before, changes in the system can be induced by non-indigenous species. But depending on the extent of these changes, this does not necessarily imply that the good environmental status is not reached or cannot be reached in future.

The good environmental status with regard to Descriptor 2 of the MSFD will be reached if non-indigenous species do not have any negative impact on indigenous communities or natural habitats and new invasions tend against zero (BLMP 2012). The last criterion is important, but not on its own decisive. The question arises whether a system fulfilling the criterion of zero invasions over a significant time period, but at the same time being irreversibly altered or damaged by high densities of non-indigenous species from former invasions and thus, significantly deviating from the natural conditions, at all can be assessed as being in a good status. Again, this does also not mean that the good status cannot be reached in future.

An assessment system on the basis of non-indigenous species has to consider various criteria. The removal of established non-indigenous species by applying management measures is very difficult if not impossible for aquatic systems. For that reason, the mere existence of a non-indigenous species intruded a long time ago should not be a KO criterion for reaching a good environmental state. On the other hand, changes in the communities and habitats, significantly deviating from natural conditions, should not be ignored. And last but not least, the necessary collection of information and analytical data will have to be done with reasonable effort and should be integrated in the existing monitoring programmes. And a poor evaluation of the Descriptor 2 should not prevent the overall assessment from getting a better rating.

In the following chapter, a concept for an assessment system will be proposed that will fulfil the criteria laid down in the previous chapters.

2.3.3 Assessment concept for the application within the frame of the MSFD

As already described in the previous chapters, the existence of non-indigenous species in the ecosystem shall be assessed by using three types of information: the proportion of alien species, the impact of these species on the system as a whole, and the occurrence of new non-indigenous species in a specified area. This information also covers the three indicators for the Descriptor 2, as defined by the Commission (chapter 2.3.2). One characteristic of the assessment system will be that the mere existence of non-indigenous species will not be a criterion for excluding the achievement of the Good Environmental Status (GES). This is due to the fact that it will be very difficult or impossible to remove already established non-indigenous species of particular organism groups from the system by management measures. Furthermore, the data required for the assessment system should be easily and efficiently extractable from existing monitoring programmes (chapter 0) that will have to be adapted accordingly if necessary. In the following, a multi-metric concept for such an assessment system will be described.

2.3.3.1 Ratio between non-indigenous and indigenous species

In a first step, the proportion of the non-indigenous species in a system will be determined on the basis of abundance or biomass. This is in accordance with the Abundance Contamination Index ACI (chapter 2.3.1.4) described by Arbačiauskas et al. (2008). The term 'contamination' implies significant negative impacts. Since many non-indigenous species do not have major effects on the ecosystem, a more neutral term for this parameter will be recommended instead: 'Quantitative Non-Indigenous Species Index' (QNISI).

The data being a basis for calculating this index will be extracted from the established monitoring programmes for the various organism groups. An important pre-condition for this system is that the non-indigenous species are clearly and unambiguously identified by the taxonomist. This is a potential major source of error, which can be minimised by taxonomic training courses and improved ecological knowledge of the system. The taxa in the samples should be analysed down to the lowest taxonomic level if possible. This is the case for most of the German monitoring programmes. On the other hand, this detection level will not be mandatory for all species in order to be able to calculate the index. For some species in certain organism groups it will not be possible to determine down to the species level within the existing routine monitoring programmes. In these cases, the species will be assigned to a higher taxonomic level. As mentioned above, it will be important that these taxa will be separately analysed and marked with a flag, respectively.

For the calculation of the index, the proportion of the non-indigenous species in relation to the total quantity will be determined for selected samplings or surveys. Arbačiauskas et al. (2008) used the number of organisms (abundances) for the determination of the ACI. This will not be applicable to all organism groups. For example, for phytoplankton the sizes of the largest and the smallest individuals of a group may vary for several orders of magnitude. Many small organisms of a species having a low biomass will contribute much less to the turnover rates for matter and energy than fewer but larger organisms with higher biomasses. For that reason, the index for some organism groups should be determined on the basis of biomass. We will call this index the 'Quantitative Non-Indigenous Species Index on basis of Biomass' (QNISI-B). For phytoplankton, the parameter 'biovolume' can be used, which has been determined in recent routine analyses much more frequently than before. Zooplankton biomass is calculated as dry weight by default from abundances using conversion factors. The macrozoobenthos communities are determined as wet weight. For higher trophical stages such as fishes and mammals, the QNISI-A is calculated on the basis of the number of individuals (abundance). The macrophyte monitoring of individual taxa includes the degree of coverage of the analysed area. The derived proportion of non-indigenous species will be defined as 'Quantitative Non-Indigenous Species Index on basis of Coverage' (QNISI-C). Regardless of whether the index is calculated as QNISI-A, QNISI-B or QNISI-C, the parameter quantifies the proportion of non-indigenous species in relation to the respective whole community. For that reason, the index will generally be called as Quantitative Non-Indigenous Species Index (QNISI).

The Richness Contamination Index (RCI) used for assessments by Arbačiauskas et al. (2008) should not be applied here, since the species richness of a system can only be determined, if all occurring taxa can be found and be analysed down to the species level. As mentioned before, this will not be possible in the routine monitoring. Arbačiauskas et al. (2008) determined the richness on the level of orders, which will not require the same taxonomic level of details. However, it will remain questionable whether such a system can provide statistically sound results and conclusions.

At this point, it seems reasonable to repeat the definition for non-indigenous species that has been discussed in chapter 2.2.1. For the calculation of the QNISI it will be essential to decide what species shall be considered as being non-indigenous. For example, it may be doubted whether the diatom *Odontella sinensis* that was determined in the North Sea in 1908 for the first time, still will have to be regarded as non-indigenous species. In the meantime, this species has been fully established in the system. It can regularly be found occurring in low to medium biomasses, any effects on the system have not been observed.

As already mentioned, the QNISI may be applied to all organism groups. Within the frame of various guidelines such as WFD, MSFD as well as the German FFH guideline, several groups of organisms are regularly analysed over almost the whole areas of the German North Sea and Baltic Sea, yielding many valuable data sets. The respective monitoring programmes and recommendations for possible adaptations will be described in chapter 0. In order to achieve optimum assessment results, the QNISI should be determined for a defined area for all organism groups separately and then be combined as a mean value. The spatial and temporal resolution of the samplings will be dependent on the type of group, its seasonal development, and the distribution of its habitat. This will be described in detail in the chapter 'Monitoring'. The following steps will result in an overall QNISI for a defined area to be investigated:

- The QNISI will be separately calculated for each sample/survey and organism group;
- For surveys over a well-defined and continuous time period that comprises of samplings at different locations, an average QNISI will be calculated for each organism group separately and over the whole area;
- Phased samplings for a selected organism group will be averaged over a year or over another defined time period;
- From all QNISI calculated for the individual organism groups, the yearly averaged total QNISI will be determined.

For the final classification of an area, a time interval of six years should be averaged. For some specific organism groups, the variability in marine and coastal waters is very high, due to environmental impacts, to seasonal developmental stages and other factors. Furthermore, it will be possible that newly introduced or invasive species will first occur in large quantities, but later on will establish on a low level.

If data for determining the QNISI are not available for all organism groups in an area to be classified, the overall QNISI may nevertheless be calculated on the basis of existing data. In this case, it will have to be taken into account that possibly some alien species being among the not detected groups will not be included in the analysis.

For the QNISI system, a five-stage classification system will be proposed, as it has already been done in the WFD or other assessment systems for non-indigenous species and as described by Arbačiauskas et al. (2008) and Olenin et al. (2010). The five classes will be denoted as 'very good', 'good', 'moderate', 'poor' and 'bad'. As boundaries between the classes, the values from the 'Abundance Contamination Index' (ACI, see Arbačiauskas et al. 2008) will be proposed for the German North Sea and Baltic Sea (Table 2-8), resulting in five categories being not equidistant between each other. A very good condition of the system will be reached when less than 1% non-indigenous species occur, a good state means < 10%, moderate < 20%, poor < 50%, and bad conditions are met when more than 50% of the species consist of alien species. In contrast to that, only two classes exist within the MSFD: Either the 'Good Environmental Status' (GES) is 'reached', or it is 'not reached'. In order to achieve a more differentiated system that also allows predictions of trends, the five-stage system should be used. With regard to the classification of the MSFD, the GES status will be reached, if the ecosystem is classified as 'good' or 'very good'. Referring to the proposed boundary values of the QNISI system, this would mean that the GES would be reached if not more than 10% of non-indigenous species occurred in a specified area over an average time period of 6 years. For the reporting sheets of the MSFD, only the GES status will be reported.

Whether the proposed classes can be applied to all possible constellations in German water of the North Sea and Baltic Sea will have to be carefully considered by carrying out statistical analyses on comprehensive data sets from the areas of interest and for the various organism groups. Also, expert judgement will be required. It was not possible to do these analyses within the frame of the current project. These analyses could also yield valuable information on the question whether it would be sufficient to restrict the analyses to particular taxonomic groups serving as 'indicator groups'. After the final assignment of the classes, extensive application tests should be carried out.

Table 2-9: Recommended classification for the 'Quantitative Non-Indigenous Species Index' (QNISI) and the respective assignment to the Marine Strategy Framework Directive.

Classification	Quantitative Non-Indigenous Species Index (QNISI)	Classification according to Marine Strategy Framework Directive (MSFD)
very good	< 0.01	Good Environmental Status (GES)
good	0.01 - 0.1	
moderate	0.11 - 0.2	Good Environmental Status failed
poor	0.21 - 0.5	
bad	> 0.5	

An example for calculating the QNISI on the basis of real data for macrozoobenthos and phytoplankton will be found in chapter 2.3.3.5.

2.3.3.2 Estimation of impact of non-indigenous species

Not only the mere abundance of non-indigenous species is an important criterion for assessing the Good Environmental Status (GES), but also the impact of these alien species on the communities and trophic structures within the ecosystem. For that reason, a simplified gradual list system for classifying the non-indigenous species will be proposed as a second indicator system for the assessment. This system will be based on the 'Black-list' system proposed by Essl et al. (2008), which has been described in chapter 2.3.1.2. This system will be adapted to the marine habitats in the North Sea and Baltic Sea.

The non-indigenous species will be classified into three lists, according to the extent of their impact on the ecosystem structures. The species of each list will have assigned weighting factors (multiplicators) that will be combined with the respective abundance or biomass or the degree of coverage (see below):

- White list: To date, no or only very minor (to be neglected) negative impacts on communities, habitats, and the trophic nets within the ecosystem have been shown. The species will get a weighting factor of 1, which means a neutral impact. Rare cases leading to improvements of the assessment will also be assigned to this list (see below).

- Grey list: For these species, moderate negative impacts on the ecosystem are known or there are some indications in that direction. Some structural modifications of the ecosystem may occur. The related species will get a weighting factor of 3.
- Black list: For the species assigned to this list, clear evidence exists that they will cause severe changes in the ecosystem. They will get a weighting factor of 5.

The compilation of these lists will require expert judgement. The decisions for the classification of the non-indigenous species in the North Sea and Baltic Sea should be made by a commission. This should include not only the observed effects in the investigated area, but also the potential risks arising from the species being known from observations and proved evidence from other areas.

The initial classification of the lists should be carried out by an expert team on the basis of the acute effects being observed in local waters. As a start, this can lead to the situation that species being classified as highly invasive in other marine regions will not be assigned to the black list, since they have only low abundances or do not show any impacts in nativ waters. For that reason, the proposed list system will not be fixed, but can be adapted in a flexible way (see below). In order to take into account the potential threats of certain species, there are two possible ways to deal with the problem, as will be shortly explained by the example of *Mnemiopsis leidyi*. This species has already caused severe effects on the ecosystem of other marine regions (chapter 2.3.2.3.2). For several years, this species has also been abundant in the North Sea and Baltic Sea in small numbers, but has not yet shown any special impact on other populations or on the trophic interactions of the system. For that reason, this species would be possibly assigned to the grey list, but not to the black list. But considering the potential danger of this species, it could alternatively be assigned to the next list (i.e. the black one) or get a higher multiplication factor within the (grey) list. Assigning *Mnemiopsis leidyi* to the black list would result in a weighting factor of 5 instead of 3 and the assessment index would increase, by this requiring stronger management measures in order to reach the Good Environmental Status (GES). A second alternative for dealing with the potential threat of *Mnemiopsis leidyi* would be to add this species to the grey list and assign a weighting factor of 4 instead of 3. At current stage, the latter alternative would be the most recommended solution. Consequently, the assignment to a particular list would then reflect the actual situation in the local waters. However, invasive species with a risk potential for the ecosystem would receive a higher weighting factor within this list, which increases the assessment index and requires more intense management with regard to this species. This system would allow a better fine tuning of the assignment. However, the list system is flexible and depending on the future development, the species could also be assigned to the black list at a later stage if considered being necessary.

The impact of a particular species on the ecosystem and the respective assignment to one of the three lists can change with time. An example is a well-known phytoplankton species, the diatom *Coscinodiscus wailesii*. This species has been introduced from the Pacific Ocean via the Pacific Oyster and was registered for the first time in European waters in the late 1970s (Boalch and Harbour 1977; Rincé and Plaumier 1986; Robinson et al. 1980). Since then the species has fast spread around the North and Baltic Sea (Rick and Dürselen 1995; Dürselen and Rick 1999). At the beginning, the species formed large stocks at the North Sea coast throughout the year and showed some negative effects on the ecosystem, such as formation of mucilage, oxygen deficits as a consequence of high biomasses, impacts on predators due to large cell sizes etc. This would have justified an assignment to the black list. Today the situation is different and the species has established in the coastal waters of the North Sea and Baltic Sea. In the German Bigh the species is existent at all times, but only in low to moderate biomasses. Neither very high abundances nor negative impacts on the ecosystem have been observed during the past years. The establishment of this species has also not changed the structure of the communities. For that reason, *Coscinodiscus wailesii* would have been assigned to the grey list over time, and nowadays would certainly be included into the white list.

On the other hand, a new species occurring with low abundance and having no impact at the beginning can become invasive over time and cause severe damage to the ecosystem. Such a species would have to be upgraded in the list system (see also the example of *Mnemiopsis leidyi* above). As a consequence, new non-indigenous species found during a six-year period will be temporarily assigned to one of the lists. The lists should be separately generated for the North Sea and Baltic Sea by experts. For each species the migration path should be identified and quantified as well as recommendations for respective management measures would be given.

In rare cases, an invasive species can exert a positive effect on the ecosystem structure. As described above, such species can then be assigned to the white list. It has to be taken into account that the assessment should include all possible effects of a species on the ecosystem. If there are only some positive aspects, but the overall effect of the species is negative, then the species will be either way classified in the grey or black list. If the overall effects are rather neutral or only slightly negative, then the species will be assigned to the white list. Non-indigenous species that show overall positive effects on other populations, habitat structures or trophic relations will also be added to the white list and receive a weighting factor of -0.5. Preliminary, the factor is not chosen to be -1 which is considered to be too high for the positive impacts of non-indigenous species. However, for the calculation of the respective impact index (INISI, see below), the weighted proportion of individuals, the biomass or the degree of coverage will remain unchanged as if the factor would be 1 as for species without remarkable effects. But the index for the taxon will be multiplied by the negative weighting factor. As a consequence, the INISI of the selected species will become negative and thus, reduce the overall INISI, due to the positive impact of this species.

The described list system for assessing the impact of non-indigenous species on the ecosystem is a simplified scheme to include the effects as required by the indicator 2.2.2. The approved lists can be used over a longer time period without having the necessity to continuously monitor the species in the investigated area. This would be different if the 'Biopollution Level Index' (BPL) (Olenin et al. 2010) was used, as described and discussed in chapter 2.3.1.3. In this case, an evaluation of the impacts on other populations, on habitats and on ecosystem structures would have to be carried out promptly and separately for well-defined areas. This approach would certainly be the best way to assess the impact on non-indigenous species; on the other hand, it will be not possible to meet the respective requirements within the routine monitoring. The proposed classification of the list system described here will have to be evaluated in a six-year interval by an expert team and - if required - to be adapted accordingly on the basis of analyses, observations, and other research results from the preceding years. If suddenly major impacts on the ecosystem are observed, then it will be also possible to shift species within the lists within shorter time intervals, provided that this is agreed on by the experts. It will be recommended to install separate list systems for the North Sea and Baltic Sea in order to take into account the aspects being specific for the individual marine systems.

The concrete compilation of the proposed three lists was not part of this project. This has to be conducted in a separate step by an expert group which members are specialists for the different organism groups. The following examples are only temporary classifications that could be revised later by the expert group.

In the following, we will describe how the species will be classified on the basis of real analyses. The main objective is to estimate the environmental impact and the calculation of a respective index.

The actual impact a non-indigenous species has got on the ecosystem is strongly related to the abundance (or to the biomass, or to the degree of coverage) of this species. Low numbers of individuals of a species considered as dangerous (black list) will not have measurable effects on other populations, communities, habitats, and trophic interactions. For that reason, the weighting factors for each non-indigenous species from the three lists introduced above will have to be multiplied by the proportion of abundance (or by the biomass, or by the degree of coverage) of the species. The portion of the indigenous species will be multiplied by 1. Afterwards, the results will be normalised to the total sum of a 100%. For non-indigenous species from the white list that have no impact on the ecosystem, the weighting factor will be neutral (1). Since there will be no effects, the individual Impact Non-Indigenous Species Index (INISI) will be set to zero for these species. By multiplying the respective weighting factors by the proportion of real biomasses, the importance of the species from the grey or black list will be more or less enhanced, depending on their hazardousness and according to their relative share of the total amount. This will result in a specific INISI for each non-indigenous species within a sampling. Finally, the total INISI for the sampling will be calculated as the sum of all non-indigenous species.

Since the INISI can be determined for all surveys for which the QNISI can be calculated, the same conditions hold as described for the QNISI. Based on existing monitoring programmes, the analysis can be carried out for all organism groups and be compiled to a total index:

- The INISI will be calculated separately for each sampling/survey and organism group;
- For surveys over a well-defined and continuous time period that comprises of samplings at different locations, an average INISI will be calculated for each organism group separately and over the whole area;
- Phased samplings for a selected organism group will be averaged over a year or over another defined time period;
- From all INISI calculated for the individual organism groups, the yearly averaged total INISI will be determined.

As discussed above, an average of a six-year period should be the basis for the classification of an area. If sufficient data are not available to calculate the INISI for all organism groups in an area to be classified, the overall INISI may nevertheless be calculated on the basis of existing data. In this case, it will have to be taken into account that possibly some alien species being among the not detected groups will not be included in the analysis.

Also, for the INISI system, a five-stage classification system will be proposed (Table 2-10). With regard to the classification of the MSFD, the GES status will be reached, if the ecosystem is classified as ‘good’ or ‘very good’. For the reporting sheets of the MSFD, only the GES status will be reported.

By multiplying the respective weighting factors of the lists by the proportion of real and existing biomasses, a newly weighting will be calculated that underlines the importance of the invasive species. The determination of the INISI as the percentage of non-indigenous species with reference to this weighting will allow values between 0 and 1 only. Since the calculation of the INISI is carried out by analogy with the QNISI by using weighted abundance, biomass or degree of coverage, the same values for the boundaries between the five classes will be recommended. Whether these boundary values can be effectively applied to the German waters of the North Sea and Baltic Sea will have to be checked by comprehensive statistical analyses of data, field tests and by expert judgement. That has not been part of this project.

Table 2-10: Recommended classification for the 'Impact Non-Indigenous Species Index' (INISI) and the respective assignment to the Marine Strategy Framework Directive.

Classification	Impact Non-Indigenous Species Index (INISI)	Classification according to Marine Strategy Framework Directive (MSFD)
very good	< 0.01	Good Environmental Status (GES)
good	0.01 - 0.1	
moderate	0.11 - 0.2	Good Environmental Status failed
poor	0.21 - 0.5	
bad	> 0.5	

Even if one or more species from the black list are found during the surveys, the Good Environmental Status (GES) may nevertheless be reached, as long as the percentage of these species with reference to the total biomass is low. This is in good accordance with field observations. The mere occurrence of invasive species in low abundances will have no or only a low impact on the biodiversity as well as on the mass and energy fluxes in the ecosystem. On the other hand, it will not be possible to classify a system as good where a third of all species consist of non-indigenous species from the black list.

An example for calculating the INISI on the basis of real data for macrozoobenthos and phytoplankton will be found in chapter 2.3.3.5.

2.3.3.3 Calculation of Non-Indigenous Species Index (NISI)

In order to achieve an overall assessment with regard to descriptor 2 ('Occurrence of non-indigenous species') of the MSFD, an average value of QNISI and INISI will be calculated. This value will be described as 'Non-Indigenous Species Index' (NISI). By using the arithmetic mean value, both QNISI and INISI will have the same weighting. That means that the abundances of non-indigenous species in an ecosystem will have the same importance as the impact of these species (and of the respective abundances) on the ecosystem. If the potential impact on the ecosystem is considered being more important, then the INISI could get a higher weighting for the calculation of the NISI. This would mean that existing invasive species from the grey and black list would downgrade the NISI while species of the white list would improve the index value. Weighting factors of 2 and 3 would then cause only minor changes in the assessment compared to the usage of the arithmetic mean value.

As already described for the QNISI and INISI, a five-stage classification system as well as the same boundary values between the classes shall also be applied to the NISI (Table 2-11). The boundary values will have to be approved by statistical analyses and expert judgement beyond this project. Field tests will then have to show the practicability. The assessment period shall cover six years.

Table 2-11: Recommended classification for the 'Non-Indigenous Species Index' (NISI) and the respective assignment to the Marine Strategy Framework Directive.

Classification	Non-Indigenous Species Index (NISI)	Classification according to Marine Strategy Framework Directive (MSFD)
very good	< 0.01	Good Environmental Status (GES)
good	0.01 - 0.1	
moderate	0.11 - 0.2	Good Environmental Status failed
poor	0.21 - 0.5	
bad	> 0.5	

2.3.3.4 Arrival of new non-indigenous species (trend indicator)

HELCOM is carrying out the assessment analysis of the descriptor 2 ('Occurrence of non-indigenous species') as a key indicator for newly arriving species within a fixed six-year period. This so-called trend indicator has already been described in chapter 2.3.1.1.

The trend indicator is suitable for evaluating and assessing management measures. But in order to obtain always an actual assessment for the current year, the analyses should rather be calculated as a floating mean value over the past six years and not be conducted in a fixed six-year period - as proposed by HELCOM.

Following the current HELOM procedures, the Good Environmental Status (GES) will be reached if no new non-indigenous species are detected in the assessment area over a fixed period of six years. This can indeed be a measure for the GES and indicate effective management measures. But it could be the case that an invasive species had arrived and established before the assessment period has started, and then suddenly this species has shown a negative impact on the ecosystem. This would well be considered by the NISI system, but not by the HELCOM assessment.

For the assessment of the descriptor 2, it will be recommended to determine also the trend indicator in addition to the calculation of the NISI. If the assessment of descriptor 2 required just one parameter, then the trend indicator could function as a bonus/malus system for the NISI. For example, the NISI could be upgraded one class within the five-stage system, if no or only a few new non-invasive species assigned to the white list had arrived within the past six years. A downgrade of one class would be conducted if a total of six new or two to three invasive species from the black list were recorded within the same period. The exact values would have to be defined by an expert team.

Alternatively, two indicators for the assessment of descriptor 2 can be used: the first one for indicating the status (NISI), and the second one for evaluating the management measures (trend indicator).

2.3.3.5 Exemplary calculation of NISI

Within this chapter, exemplary calculations of the QNISI and the INISI as well as of the NISI based on real macrozoobenthos and phytoplankton data will be made, following the method described above.

The macrozoobenthos-samplings were carried out in spring time 2012 in the inner coastal waters of Mecklenburg-Vorpommern. Each investigated area consisted of 10 stations. For three of these areas, the results of the QNISI and INISI for the individual stations are shown in Table 2-12.

Table 2-12: Exemplary calculations of QNISI, INISI and NISI for macrozoobenthos monitoring data from inner Mecklenburg-Vorpommern coastal waters during spring 2012.

	Achterwasser		Salzhaff at Rerik		Wohlenberger Wiek	
	QNISI-B	INISI	QNISI-B	INISI	QNISI-B	INISI
1	-	-	0,00	0,00	0,00	0,00
2	0,00	0,00	0,02	0,10	0,00	0,00
3	0,19	0,54	0,05	0,22	0,01	0,05
4	0,09	0,34	0,02	0,11	0,00	0,00
5	0,27	0,65	0,01	0,04	0,00	0,00
6	0,27	0,65	0,01	0,04	0,00	0,00
7	0,23	0,60	0,00	0,00	0,00	0,00
8	0,98	1,00	0,00	0,00	0,00	0,00
9	0,37	0,74	0,02	0,09	0,00	0,00
10	0,22	0,58	0,00	0,00	0,00	0,00
Mean (normalised)	3,39	4,28	1,17	1,68	0,59	0,64
NISI (normalised)	3,84		1,43		0,62	

The calculation of the QNISI has been done as QNISI-B on the basis of biomass data (g·m^{-2}). In the investigated areas, the following four non-indigenous species have been found: *Potamopyrgus antipodarum*, *Dreissena polymorpha*, *Marenzelleria neglecta* and *Gammarus tigrinus.* For the calculation of the INISI, a temporary classification of these species into one of the lists has been made in order to assign a respective weighting factor. In the table, significant differences are to be seen: Within the area ‘Achterwasser’, greater amounts of non-indigenous species (unevenly distributed) with significant impacts on the ecosystem have been found. As a consequence, the GES has not been reached with regard to descriptor 2 (following normalised averaging). This has been in contrast to the other two areas shown here.

At the continuous monitoring station on the island of Norderney, weekly phytoplankton samples are taken at high and low tide by the ‘Niedersächsischer Landesbetrieb für Wasserwirtschaft, Küsten- und Naturschutz’ (Lower Saxony Water Management, Coastal Defence and Nature Conservation Agency, NLWKN). On the basis of the respective biomass data, the QNISI-B and the INISI as well as the NISI have been calculated for the year 2011 at high tide. The values and the class boundaries are shown in Table 2-13. With *Coscinodiscus wailesii*, *Odontella sinensis*, *Mediopyxis helysia* and *Prorocentrum triestinum* four non-indigenous species have been found. Here again, these species have temporarily been classified into the three lists and respective weighting factors been assigned. Before averaging the complete data of a year, the individual results from the non-equidistant classes have been normalised. Following the resulting NISI assessment, the Good Environmental Status (GES) with regard to the MSFD has not been reached in this example. If *Odontella sinensis* is not considered as non-indigenous any longer – as proposed in chapter 2.2.1 – the normalised NISI will increase to

2,52 and hence, only slightly exceed the limit for the class boundary for reaching the good environmental status.

Table 2-13: Exemplary calculation of QNISI, INISI and NISI for phytoplankton monitoring data from Norderney during high tide in 2011.

Datum	QNISI-B	IN-ISI	Datum	QNISI-B	IN-ISI	Datum	QNISI-B	IN-ISI	Datum	QNISI-B	IN-ISI
04.01.	0,21	0,44	07.04.	0,47	0,72	04.07.	0,00	0,00	07.10.	0,37	0,04
10.01.	0,13	0,31	14.04.	0,44	0,71	11.07.	0,06	0,16	11.10.	0,54	0,00
17.01.	0,14	0,34	20.04.	0,76	0,88	18.07.	0,26	0,08	19.10.	0,50	0,00
24.01.	0,15	0,34	28.04.	0,22	0,46	27.07.	0,03	0,08	24.10.	0,00	0,00
31.01.	0,29	0,56	05.05.	0,25	0,50	02.08.	0,01	0,04	31.10.	0,51	0,09
08.02.	0,39	0,66	09.05.	0,02	0,05	10.08.	0,02	0,01	08.11.	0,07	0,18
15.02.	0,44	0,70	19.05.	0,00	0,00	16.08.	0,00	0,00	15.11.	0,00	0,00
22.02.	0,22	0,46	24.05.	0,04	0,10	25.08.	0,32	0,00	23.11.	0,00	0,00
01.03.	0,14	0,32	30.05.	0,39	0,11	31.08.	0,41	0,00	30.11.	0,47	0,00
08.03.	0,27	0,53	06.06.	0,00	0,01	07.09.	0,31	0,00	07.12.	0,34	0,61
17.03.	0,68	0,22	17.06.	0,04	0,04	13.09.	0,16	0,00	15.12.	0,38	0,65
24.03.	0,25	0,50	21.06.	0,00	0,01	19.09.	0,28	0,00	22.12.	0,71	0,59
31.03.	0,46	0,72	28.06.	0,01	0,00	28.09.	0,83	0,00	29.12.	0,23	0,47
Mean QNISI-B (normalised) = 3,07						Mean INISI (normalised) = 2,64					
NISI (normalised**)** = 3,09											

2.4 Monitoring

It is required to invent a monitoring that captures all parameters needed for the assessment. For that the existing monitoring should be included and if necessary adjusted. In the following part the monitoring and time series currently in operation are described.

2.4.1 Current monitoring programs

2.4.1.1 Phytoplankton

2.4.1.1.1North Sea

Several agencies provide the data for the regular phytoplankton monitoring of the German part of the North Sea. Location of all stations can be found in Figure 2-3, which also shows the classification of water areas according to the WFD. Normally within the sampling of phytoplankton several additional parameters like chlorophyll, nutrients etc. are measured.

The NLWKN (Niedersächsischer Landesbetrieb für Wasserwirtschaft, Küsten- und Naturschutz, Lower Saxony Water Management, Coastal Defence and Nature Conservation Agency) is currently sampling weekly at high tide and low tide at Norderney, biweekly at Wilhelmshaven and monthly at several stations at the East Frisian coast and some transitional water locations during the growing season. Samples are analysed concerning taxonomic species composition, abundances and biovolume of the phytoplankton. An additional biweekly monitoring program at 10 stations along the coast of Lower-Saxony with special regard to harmful and bloom-forming algae is carried out during summer.

The LLUR (Landesamt für Landwirtschaft, Umwelt und ländliche Räume Schleswig-Holstein, State Agency for Agriculture, Environment and Rural Areas) samples biweekly at five stations during the vegetation period (approximately from April to October) along the North Frisian coast. Samples are analysed concerning taxonomic species composition, abundances and biovolume. Additionally, the algae early detecting system (Algenfrüherkennungssystem, AlgFES) is carried out. Therefore bloom-forming and (potentially) toxic algae are monitored at 15 stations in the North Frisian Wadden Sea and the associated open sea of the German bight from May to September.

The best-known phytoplankton long-term time series is the one of Helgoland Reede which runs since 1962 conducted by the AWI (Alfred-Wegener-Institute for Polar and Marine Research). Several physical, chemical and biological parameters are determined every working day, including taxonomic species composition, abundances and biovolume of the phytoplankton. Additionally, monthly three transects (from Helgoland to the Eider, to the Elbe and to the Northwest) with 6 stations each are sampled. At the AWI Wadden Sea station at Sylt phytoplankton samples are taken weekly and taxonomic species composition, abundances and biovolume are determined.

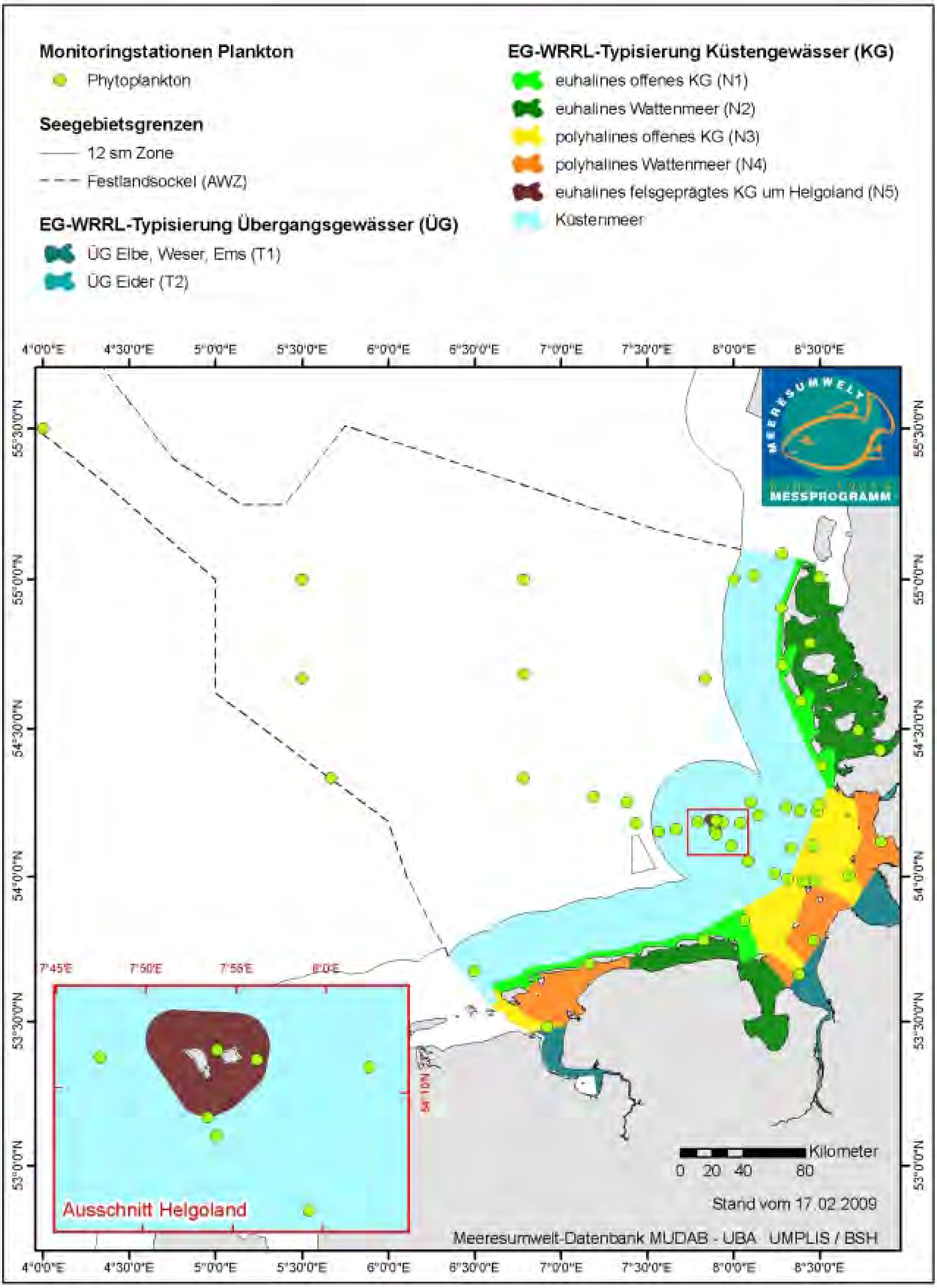

Figure 2-3: Phytoplankton monitoring stations in the German part of the North Sea (BLMP, 2010).

From 2008 to 2011 the IOW (Institut für Ostseeforschung Warnemünde, Leibniz Institute for Baltic Research) on behalf of the BSH (Bundesamt für Seeschifffahrt und Hydrographie; Federal Maritime and Hydrographic Agency) conducted for some years five excursions yearly within the German exclusive economic area (AWZ) of the North Sea. 12 stations were sampled and taxonomic species composition, abundances and biovolume of the phytoplankton were determined.

2.4.1.1.2Baltic Sea

In the Baltic Sea the monitoring data for phytoplankton are collected by LLUR, LUNG (Landesamt für Umwelt, Naturschutz und Geologie Mecklenburg-Vorpommern) and IOW (Figure 2-4).

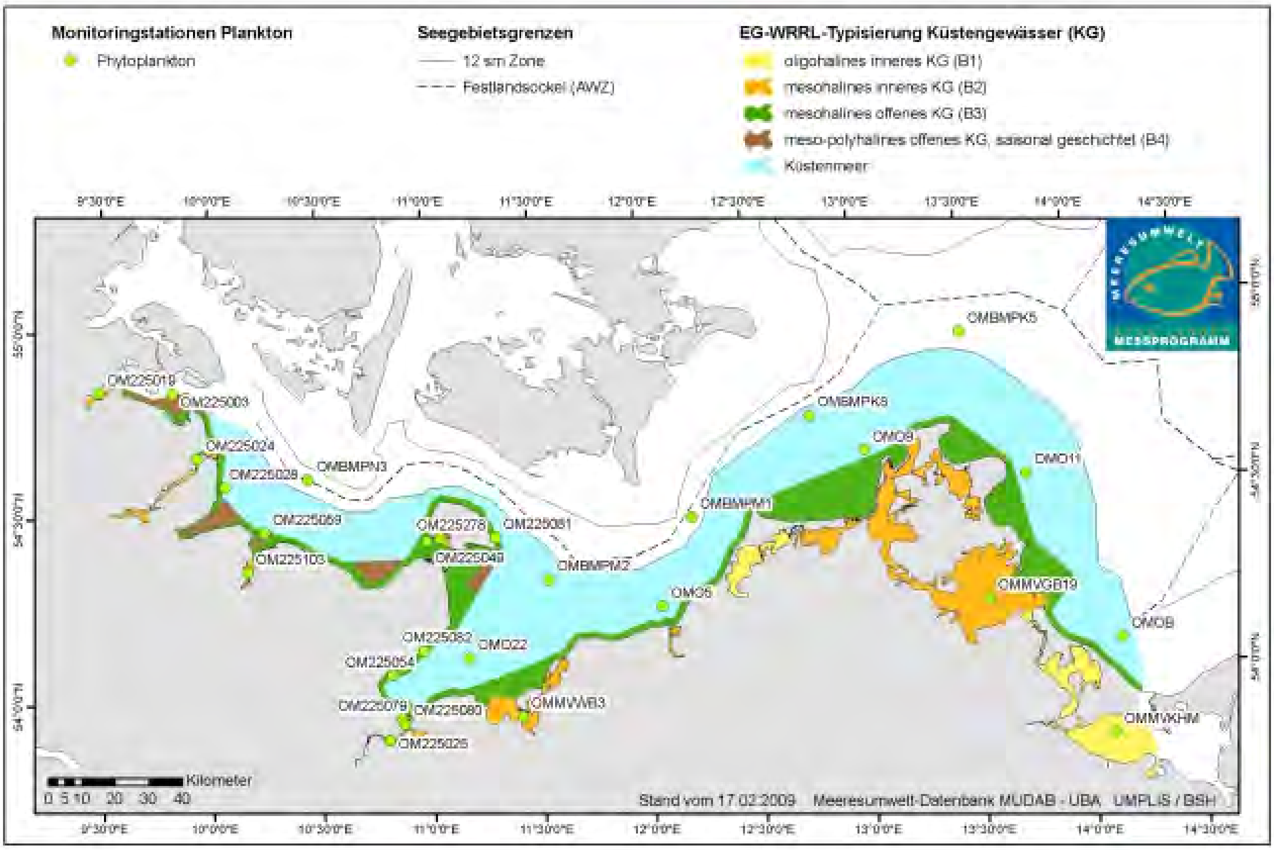

Figure 2-4: Phytoplankton monitoring stations in the German part of the Baltic Sea (BLMP, 2010).

The sampling of LLUR is carried out irregularly at different stations if required in order to build up the variability of seasonal phytoplankton succession.

The LUNG is sampling 8 stations regularly 7 times a year from the sublitoral at 1 m depth. The analyses include the taxonomic species composition and the calculation of abundance and biovolume. Additionally, Chlorophyll a is determined.

The open sea sampling within the German AWZ is conducted by the IOW on behalf of the BSH within the HELCOM monitoring. This is done by 5 excursions per year.

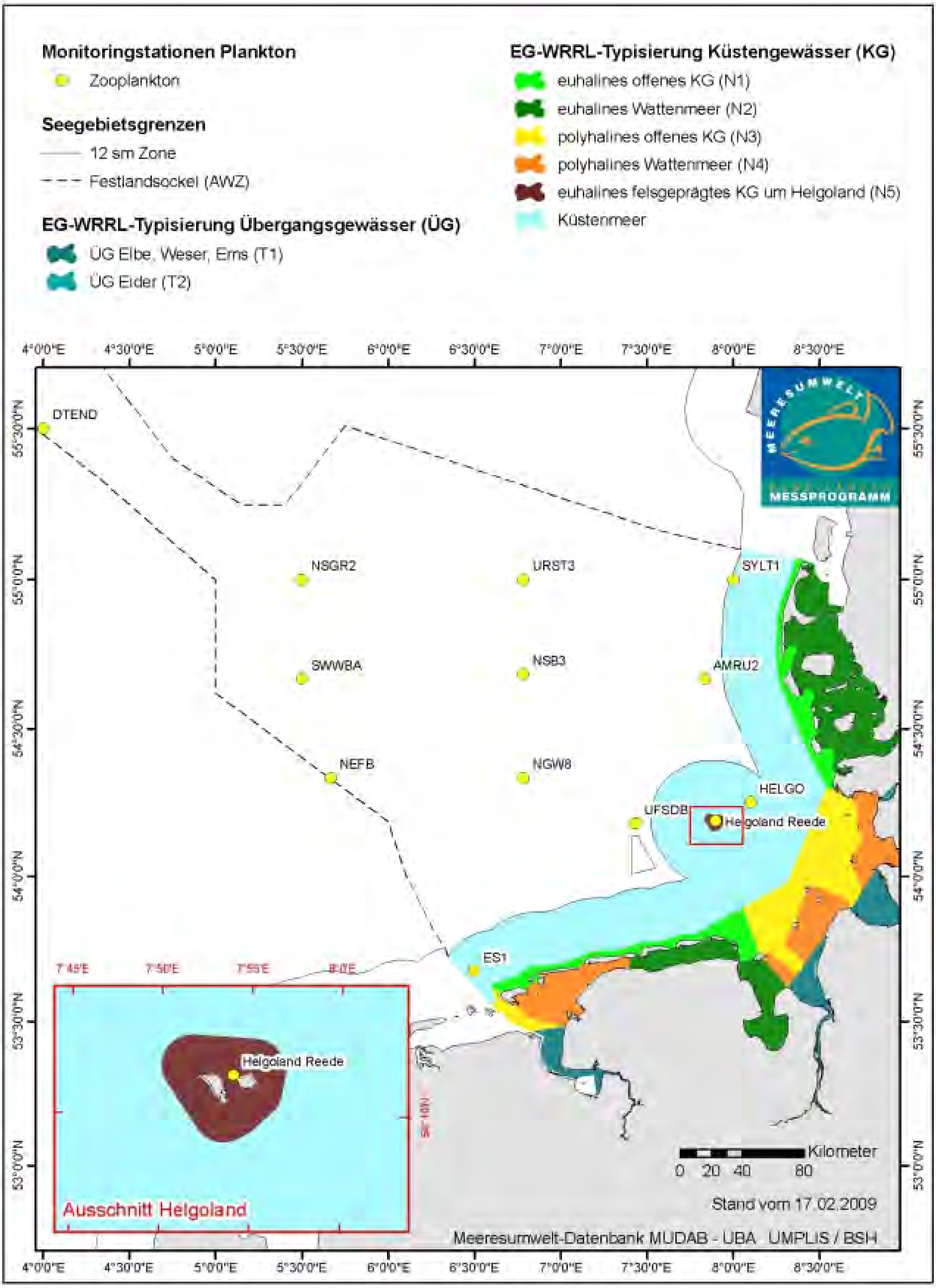

Figure 2-5: Zooplankton monitoring stations in the German part of the North Sea (BLMP, 2010).

2.4.1.2 Zooplankton

2.4.1.2.1North Sea

A regular and comprehensive zooplankton monitoring is not yet been carried out so far, because this group of organisms is not considered within the WFD. Location of monitoring stations can be found in Figure 2-5.

A zooplankton long-term time series is conducted by the AWI at Helgoland Reede since 1973. Samples are taken three times a week and taxonomic species composition, abundances and biomasses are determined. At the AWI Wadden Sea station at Sylt zooplankton samples are analysed weekly on taxonomic species composition, abundances and biomasses.

From 2008 to 2011 the IOW (Institut für Ostseeforschung Warnemünde, Leibniz Institute for Baltic Research) on behalf of the BSH conducted for some years five excursions yearly within the German exclusive economic area (AWZ) of the North Sea. 12 stations were sampled and taxonomic species composition, abundances and biomasses of the zooplankton were determined.

2.4.1.2.2Baltic Sea

Within the COMBINE-Program of HELCOM zooplankton is a Core-Variable in the Baltic Sea and measured 5 times a year at 6 stations in the German AWZ by the IOW. Additionally, HELCOM recommends to measure with a higher frequency of minimum 12 times a year but weekly during the growing season (BLMP, 2010). The analyses include the identification of the species and the calculation of abundances and biomasses.

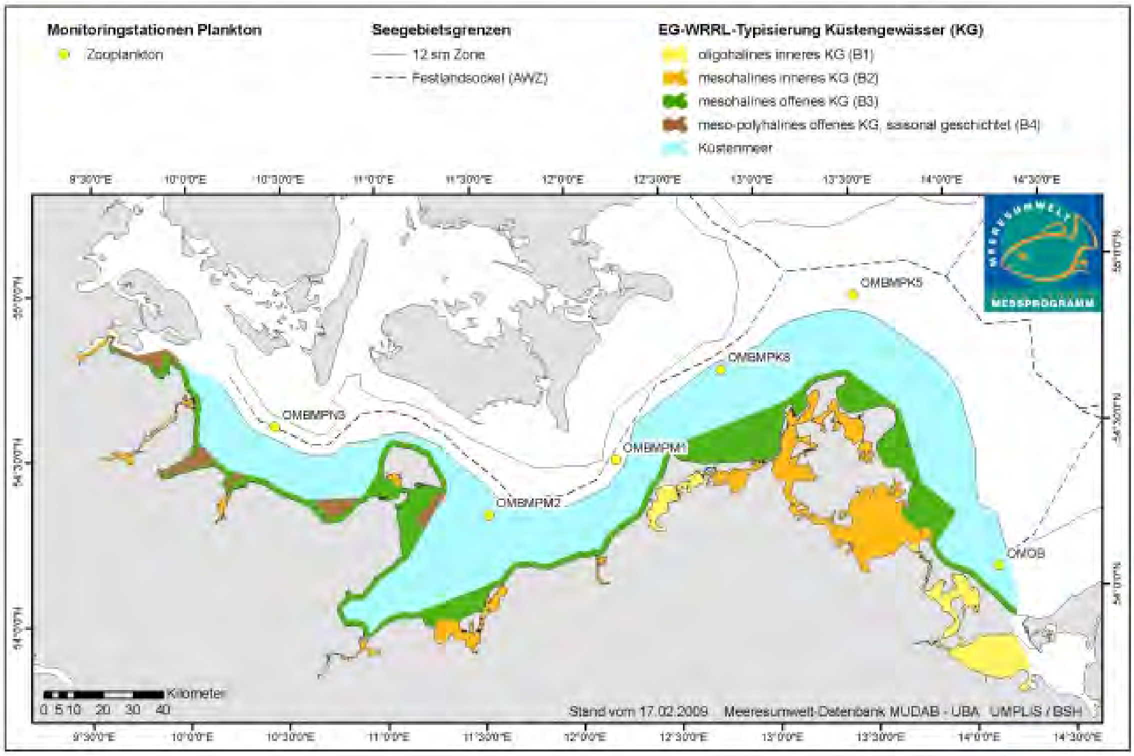

Figure 2-6: Zooplankton monitoring stations in the German part of the Baltic Sea (BLMP, 2010).

2.4.1.3 Macrophytes

2.4.1.3.1 North Sea

Two agencies (NLWKN and LLUR) provide data for the macrophyte monitoring of the German part of the North Sea. The monitoring is executed either by the agencies themselves, scientific institutes (AWI) or private consultants.

The monitoring program is presently based on the requirements of the WFD (Water Framework Directive) but fulfills also the requirements of TMAP (Trilateral Monitoring and Assessment Program) and the Habitat Directive. Several assessment systems for macrophytes have been developed focusing on different macrophyte subcomponents and spatial zones in different federal states:

- Eelgrass and opportunistic green algae in the eulittoral zone of the Wadden Sea (= N2 and N4 water type) in Lower Saxony and Schleswig-Holstein. Assessment parameters are spatial extent, density and species composition of eelgrass (Dolch et al. 2009, Jaklin et al. 2007, Kolbe 2006)
- Macroalgae in the eulittoral and sublittoral zone of Helgoland (= N5 water type) in Schleswig-Holstein. Assessment parameters are depth limits of specific macroalgae species, spatial extent and density of green algae and *Fucus* spp. and a reduced species list (Kuhlenkamp and Bartsch, 2008)
- Reed beds, brackish and salt marshes in the eulittoral (coastal) zone of the Wadden Sea (= N2 and N4 water type) in Lower Saxony only and in the transitional zone of Elbe, Weser, Ems (= T1 water type) in Lower Saxony, Hamburg and Schleswig-Holstein). Assessment parameters in the coastal zone are spatial extent and vegetation zonation, in the transitional zone spatial area, area of near-natural biotope types, width of reed and species and structure of the reeds (Adolph et al. 2007; Stiller, 2005; Stiller, 2008).

The monitoring procedures are partly standardized in a national SOP (Standard Operational Procedure, BLMP 2008) and described in several guidelines for each of the WFD assessment systems. In Figure 2-7 the monitoring sites and the water type classification (N1-N5 types) relevant for the WFD status assessment are illustrated.

NLWKN and NLPV (Nationalparkverwaltung Niedersächsisches Wattenmeer): Eelgrass and opportunistic green algae are assessed with aerial surveys in combination with field mapping and sampling at selected sites. The aerial survey covers the total area every 6 years, selected sites are surveyed annually. Reed beds, brackish and salt marshes have been assessed in 1988-1992 and 2004-2008 in 8 water bodies of coastal and transitional waters.

LLUR and LKN (Landesbetrieb für Küstenschutz, Nationalpark und Meeresschutz Schleswig-Holstein): Eelgrass and opportunistic green algae are assessed with aerial surveys in combination with field mapping and a detailed sampling at selected sites. Aerial surveys are conducted three times per year (June, July, and August), the whole area is mapped in the field once every six years and selected sites are surveyed annually. Macroalgae at Helgoland are investigated annually in the eulittoral since 2004/2005; in the sublittoral depth limits are assessed every three years since 2007. All surveys are conducted in summer.

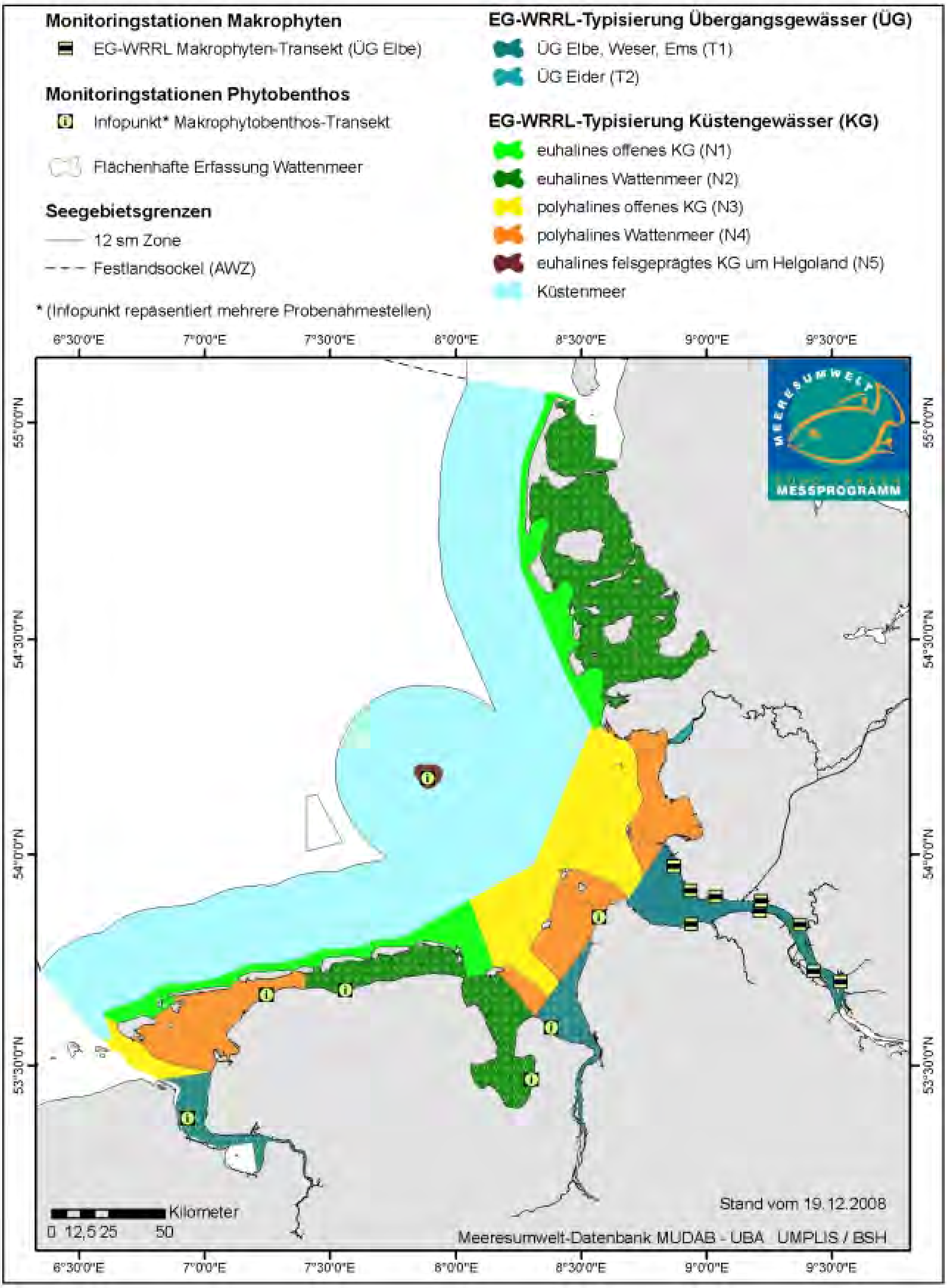

Figure 2-7: Macrophyte monitoring in the German part of the North Sea (BLMP, 2010). Eelgrass and opportunistic green algae are surveyed in the whole eulitoral area (dark green and orange areas).

2.4.1.3.2Baltic Sea

Two agencies (LLUR and LUNG) provide data for the macrophyte monitoring of the German part of the Baltic Sea. The monitoring is executed by private consultants.

The monitoring program is presently exclusively based on the requirements of the WFD (Water Framework Directive) and their valid assessment systems for German Baltic macrophytes:

- The ELBO system (Schubert et al., 2003; Selig et al., 2009) is valid for inner coastal waters (B1, B2 types) and evaluates the status of soft bottom macrophytes (higher plants and charophytes). Assessment parameters are depth limits of charophytes and higher plants as well as the assessment of type specific plant communities (by coverage estimations along the depth gradient).
- The BALCOSIS system (Schories et al., 2006, Fürhaupter et al., 2007) is valid for the outer coastal waters (B3 type) and evaluates the status of soft (eelgrass) and hard bottom macrophytes (macroalgae). Overall seven parameters are assessed (depth limits of *Zostera marina* and *Fucus* spp., biomass proportions of opportunistic algae in eelgrass beds and red algae meadows, dominance of *Fucus* spp., biomass of *Furcellaria lumbricalis* and species reduction).

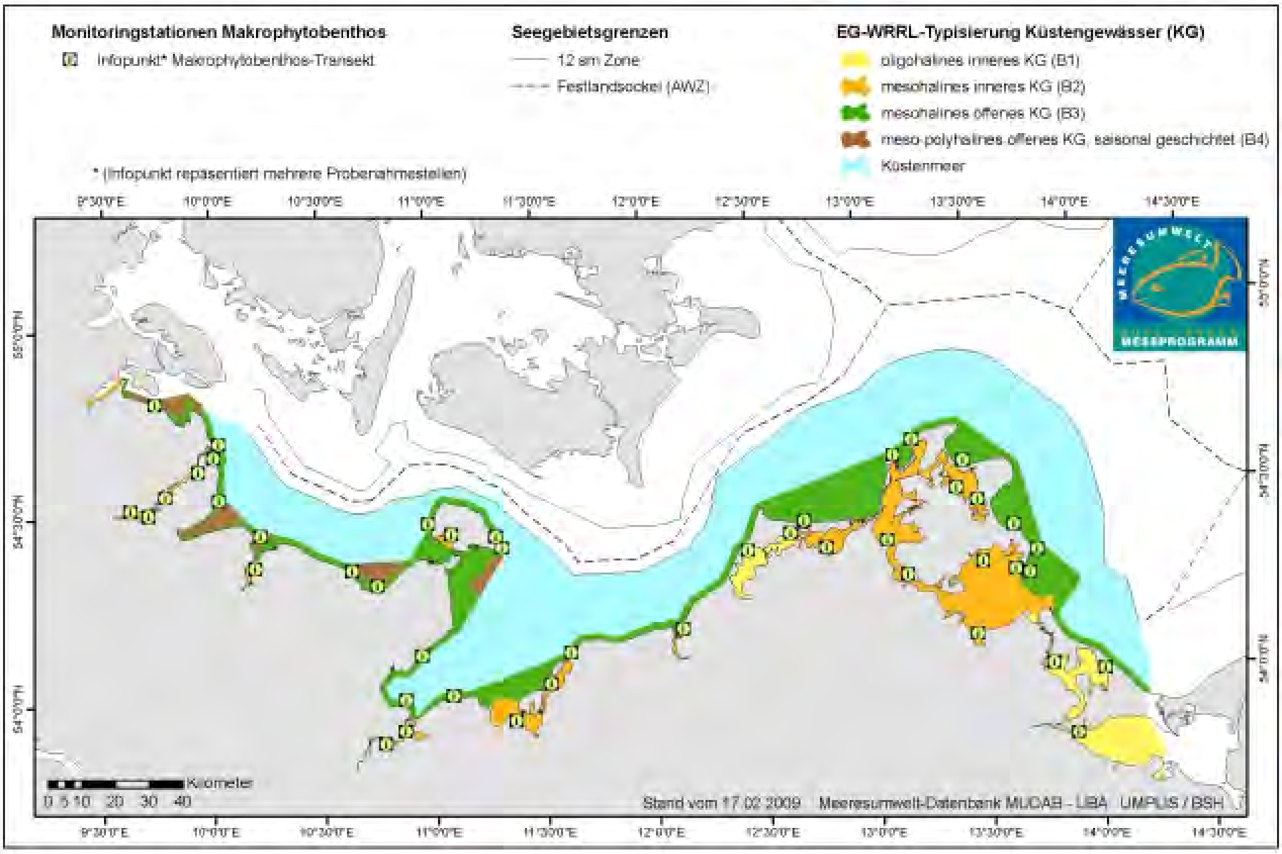

Figure 2-8: Macrophyte monitoring in the German part of the Baltic Sea (BLMP, 2010).

The monitoring procedure is standardized in a national SOP (Standard Operational Procedure, BLMP, 2008) and described in guidelines for each WFD assessment systems (ELBO: Selig et al., 2009, BALCOSIS: Fürhaupter et al., 2009). In Figure 2-8 the monitoring sites and the water type classification (B1–B4 types) relevant for the WFD status assessment are illustrated. Those water types are subdivided into geomorphological similar water bodies, which form the spatial unit for the assessment program.

LLUR: Monitoring for ELBO started in 2005. Sampling is carried out yearly at the beginning of the vegetation period (mid June to mid July) in eight water bodies. The BALCOSIS system is assessed since 2006 in eleven water bodies. Sampling is carried out yearly in the main vegetation period (July to August). Before the WFD monitoring started, macrophytes were monitored at eight transects (1995–2004) based on the HELCOM requirements for phytobenthos sampling.

LUNG: The ELBO system is assessed since 2004 in 13 water bodies and the BALCOSIS system is assessed since 2006 in four water bodies. The timing of sampling is identical to the description given above. Before the WFD monitoring started, macrophytes were monitored at six transects (1995–2003) based on the HELCOM requirements for phytobenthos sampling.

2.4.1.4 Macrozoobenthos

2.4.1.4.1 North Sea

Three agencies (NLWKN, LLUR and BfN - Federal Agency for Nature Conservation) provide data for the macrozoobenthos monitoring of the German part of the North Sea. The monitoring is executed either by the agencies themselves, scientific institutes (AWI) or private consultants.

The monitoring program is presently based on the requirements of the WFD (Water Framework Directive), TMAP (Trilateral Monitoring and Assessment Programme), OSPAR (Oslo Paris Commission), HD (Habitat Directive) or MSFD (Marine Strategy Framework Directive).

The general monitoring procedure is standardized in a national SOP (Standard Operational Procedure, BLMP 2008), described in guidelines for several WFD assessment systems and in JAMP (Joint Assessment and Monitoring Programme) or TMAP monitoring guidelines (JAMP 1997, TMAP 2008). In Figure 2-9 the monitoring sites and the water type classification (N1–N5 types) relevant for the WFD status assessment are illustrated.

Transitional zone (T1–T2 water types): Responsible agencies are NLWKN and LLUR. The applicable assessment system is AETV (AestuarTypieindex-Verfahren, Krieg 2005, 2006), which evaluates the status of soft bottom macrozoobenthos. Measured parameters are species composition, abundance and biomass. Sampling device is a Van Veen grab; samples are sieved with mesh sizes of 250 µm (in contrast to the standard mesh sizes of 500 µm and 1 mm).

Coastal (1 sm) zone (N1–N5 water types): Responsible agencies are NLWKN and LLUR. The valid assessment system is M-AMBI (Borja et al., 2000; Muxika et al., 2007) for eulitoral and sublittoral soft bottom macroozoobenthos. Assessment parameters are species composition, abundance and (partly) biomass. Sampling devices are box corers and piston corers (eulitoral) or Van Veen grabs, box corer, and dredges in combination with remote sensing/video techniques (sublittoral). Sampling is carried out annually.

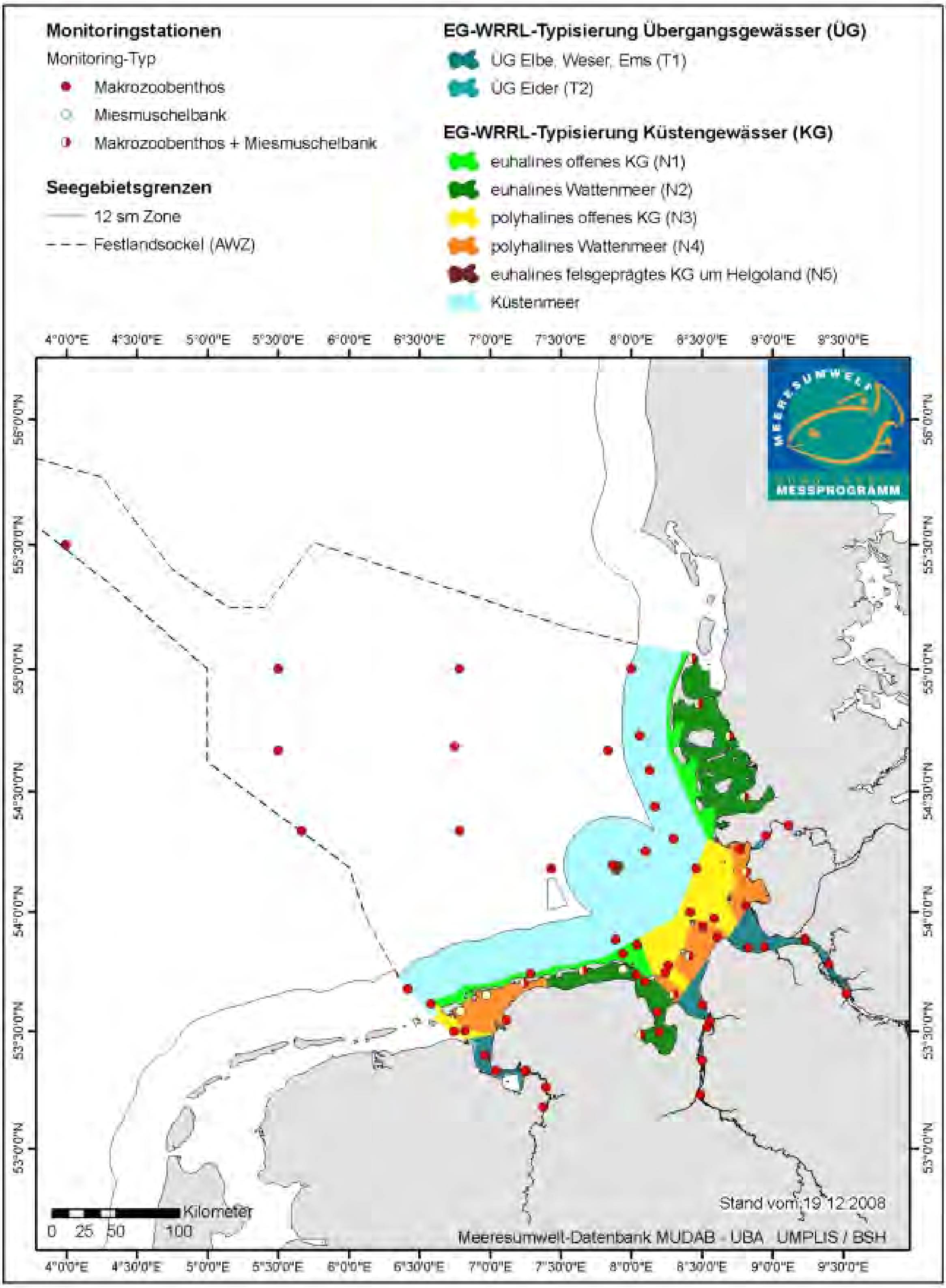

Figure 2-9: Macrozoobenthos monitoring in the German part of the North Sea (BLMP, 2010).

Eulitoral mussel banks are monitored on the basis of medium-scale aerial images and, to a certain extent, GPS-supported inspections in the field as well as sampling in specific mussel beds. Assessment parameters are areal extent of the bank, coverage, stocking density, biomass, abundance and length-frequency distribution. Sampling frequency is annual (spring), certain banks are sampled monthly to obtain higher levels of temporal resolution. The Baltic WFD assessment tool MarBIT (Marine Biotic Index Tool) is currently tested for application in Eulitoral mussel beds and sublitoral hard bottom makrozoobenthos at Helgoland.

2.4.1.4.2Baltic Sea

Three agencies (LLUR, LUNG and BfN) provide data for the macrozoobenthos monitoring of the German part of the Baltic Sea. The monitoring is executed either by the agencies themselves, scientific institutes (IOW) or private consultants.

The monitoring program is presently based on the requirements of the WFD (Water Framework Directive) with its assessment system MarBIT (Marine Biotic Index Tool) for German macrozoobenthos in the German coastal (1 sm) zone and the requirements of HELCOM (HELsinki COMmission), the HD (Habitats Directive) and MSFD (Marine Strategy Framework Directive) in the coastal (12 sm) zone and the EEZ.

The general monitoring procedure is standardized in a national SOP (Standard Operational Procedure, BLMP 2008), described in a guideline for the MarBIT assessment (Meyer et al. 2008) and the HELCOM Combine program (HELCOM 1999, 2003, Rumohr 2009). In Figure 2-10 the monitoring sites and the water type classification (B1–B4 types) relevant for the WFD status assessment are illustrated. Those water types are subdivided into geomorphological similar water bodies, which form the spatial unit for the assessment program.

Coastal (1 sm) zone (B1–B4 water types): Responsible agencies are LLUR and LUNG. The valid assessment system is MarBIT (Meyer et al. 2005), which evaluates the status of soft bottom (infauna), hard bottom and phytal macrozoobenthos (epifauna). Assessment parameters are TSI (Taxonomic Spread Index), log-normal distribution and proportion of sensitive and tolerant species. Measured parameters are species composition, abundance and biomass. Sampling devices are diver-operated sampling frames with net bags and video/photo documentation of sampling stations. The program started in 2006. Sampling is carried out yearly in spring (March-April) for soft bottom fauna and in summer (June-July) for phytal and hard bottom fauna. The number of sampled and assessed water bodies varies between years. All water bodies are assessed in minimum once in a 6-year period. Before the WFD monitoring started (1995–2004), macrozoobenthos was monitored at several transects based on the HELCOM requirements.

Coastal (12 sm) zone: Responsible agencies are LLUR and LUNG. Sampling is based on the HELCOM Combine Program for soft bottom macrozoobenthos. Measured parameters are species composition, abundance and biomass. Monitoring devices are ship-operated Van-Veen grabs. Sampling is carried out yearly in autumn to enable the detection of oxygen deficiencies at the bottom.

AWZ: The responsible agency is the BfN. The monitoring covers several Natura 2000 areas. Sampling is based on the HELCOM Combine Program for soft bottom macrozoobenthos but enables also the detection of epibenthic animal communities. Measured parameters are abundance biomass and species composition. Monitoring devices are ship-operated Van-Veen grabs, dredges and video. Sampling frequency is annual.

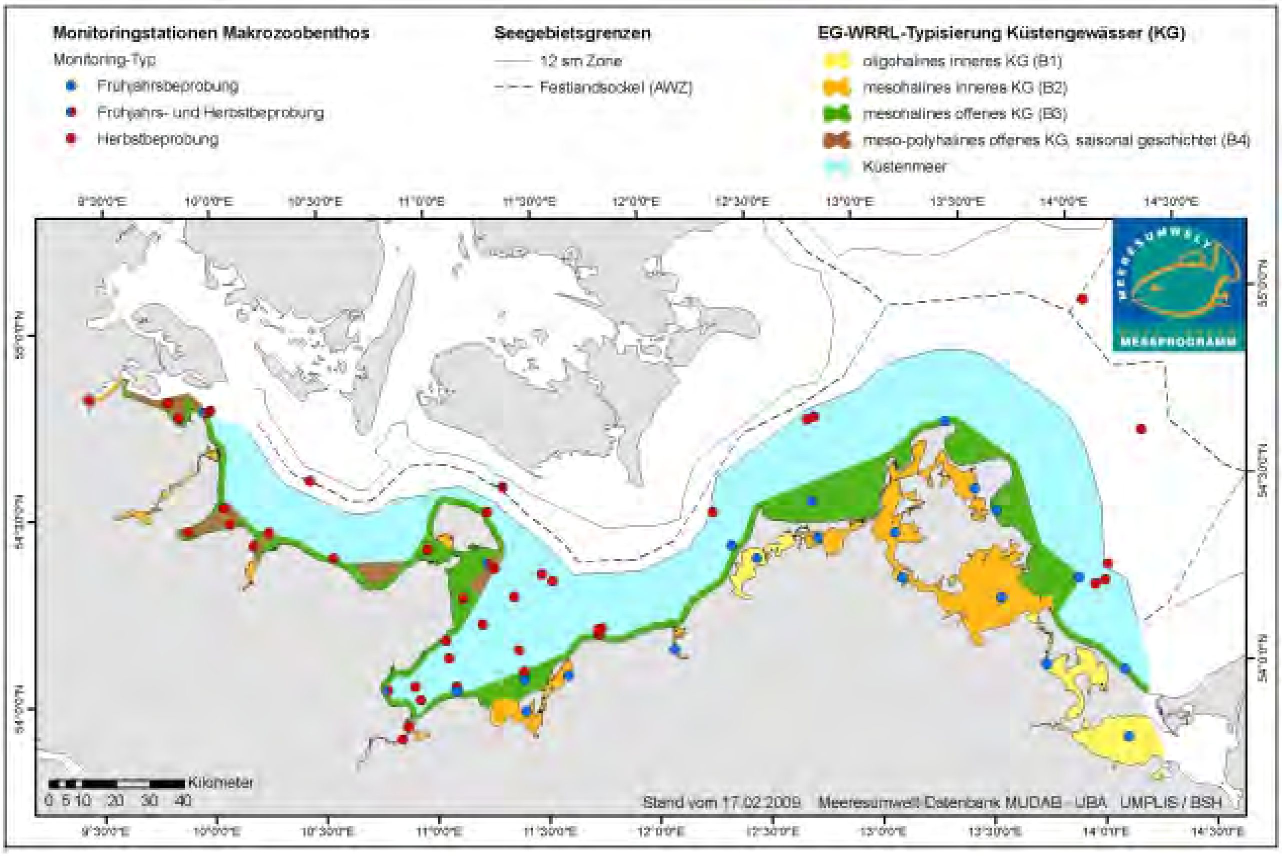

Figure 2-10: Macrozoobenthos monitoring in the German part of the Baltic Sea (BLMP, 2010).

2.4.1.5 Fish

The monitoring of the fish fauna in the North Sea, Baltic Sea and in transitional waters consists of stow net fishing (pelagic fish fauna) (Vorberg 1998), reports of rare species from professional and recreational fishermen (Thiel et al. 2007), beam trawling (demersal fish fauna) (DYFS) and trawl fishing (Survey Trawl Standardisation, ICES 2005; IBTS Manual, ICES 2006; GSBTS: Ehrich et al. 2007). To investigate the fish populations, the determination of age structure, biomass, distribution of priority species, habitat quality (spawning habitats), the ratio of cyprinids to percids and the size composition are essential.

2.4.1.5.1 North Sea

DYFS (Demersal Young Fish Survey)

The monitoring is carried out since 1974 by the vTI-SF (Johann Heinrich von Thünen-Institute - Institute of Sea Fisheries) and enables the assessment of spatial and temporal changes in fish communities in shallow coastal waters. The investigations take place annually with a beam trawl as fishing device. Figure 2-11 shows the vTI monitoring network.

GASEEZ (German Autumn Survey EEZ)

Spatial and temporal changes in fish communities are assessed by the vTI-SF annually, particularly in autumn, at 80 permanent stations distributed over the whole area of the EEZ in the North Sea. This monitoring started in 2004 with bottom and beam trawl as fishing devices in alternately years.

GSBTS (German Small-Scale Bottom Trawl Survey)

To survey small-scale and long-term changes in demersal fish fauna, investigations in three permanent study areas (boxes) are done. Each box measures 10x10 nautical miles. Samples are taken six-monthly in two boxes and annually in one box with a bottom trawl. The investigation takes place since 1987 and is executed by vTI-SF.

Stow Net Fishing in the Schleswig-Holstein Wadden Sea

The monitoring institution is the NPV SH (National Park Administration of Schleswig- Holstein) and takes place since 1991 (Figure 2-12). The monitoring is based on the requirements of TMAP (Trilateral Monitoring and Assessment Program) and gives a general view on the occurrences of Red List species. The annual sampling takes place in August with a stow net at three stations in the Hörnum Deep and at three stations in the Meldorf Bight.

Hydroacoustic Survey (Herring)

The monitoring to survey stock parameters of herring and sprat as basis for fishery assessment and managements carried out by vTI-SF and started 1987. It is an annual acoustic monitoring with accompanying fish catches by using pelagic trawls for the validation of the sonar data.

IBTS (International Bottom Trawl Survey)

Since 1991 the assessment of stock parameters for commercially exploited demersal fish species takes place (as basis for fisheries assessment and management). The executing institution is the vTI-SF. The investigation is carried out once a year (3rd quarter) in the ICES (International Council for the Exploration of the Sea) rectangles of the German Bight by using a trawl tow with GOV (Grande Ouverture Verticale).

Monitoring in the East Frisian Wadden Sea

Species composition, abundance and biomass of all fish species and decapods was started in 1998 by the AWI (Alfred-Wegener-Institut) to facilitate ecosystem research. It is presently continued in the context of climate research activities. Sampling takes place twice a year (March and July/August) in the Spiekeroog and Langeoog tidal channel system and along the 5 m line two hours before and after low tide off both islands with beam trawl of 3 m width.

Sole Survey

This annual monitoring of demersal fish in coastal waters by beam trawling takes place since 1976 and 1999 in selected areas of the EEZ (Natura 2000 areas). The executing institution is the vTI-SF. The monitoring data enable the determination of spatial and temporal changes in fish communities.

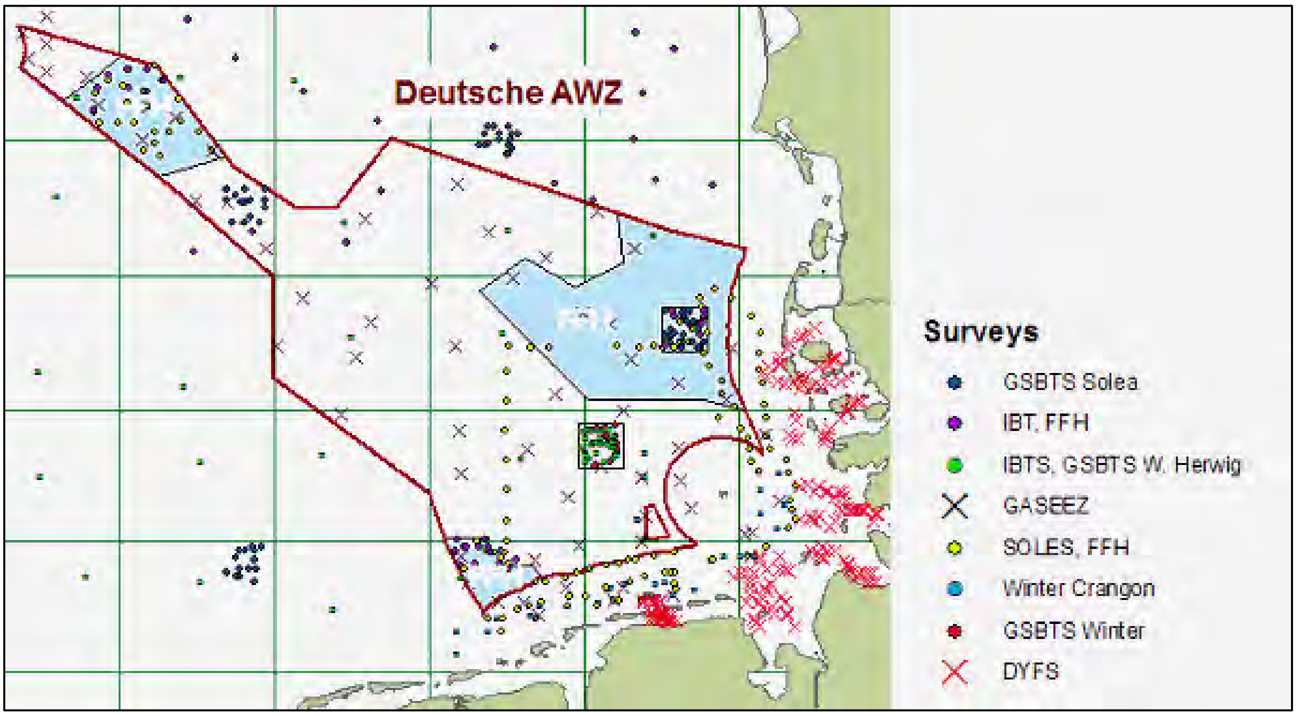

Figure 2-11: Fish monitoring (vTI-SF) in the German part of the North Sea (BLMP, 2010).

The monitoring and assessment of fish communities in transitional waters (currently Eider, Elbe, Ems and Weser) is done by stow net fishing and is based on the requirements of the WFD.

- Eider: The monitoring is carried out since 2006 each third year in early summer and autumn by the executing institution LLUR (State Agency for Agriculture, Environment and Rural Areas). The stow net fishing is done during two tides.
- Elbe: The monitoring is carried out since 2000 by the Elbe Water Quality Office (WGE). The investigation is practiced parallel to the monitoring of pollutants in fish, which has been a component of the German Marine Monitoring Program (BLMP) since 1986.
- Ems: The monitoring is carried out since 2006 by the executing institution LAVES (Lower Saxony State Office for Consumer Protection and Food Safety).
- Weser: The monitoring is carried out since 2002 by the executing institution LAVES.

2.4.1.5.2Baltic Sea

BITS (Baltic International Trawl Surveys)

The monitoring to determine stock parameters for commercially exploited demersal fish species as basis for fisheries assessment and management was started in 1991 and is, carried out by vTI-OSF (von Thünen Institute for Baltic Sea Fisheries) (Figure 2-13). The sampling takes place twice a year in the 1st and 4th quarter (approx. 50 trawl tows).

Box Monitoring in the Western Baltic Sea

The monitoring to survey small-scale and long-term changes in demersal fish fauna takes place since 2003 by vTI-OSF. Samples are taken in five permanent study areas (boxes) once per year (June) with ten tows per box using a TV trawl.

Hydroacoustic Surveys (Sprat and Herring)

This annual international acoustic monitoring with parallel pelagic trawls for the validation of sonar readings is used to survey stock parameters for herring and sprat as basis for fisheries assessment and management. It is carried out since 1992 by the vTI-OSF.

Monitoring in the Pomeranian Bight (previously Eel Survey)

The survey of long-term changes in demersal fish fauna takes place since 1993. 15 hauls with eel trawl on each occasion and, since 2002, an additional survey of small fish fauna with 2 m beam trawls are carried out.

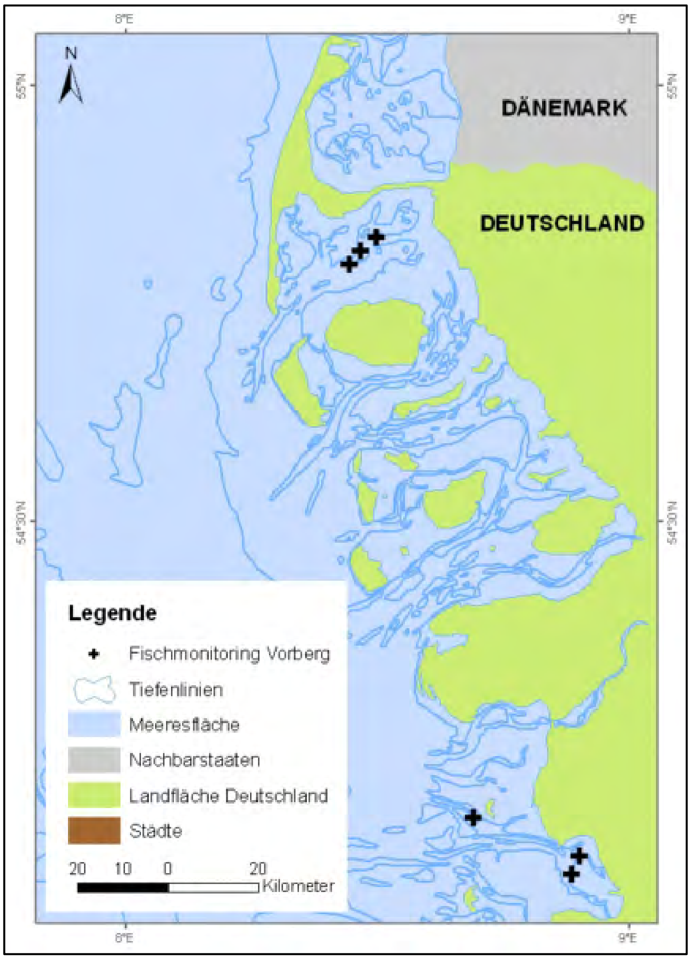

Figure 2-12: Fish monitoring (NPV), Schleswig-Holstein - annual stow net fishing in the Wadden Sea in the German part of the North Sea (BLMP, 2010).

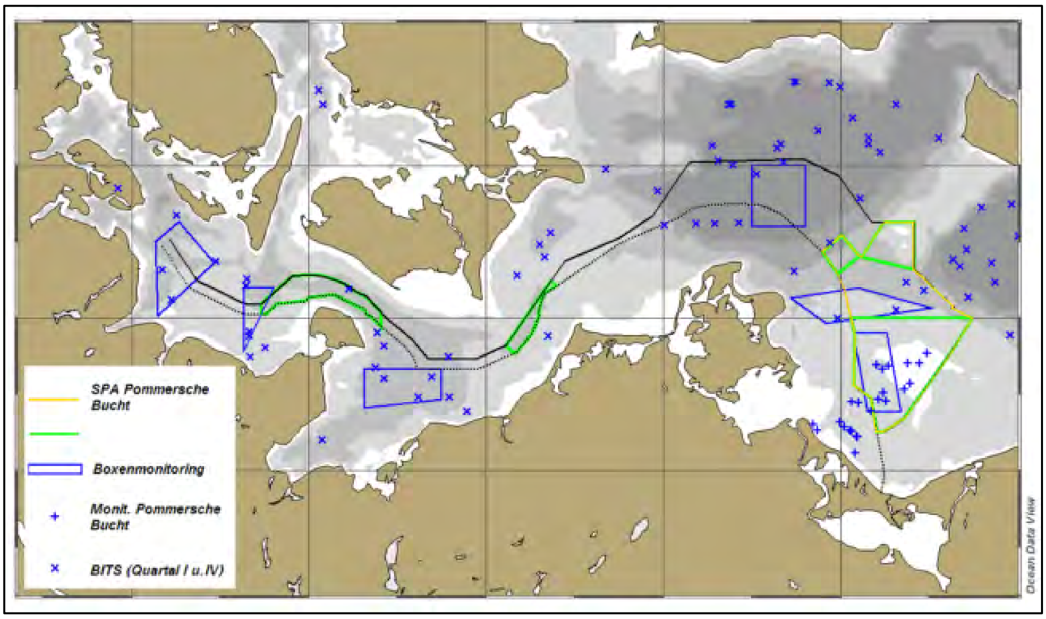

Figure 2-13: Fish monitoring (vTI-OSF) in the German part of the Baltic Sea (BLMP, 2010).

2.4.1.6 Birds

2.4.1.6.1 North Sea - Resting Birds

The monitoring of resting birds in the North Sea (Table 2-14) consists on investigations from spring tide counts based on Rösner 1995, resting population surveys based on Wahl et al (in preparation), resting birds surveys at sea by ship (Garthe et al., 2002) and plane (Diederichs et al. 2002; BSH 2007) and surveys of sea ducks at low tide (Kemp and Eskildsen, 2000).

Table 2-14: Overview of the monitoring program of resting birds (North Sea).

Frequency	Survey programme	Target species
Mid-winter count (January)		Seabirds, coastal birds
26 surveys a year		Representative resting sites at spring tide around high tide
Annually		Species specific surveys: 3x geese, 1x sanderling, at least 2x common eider, 3x moulting common shel-duck
Twice in six years (January)	Aerial survey at sea	Seabirds
Annually, at least every two years (spring and winter)	Two aerial surveys at sea (off-shore)	Gavia-Divers, little gull
Annually (summer and au-tumn/post-breeding period)	Ship-based survey at sea (offshore)	Terns, gulls and Helgoland cliff breeders
Annually	Aerial survey (offshore)	Common scoter

Table 2-15: Overview of the monitoring program of resting birds (Baltic Sea).

Frequency	Survey programme	Target species
Mid-winter count (January)		Seabirds, coastal birds
Eight mid-monthly counts (September – April)	Carried out from land	All wader and waterfowl species
Annually (January and spring)	Aerial survey (shallow grounds off Schleswig-Holstein)	Sea ducks
Annually (spring)	Two aerial surveys (deep-water areas off Schleswig-Holstein)	Sea ducks
Twice in six years (January)	Aerial survey at sea (entire German Baltic Sea)	Seabirds
Annually, at least every two years (spring and August)	Two aerial surveys at sea (conservation areas in the EEZ and Mecklenburg-Western Pomerania)	
Every two years (January)	Ship-based survey (Pomeranian Bay SPA and neighbouring areas)	
October - April	Monitoring of beached birds: gillnet victims	Representative random samples

2.4.1.6.2Baltic Sea - Resting Birds

The monitoring of resting birds in the Baltic Sea (Table 2-15) consists on investigations for resting populations from land, resting birds surveys at sea by ship (Garthe et al. 2002) and plane (Diederichs et al. 2002, BSH 2007) and the surveys of sea ducks at low tide (Kemp and Eskildsen 2000).

2.4.1.6.3North Sea and Baltic Sea - Breeding Birds

The monitoring of breeding birds consists on investigations for breeding populations based on Südeck et al. (2005) and Hälterlein et al. (1995), breeding success based on Thyen et al. 1998 and ringing programs (Table 2-16).

Table 2-16: Overview of the monitoring program of breeding birds (North Sea and Baltic Sea).

Frequency	Species/survey programme
Annually	Population survey of colony breeders and selected species
Annually (sample site) Every six years (complete survey)	Population survey of other species on the species list
Annually	Breeding success measurements of indicator species at selected breeding sites
	Studies of population structure (ringing programs)

2.4.1.6.4North Sea and Baltic Sea - Beached Birds

The monitoring of beached birds consists on investigations from driftline monitoring (TMAP Monitoring Handbook) and the monitoring of gillnet victims (presently only Baltic Sea). The monitoring frequency for the driftline monitoring comprises 13 surveys from October to April.

Figure 2-14 shows the investigation area of seabirds in North Sea and Baltic Sea. The monitoring program is based on the requirements of TMAP (Trilateral Monitoring and Assessment Program), Habitats and Birds Directive Directive, MSFD (Marine Strategy Framework Directive), HELCOM (Helsinki Commission), OSPAR

(Oslo and Paris Commission), RAMSAR (Ramsar Convention) and AEWA (African-Eurasian Waterbird Agreement).

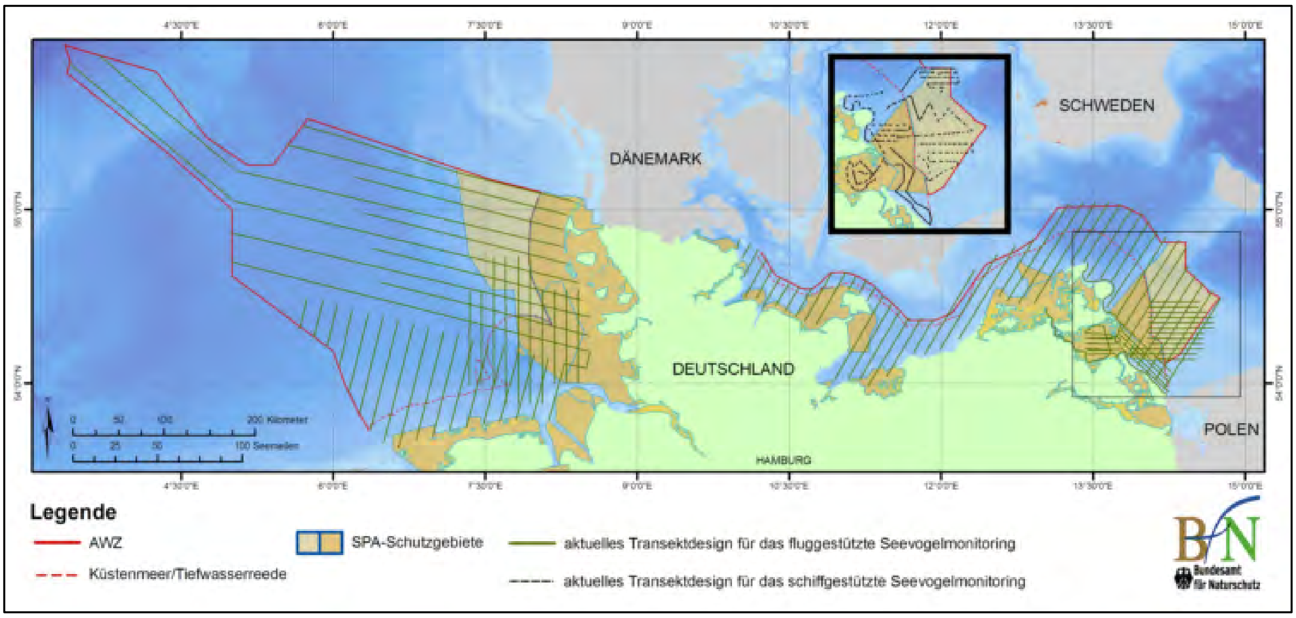

Figure 2-14: Monitoring of seabirds (BLMP, 2010).

2.4.1.7 Mammals

The monitoring program in North Sea and Baltic Sea includes the marine mammals common seal (*Phoca vitulina*), grey seal (*Halichoerus grypus*) and harbour porpoise (*Phocoena phocoena*).

Common seals and grey seals

The monitoring methods for common seals and grey seals are based on the SMP (Seal Management Plan as amended to cover grey seal monitoring) and the specifications of the LUNG (State Agency for Environment, Nature Conservation and Geologie of Mecklenburg-West Pomerania) concept in North Sea and Baltic Sea. The execution of dissection and diagnosis of health status follow Siebert et al. (2007) and Müller et al. (2004). Parameters of distribution, habitat use/quality, health status, mortality due to by-catches, population size and the reproduction/birth rate (proportion of mother/calf groups) are to be considered.

The BLMP (Bund-Länder Messprogramm) proposes two overflights at moulting time to determine the population and three overflights/ship-based/aerial surveys at the pupping time. Furthermore, the assessment of potential and current haul-out sites of juvenile and adult animals (monthly), a survey of as many animals found dead as possible and an examination of all suitable specimens is included.

Harbour porpoises

The monitoring methods for the distribution, habitat use/quality, health status, mortality, population size and the reproduction and birth rates are carried out by line transects with aircrafts (Buckland et al., 2001; Diederichs et al., 2002; Hiby and Lovell, 1998), static acoustic monitoring (POD = acoustic porpoise detector) (BSH StUK 2007) and the dissections and diagnosis of health status (Siebert et al., 2001).

The BLMP proposes for the North Sea:

Line transects by plane (Figure 2-15):

- Complete survey twice in six years in June (MINOS (Marine Warm-blooded Animals in North Sea and Baltic Seas) Area A-D)
- Surveys in protected areas annually (MINOS Area C)
- Lower Saxony and Hamburg territorial waters with extension to Borkum Reef Ground SCI (Proposed Sites of Community Importance): annually, twice in March/April

A year-round stationary acoustic monitoring and a complete and year-round survey of by-catches in accordance with international regulations should be carried out.

The BLMP proposes for the Baltic Sea:

Line transects by plane (MINOS Area E) twice in six years in summer and in combination with bird surveys in winter (MINOS Areas F and G) (Figure 2-15).

A year-round stationary acoustic monitoring (Figure 2-16), a complete, year-round survey of by-catches in accordance with international regulations and an examination of all suitable specimens should take place.

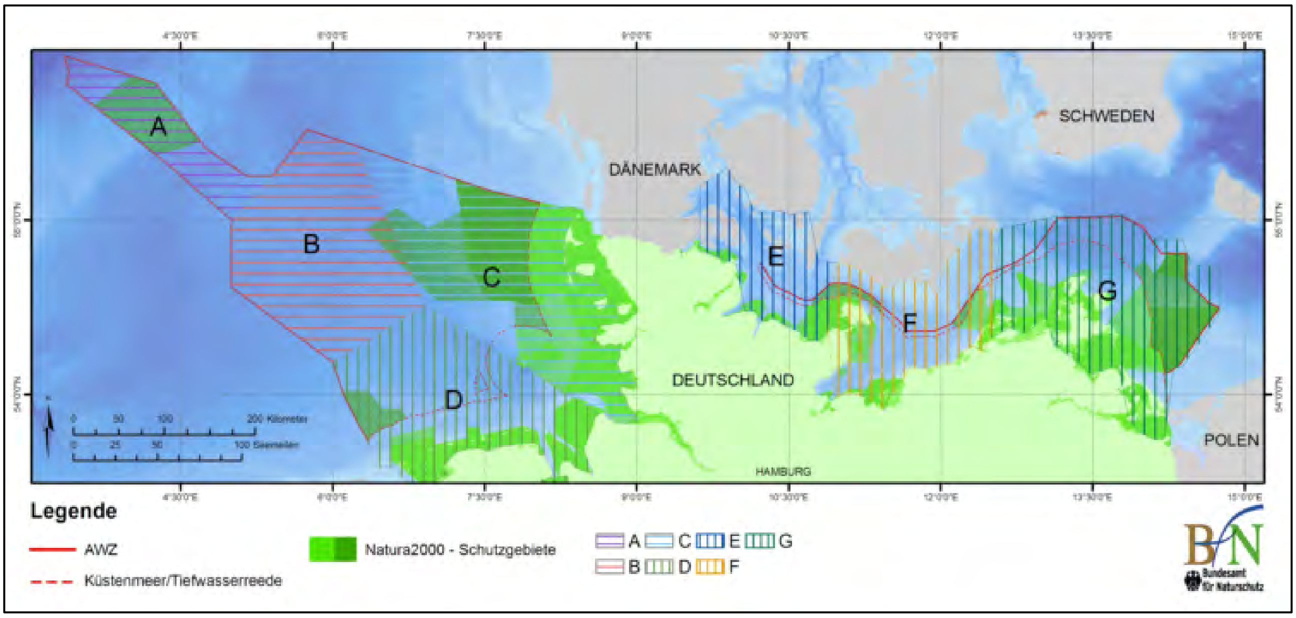

Figure 2-15: MINOS area and transect design for harbour porpoise survey flights (BLMP, 2010).

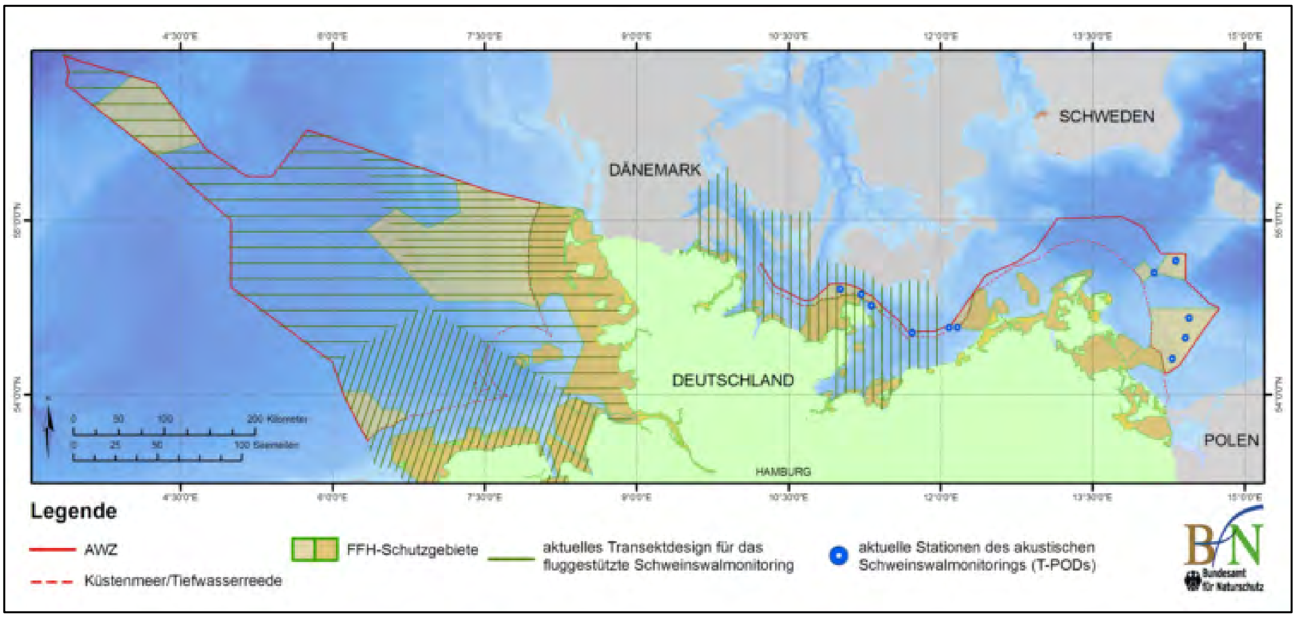

Figure 2-16: Monitoring of harbour porpoises, acoustic stations Baltic Sea (BLMP, 2010).

2.4.2 Proposed monitoring program with regard to the assessment of non-indigenous species

For the calculation of the proposed Non-Indigenous Species Index (NISI) as a basis for the classification of Descriptor 2 within the Marine Strategy Framework Directive, results from the existing biological monitoring programmes of the North and Baltic Sea can be used. Apart from some exceptions, the organism groups being relevant for the MSFD are regularly analysed on a qualitative and quantitative basis and do cover the main German marine areas, as described in chapter 2.4.1 Some strategic adaptions and supplements will be recommended within the next paragraphs.

The existing phytoplankton monitoring provides an almost complete temporal and spatial coverage of the coastal areas of the German North and Baltic Sea. For that reason, it is very likely that all non-indigenous species will be detected. A slightly higher sampling frequency along the coastline of Schleswig-Holstein during the growing season would be preferable. For the Baltic Sea, also the Exclusive Economic Zone (EEZ) is sufficiently covered with regard to phytoplankton analyses. Unfortunately, this does not apply for the German EEZ of the North Sea, since the regular monitoring of the phytoplankton communities established in 2008 had been cancelled after a four-year period. Therefore, it will be highly recommended to re-establish these monitoring acitivities with the previous frequency of sampling.

In Germany, the least intensive monitoring is performed for zooplankton. For that reason, it is recommended that some additional coastal stations should at least be sampled at a monthly sampling rate, in order to yield additional qualitative and quantitative information. The zooplankton sampling in the EEZ of the Baltic Sea will be sufficient, even if HELCOM recommends a higher sampling frequency. The current strategy seems suitable for detecting non-indigenous species. In the North Sea, there was no zooplankton monitoring in the EEZ at all after the samplings had ended in 2011. As for phytoplankton, it will be highly recommended to restart the regular monitoring activities for zooplankton in the EEZ as soon as possible.

As long as the species composition is part of the macrophyte monitoring, the system established in the Baltic Sea will have a sufficient spatial and temporal coverage with regard to the detection of any non-indigenous species. In order to achieve the same status for the North Sea, the sampling frequencies should be enhanced for selected sampling sites. Arial surveys being carried out only once a year will not be sufficient to reach that goal. With this method it is not possible to detect species.

The regular annual monitoring of macrozoobenthos covers both the EEZ and the coastal and transitional areas in the North Sea and Baltic Sea. Thus, non-indigenous specis can be detected for this group of organisms without requiring additional adjustments for these programs.

The same holds for the monitoring of the fishes. In the North Sea as well as in the Baltic, most areas are regularly monitored, facilitating an assessment of Descriptor 2 on the basis of these running programmes.

Current programmes for sea birds and marine mammals are mostly targeting abundances of selected species. If it is possible to detect newly occurring species with this monitoring, data from these programmes in the North and Baltic Sea can be used to calculate the NISI.

In general, the data of most German longer-lasting monitoring programmes have the potential to be used for calculating reliable NISI values for the assessment of Descriptor 2. In some cases, slight modifications or adaptations of the programmes might become necessary to achieve optimised surveys of the relevant areas in the North and Baltic Sea.

Within the routine monitoring, not all occurring non-indigenious species can be detected and classified. Very often, resources such as financial means, taxonomic knowledge and respective equipment are sparse and prevent the organisms from being analysed down to the species level or further down. In some cases, classification is conducted only on a higher taxonomic level. Those non-indigenious species which have persisted in the ecosystem for a longer time will normally be detected and be correctly identified. Species occurring less frequently or having invaded only recently will often be assigned to higher taxonomic groups, because analysts won’t have the time to determine down to the species level. Very often, the methodical configuration within the routine monitoring does not allow to detect certain genera down to the species or sub-species level, or to the variety or form.

With regard to the detection of all non-indigenious species, it has to be paid attention to the fact that some organism groups are not routinely analysed, but will only be detected intermittently within the frame of special research projects. This especially holds for the protozooplankton and the benthic microflora and microfauna.

Sometimes, ‘Rapid Assessment’ methods are carried out in order to monitor non-indigenious species. For that purpose, qualitative samples are taken and analysed for alien species, especially in particular high-risk areas. These methods might be suitable for a broad-brush analysis of the distribution and spreading of known and reliably detectable non-indigenious species. But it is questionable whether such methods are suitable for detecting new species. Very often the financial support for these programmes is not sufficient to carry out intensive taxonomic analyses of challenging organism groups and to apply complex analysing methods.

A complete and overall surveillance and inventorying of the marine environment with regard to non-indigenious species of all organism groups cannot be achieved by the current routine monitoring. Nevertheless, exact and comprehensive investigations should be carried wherever and whenever possible. In order to achieve that goal, existing resources will have to be used in an economic way, currently running programmes will have to be optimised and supplemented. The persons in charge for analysis of samples have to be trained optimal in taxonomy. These include the implementation of taxonomic workshops and regular participation in interlaboratory comparison tests.

Especially the transport by ballast water of ships is one of the main vectors for the dispersal of non-indigenous species over long distances and into far distant areas. Therefore, institutions and governments have a focus on invaders in that context. Within HELCOM, a project with regard to the risk assessment for exemptions from ballast water, treatment relating to the 'Ballast Water Management Convention' according to Regulation A-4 was conducted (Heyer 2012). Existing ballast water risk assessments were checked and tested on three shipping routes between international ports in the North Sea and Baltic Sea. The methods were discussed and implications for a further development given. Most of the methods were based on target species and environmental matching between donor and recipient port, some add the travel duration of the ships. They highly rely on the information on ports and species attributes which often is limited. One conclusion was that for all international ports around the world baseline studies should be started with the focus on non-indigenous species and that they should be appended to regular monitoring programmes. However, for assessing the environmental matching between ports the information on the species' environmental tolerances will have to be known. This information will not be available for each species. Furthermore, the assessment may even fail when this information is known, because many potential invaders have broad environmental tolerances, for example with regard to salinity. Thus, a decision will be not trivial. The assessment systems rely on the definition of target species. Target species are defined e.g. by their invasiveness, evidence of prior introduction, their distribution and their relation to ballast water introduction. On one hand, missing data will limit this definition of target species. On the other hand, only potentially harmful species are taken into account and native species in the donor ports are not included. This has practical reasons since the impact of the latter cannot be calculated easily. Additionally, many methods will not assign species as target species when these species are able to reach the recipient port on other ways than the ballast water pathway, for example by natural spreading or by introduction by another vector. As a conclusion, we propose to include all non-indigenous species.

A further study within HELCOM analyzed the necessary monitoring and the sampling procedures (HELCOM, 2013a). The proposed sampling methods were based on the CRIMP sampling protocol (Hewitt and Martin 2001), combined with rapid assessment protocols (Pederson et al. 2005, Cohen et al. 2005, Buschbaum et al. 2010) and HELCOM monitoring protocols (HELCOM Combine manual) and were tested in the field. Currently, Estonia is the only country in the Baltic Sea area carrying out a regular monitoring on non-indigenous species in harbours with a relatively high temporal resolution. Sporadic and isolated investigations are also conducted in Poland, Lithuania, Germany and Finland, mainly in form of rapid assessment procedures. Based on the outcome of these studies, we recommend that samplings should be carried out at least once or twice per year, depending on the organism group of interest.

Both studies mentioned before led to the Joint HELCOM/OSPAR Guidelines (HELCOM 2013 b) where beside the assessment system the monitoring and sampling is described in detail. Although the resulting proposed assessment system is limited for the application on ballast water exemptions only, it could be a very usable supplement for the existing monitoring, because it would cover additional risk areas.

In spite of the existing restrictions described above, it will be an ecologically worthwhile approach to continue and optimise the monitoring of the distribution and spreading of non-indigenious species. This will be the only way to check the efficiency of the applied measures and to develop new instruments with regard to the prevention of the distribution and further spreading of non-indigenious species.

2.4.3 Early warning system

Once a species is introduced, established and may spread further it is not trivial to remove it from an ecosystem, especially in the marine area. Successful invaders have certain features like broad environmental tolerances and high dispersal potential often combined with effective reproduction strategies and high numbers of propagules. Therefore, the best preventive measure would be to avoid new introductions. This is reflected by the proposed trend indicator (see chapters 2.3.1.1 and 2.3.3.4).

With the ‘International Convention for the Control and Management of Ships’ Ballast Water and Sediments’ (International Maritime Organization 2004) and the ‘ICES Code of Practice on the Introductions and Transfers of Marine Organisms’ (ICES 2005) there already exist international agreements to prevent the spread of non-indigenous species via two important vectors, ballast water and transfer of aquaculture organisms.

In addition to other, yet to be developed preventive measures, an early warning system could be conceivable. Foreign species could be detected and identified at a very early stage of introduction to possibly eradicate them and prevent their further establishment and dispersal. In the terrestrial area, early warning systems for the detection of non-indigenous species with the participation of the public and nature conservation organisations are quite practical concerning higher plants and animals. Whether such an approach can be transferred to the marine area seems doubtful. At least, this is excluded for all microorganisms in the different habitats of the ecosystems. A professional early warning system requires an intensive monitoring with a high temporal resolution at particular high-risk regions like ports, aquaculture areas, river deltas and channels where introductions are especially likely. All important organism groups have to be included into this monitoring and expert taxonomic knowledge is crucial.

For many organism groups, a sampling just once or twice a year will not be sufficient to early detect the arrival of new species. Consequently, monthly or weekly checks in the problem areas will be necessary, especially with regard to the planktonic system. Most benthic organisms spread as planktonic larvae and sessile macroalgae via spores and other spreading mechanisms, using water as a transport medium. But for larvae and spores it is very difficult to get analyses down to the species level. In that case, it will be highly questionable whether the great efforts – such as necessary high temporal and spatial sampling frequencies, special and cost-intensive methods for taxonomic classifications etc. – will lead to adequate results.

Instead, we would like to propose an optimisation of the existing long-lasting monitoring programmes by adding slight modifications and supplements, as described in chapter 2.4.2. In addition, the monitoring system proposed by OSPAR/HELCOM should be established in all German harbours with international traffic, and be applied with the minimum sampling frequency. By that measure, potentially problematic areas would be subject to regular surveillance. And last but not least, the taxonomic knowledge and skills of the analysts as well as the regular quality assurance measures should be intensively and continuously improved.

Main focus should lie on avoiding the introduction of non-indigenious species. The compliance with existing guidelines and conventions, such as the rules for treating ballast water, will have to be regularly validated with scientific methods. Currently applied rules will possibly have to be checked, adjusted, concretised and maybe strengthened. For areas not being covered at the moment, new rules and guidelines will have to be set up by experts.

3 Work package 2: Seafloor integrity - Physical damage, having regard to substrate characteristics (Descriptor 6)

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3.1 Summary

3.1.1 Objective

Within the framework of the research and development project 'Compilation and assessment of selected anthropogenic pressures in the context of the Marine Strategy Framework Directive' BioConsult Schuchardt & Scholle GbR was commissioned with the development of a concept to assess indicator 6.1.2: 'Extent of the seabed significantly affected by human activities for the different substrate types'. This work was also funded by the Bundesamt für Naturschutz. This report is the preliminary draft of a methodology for the national assessment concept and also presents suggestions for setting baselines and targets for the Good Environmental Status. Results of the first application of the proposed concept for the German Exclusive Economic Zone of the North Sea are shown and discussed.

3.1.2 Methodology

Identification of human activities and pressures

Human activities affecting the seabed and their impacts are described. Activities are assigned to predefined pressures based on specifications by the MSFD: Physical loss (sealing, smothering) and physical damage (selective extraction, abrasion, changes in siltation). Anthropogenic activities considered in the EEZ of the North Sea are bottom trawling, permanent offshore installations, aggregate extraction and pipelines. In order to assess the spatial extent of pressures the area affected by each activity is defined. The temporal extent is determined by means of a five-step scale ranging from rare (once per reporting period) to persistent (permanent installation or more than three times per year). Each pressure is visualized separately on a GIS-based map.

Assessment of habitat sensitivity

The MSFD differentiates between 'predominant' (broad-scale habitats based on EUNIS level 3) and 'special habitats' (habitats protected under EU, regional or national legislation). Based on a preliminary map on sediments and Natura 2000 habitats, three predominant and three special habitats are identified in the EEZ of the North Sea.

The method to assess habitat sensitivity is mainly adopted from the MarLIN approach developed by Tyler-Walters et al. (2001). The sensitivity of ecosystem components is determined by two aspects: the ability to withstand disturbance or stress (resistance or tolerance) and the ability and time needed to recover from a perturbation and return to the previous state (resilience or recoverability). Resistance and recovery time are categorized in relation to each pressure both for the physical habitat features and the characteristic species. Information on the potential impact of physical disturbance and the response of specific habitats and species is based on evidence as far as available. A decision matrix is used to automate the combination

of resistance and recoverability and to obtain sensitivity categories for the physical habitat and the characteristic species. The highest (i.e. most sensitive) rank assigned to either habitat structure or species determines the overall habitat sensitivity.

Characteristic species used for the sensitivity assessment of benthic habitats in the EEZ were mainly those identified by Rachor & Nehmer (2003) for the classification of benthic communities in the south-eastern North Sea. This selection of characteristic species is assumed as a preliminarily approach for the initial application of the methodology. For future assessments it is proposed to mainly refer to results of an ongoing habitat mapping project, which also should provide information on characteristic species of benthic habitats.

Physical impacts on benthic habitats

The degree of physical impact on a habitat is a product of its sensitivity and the exposure to a specific pressure. An impact assessment thus requires the linkage of sensitivity information with pressure data. A matrix combining pressure intensity in terms of the temporal extent and habitat sensitivity supports the classification in nine categories of physical impact. A percentage value is assigned to each rank which should provide an approximation of the relative impact on the habitat with regard to e.g. habitat structure, species richness, abundance or biomass. Due to the different nature of the pressures ‘selective extraction’, ‘abrasion’ and ‘changes in siltation’, for each of these physical damage pressures a separate impact matrix is provided in order to include a weighting factor in the impact assessment. ‘Sealing’ and ‘smothering’ are persistent pressures which are associated with an impact that destroys habitat structures as well as benthic organisms. The habitat is not expected to recover, thus sealing and smothering always result in a very high impact or total loss of habitat (100%).

In order to determine the cumulative physical impact on a particular habitat, the separate impact maps have to be summarised. Most approaches to assess cumulative impacts assume additive effects for lack of knowledge on actual responses of benthic habitats. It is proposed to follow this practice as the physical pressures regarded here are assumed to affect habitat structure and suitability in a similar mode. This means that percentages for overlapping physical impacts are added up with 100 % (total loss) as maximum value. The cumulative physical impact is calculated from the proportion of area impacted (A, [%]) for each habitat and the corresponding value for impact intensity (I, [%]) as derived from the impact matrices. The cumulative impact (CI, [%]) for each habitat results from the sum of individual values for the relative impact on habitat:

$$CI = \sum I \times A / 100 \ [\%]$$

High values of cumulative impact indicate either pressures with considerable temporal and spatial extent or habitats with high sensitivity towards the occurring pressures. The cumulative impact value may range from 0% which would be a habitat completely without impacts to 100% meaning the total loss of the habitat.

This method provides the advantage of easily comparing the different impacts of the pressures physical loss (reduction in extent) and physical damage (impairment of condition) and results in a single percentage value of physical degradation for each habitat.

3.1.3 Application of assessment concept

A first application of the proposed assessment concept was carried out for the German EEZ of the North Sea. Data used for the assessment were VMS data from 2006, the area extracted in 2005 / 2006 and permanent offshore installations under construction or in operation in 2013.

Pressures in the EEZ

In terms of area, 'abrasion' caused by bottom trawling is the main pressure which covers nearly the complete seabed of the EEZ (98.9 %). Areas without abrasion are solely the construction sites of offshore wind farms as well as operational wind farms. Areas subject to physical loss currently account for less than 0.01 % of the total area. The pressure 'changes in siltation' affects 1 % of the EEZ with the predominant activity being the construction of offshore wind farms. Selective extraction in 2005 / 2006 was restricted to an area of 0.02 % of the EEZ.

Impacts which interfere with each other are areas with aggregate extraction and bottom trawling as well as pipelines and bottom trawling. Other human uses are mutually exclusive, for example construction works and bottom trawling or operational wind farms, where fishing is excluded.

Cumulative physical impact on benthic habitats

The calculated cumulative impact values range from 20.1 % for sandbanks on the Borkum Reef Ground / Sylter Outer Reef to 47.6 % for reef habitats. The cumulative impact of predominant habitats adds up to approximately 40 % (sublittoral sand 37.9 %, sublittoral mud 40.3 %, sublittoral coarse sediment 43.8 %). For the special habitat 'species-rich habitats on coarse sands, gravel or shell debris' an impact value of 34.0 is calculated and for the separately assessed sandbank at the Doggerbank the cumulative impact accounts for 44.5 %.

The impact values mainly arise from high impacts of bottom trawling. Major parts of the benthic habitats are fished more than once a year, e.g. 65 % of the widespread sand habitats are subject to trawling more than 1.5 times per year. The comparatively low cumulative impact value for 'other sandbanks' originates from the lower fishing pressure on the Borkum Reef Ground in 2006, where half of the sandbank area was trawled less than once a year. The high impact value for reefs is mainly caused by the high sensitivity towards 'abrasion' determined for this habitat.

Physical impacts on marine protected areas

The physical impact of the individual pressures has been calculated for benthic habi-tats in marine protected areas as well.

<u>Sylter Outer Reef:</u> The cumulative impact on benthic habitats in the Sylter Outer Reef ranges from 31.0 % for the predominant habitat 'sublittoral sand' to 56.1 % for 'sublittoral mud'. High impact values were also calculated for 'sublittoral coarse sediment' (46.7 %), 'reefs' (51.5 %) and 'sandbanks' (56.0 %). The wide range of cumulative impact values corresponds to varying fishing intensity in the Sylter Outer Reef. While large parts of the Natura 2000 site were fished with low intensity, other areas were subject to persistent fishing pressure of up to five times per year.

<u>Borkum Reef Ground:</u> The only physical pressure affecting benthic habitats at the Natura 2000 site Borkum Reef Ground is 'abrasion' caused by bottom trawling. In 2006, fishing intensity was comparatively low with generally less than once per year. With the exception of reef habitats, the cumulative impact values for habitats in the Borkum Reef Ground were likewise relatively low, varying from 6.0 % to 23.7 %. The habitat 'sandbank' which covers the major part of the protected site (77.4 %) holds a cumulative impact of 8.7 %. Due to the high sensitivity rank of reefs towards 'abrasion', the cumulative impact of this habitat type amounts to 40.1 %.

<u>Doggerbank:</u> The total area of the Doggerbank is subject to 'abrasion' by bottom trawling and is additionally crossed by three gas pipelines. The cumulative impact of the main habitat 'sandbank' (95.6 % of total

area) at the Doggerbank accounts for 51.2 %. The impact values for 'sublittoral sand' amounts to 34.4 % and for 'sublittoral mud' 20.2 %. However, muddy habitats cover only 0.02 % of the total area.

3.1.4 Baseline and GES targets

Within the MSFD the baseline is defined as a state or condition against which the Good Environmental Status can be assessed. In the proposed concept the determination of a baseline or reference state is also important for habitat sensitivity as the assessment should be based on habitats in an optimum state. Methods for setting baselines for marine benthic habitats are described and in accordance with most experts the use of the 'reference state' is recommended, i.e. the state when pressures are absent or negligible. For most habitats it is proposed to use the existing reference state as a baseline, which is the current extent of habitats and a condition without impacts from anthropogenic pressures. This reference state reflects current physiographic, geographic and climatic conditions and should be also used to determine habitat sensitivities. For some aspects of the reference state it is recommended to take account of historical data, e.g. regarding larger and long-lived species which are presently underrepresented in benthic habitats. Expert judgement or spatial modelling may also aid in setting reference conditions. However, this approach does not take into account habitats which have deteriorated in range and extent due to human impacts, e.g. European oyster beds or *Sabellaria spinulosa* reefs. For these habitats the historical extent and reference conditions in combination with expert judgement have to be considered.

Targets for the Good Environmental Status represent boundaries or thresholds between the acceptable and unacceptable status of the marine environment (GES or below GES). Targets used for the assessment of benthic habitats within the Habitats Directive, the Water Framework Directive and Regional Seas Conventions are described and discussed. For the assessment of indicator 6.1.2 two major options are considered:

- a quantitative target based on the cumulative impact value, e.g. the cumulative impact value on predominant habitats must not exceed 15 %
- a dual quantitative target for area with no or low impact and for cumulative impacts, e.g. area with no / low impact must be at least 15 % and the cumulative impact value must not exceed 15 % for predominant habitats

It is also acknowledged that different targets have to be established for predominant and special habitats, as the latter are already designated for special protection. An additional target is recommended for habitats which have suffered major deteriorations due to human impacts in the past, e.g. European oyster beds. The objective should initially be an increase in habitat area towards historical extent. The indicator also provides the opportunity to set a GES target for marine protected areas (habitats in particular areas), which include both predominant and special habitats. This could be a designated area without any human uses.

3.1.5 Further development of the assessment concept

With the present report, the assessment concept is already at an advanced stage so as to allow for a good estimation of physical impacts on benthic habitats. In order to improve the results of future assessments several enhancements are suggested which include the improvement of sensitivity assessments, introduction of levels of confidence, analysis of possible linking between indicator 6.1.2 and 'condition indicators' and modification of the concept for coastal waters. For further assessments it should as well be tried to improve data base, especially on fishing pressure and aggregate extraction. In spite of these unresolved issues, the proposed methodology presents a major step for assessing cumulative physical impacts on benthic habitats. The concept provides a simple, cost-effective and informative method which is easily applicable to other marine regions.

3.2 Objective

According to the Marine Strategy Framework Directive (MSFD), the Good Environmental Status of Descriptor 6 is achieved when 'seafloor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected' (EC 2008). The objective is that human pressures on the seabed do not hinder the ecosystem components to retain their natural diversity, productivity and dynamic ecological processes, having regard to ecosystem resilience (EC 2010). The indicator 6.1.2 'Extent of the seabed significantly affected by human activities for the different substrate types' aims to address pressures causing physical damage or loss to seafloor habitats and to assess the proportion of habitat area permanently or temporarily affected by anthropogenic use. The assessment of the indicator integrates information on the spatial extent and intensity of physical pressures and on the spatial extent and sensitivity of benthic habitats.

Within the framework of the research and development project 'Compilation and assessment of selected anthropogenic pressures in the context of the Marine Strategy Framework Directive' BioConsult Schuchardt & Scholle GbR was commissioned with the development of a concept to assess indicator 6.1.2. This work was also funded by the Bundesamt für Naturschutz. This report is the preliminary draft of a methodology for the national assessment concept and also presents suggestions for setting baselines and targets for the Good Environmental Status. Results of the first application of the proposed concept for the German Exclusive Economic Zone of the North Sea are shown and discussed.

3.3 Rationale

The MSFD requires an analysis of the state of habitats and the distribution and intensity of anthropogenic pressures impacting upon them. National marine strategies should include an assessment of pressures and impacts arising from human activities in order to obtain a better understanding and management of those pressures and impacts with the objective of reducing them and to achieve or maintain Good Environmental Status in 2020.

OSPAR and HELCOM as Regional Seas Conventions in the area are currently developing indicators for the assessment of physical pressures and impacts on benthic habitats, both to cover the MSFD requirements and regional projects such as the Baltic Sea Action Plan. The general importance of these indicators is agreed among the experts of member states, however, work on them is still in progress and the respective indicators are not yet approved.

Indicator 6.1.2 is considered to be highly sensitive to physical pressures such as sealing, smothering, abrasion or extraction. The assessment of human activities allows for an adequate deduction and quantification of pressures and impacts on benthic ecosystems. Principally, the indicator should be applicable throughout the national waters and be able to assess all kinds of habitats, predominant as well as special habitat types. As it is designed as a pressure indicator, this presents the advantage of directly providing information on the cause of changes in ecosystem components. Pressure indicators are regarded as providing the evidence for the need of management and may offer the opportunity to appropriately manage human activities affecting the environment.

As the suggested approach for the assessment of indicator 6.1.2 is mainly based on modelling the impact by combining habitat sensitivity maps with spatial pressure data, it is considered to be highly cost efficient. Information on pressures and human activities should be available e.g. from projects requiring licensing procedures or from VMS data for bottom trawling. Sensitivity data may be derived from existing programmes such as monitoring for the Habitats Directive in the case of special habitat types. Once the methodology is established, further application needs only current data on localisation and quantification of the different physical pressures. Additional monitoring is currently not regarded as required for the assessment, although it may become necessary to calibrate the method and improve confidence in the results. Validation of the concept may be done by means of the condition indicators of Descriptor 6 or by directly monitoring different levels of known human impact.

The proposed concept is based on guidance provided by the European Commission and a literature review of existing scientific studies dealing with similar subjects. Current discussions regarding the implementation of the MSFD, taking place e.g. within the Regional Seas Conventions such as HELCOM and OSPAR were likewise considered.

3.4 Methodology

3.4.1 Principles

The parameter to be modelled and measured for the assessment of indicator 6.1.2 is the area of damaged and lost habitats. The approach proposed is based on modelling the impact by combining pressure-specific sensitivity maps for benthic habitats with data on the spatial and temporal extent of physical pressures. Habitat sensitivity is determined by resistance (the ability to withstand disturbance or stress) in relation to a specific pressure and recoverability following the disturbance. The responses of habitats to physical pressures are linked to assess the cumulative physical impact on habitats.

The suggested methodology refers to existing approaches for vulnerability or impact assessments, trying to combine already approved and accepted concepts with the requirements of indicator 6.1.2. The magnitude of pressures and the sensitivity of habitats are qualitatively expressed in ranks and these ranks are combined by means of a decision matrix. This standardisation shall ensure that the assessment is able to compare different pressures and the responses of different habitats. The scales proposed for the assessment concept are intended to reflect likely levels of intensities or damage and are to be used within the evidence base. The setting of categories is based on existing concepts for vulnerability assessments (e.g. Tyler-Walters et al. 2001, Tillin et al. 2010) as well as expert judgement. It may still prove necessary to revise or adjust the categorisation in the further development process of this indicator concept.

The proposed concept relies mainly on available scientific evidence, which enables the assessment process to be automated and thereby ensures its reproducibility. Expert judgement also plays an important role, e.g. in the setting of scales or in the case of insufficient data on pressures or habitats. Data are processed and visualised on maps by means of a Geographic Information System (GIS).

3.4.2 Anthropogenic activities and pressures

Human activities and associated pressures potentially causing physical damage to benthic habitats must be identified. Initially emphasis will be on activities in the German EEZ such as bottom trawling, offshore constructions or sediment extraction, however, the approach should also be able to encompass anthropogenic uses in coastal waters. An indicative list of human activities with the potential of physically disturbing the seabed is provided by the EC (2011).

Pressures resulting from anthropogenic activities can be described as changes in physical, chemical or biological properties of the environment compared with background levels or a reference condition. Depending on the intensity, pressures have the potential to cause direct or indirect impacts on the components of the ecosystem (WG GES 2011). Physical pressures on the seabed may alter the structure and functioning of marine habitats and thus indirectly affect the benthic community.

According to Annex III, table 2 of the Directive a distinction is made between physical loss, which relates to the spatial extent of the habitat and physical damage, which affects the condition of habitats. Physical loss is defined as a permanent or long-term alteration of the habitat by changing the natural substrate (smothering) or by conversion of marine to terrestrial or freshwater habitats (sealing). In contrast, physical damage refers to a disturbance of the habitat where the same or similar natural substrate is retained but its structure and biota are altered (MSCG 2012). Effects associated with physical damage according to the MSFD Annex III are changes in siltation, abrasion and selective extraction. Definitions of pressures are proposed based on existing definitions of physical disturbance by the MSFD, OSPAR (2012), Tyler-Walters et al. (2001) and Tillin et al. (2010) (Table 3-1).

Table 3-1: Proposed definitions of physical pressure, adapted from EC (2008) and OSPAR (2012).

Physical loss
Smothering – change to another seabed type Permanent or long-term change of one marine habitat type to another marine habitat type, e.g. where soft sediments are replaced by hard or coarse substrates including artificial substrates. Alteration of habitat features will result in distinct changes in the benthic community. Associated activities: offshore installations, scour protection, aggregate extraction, capital dredging, disposal of dredged material, coastal defence structures. **Sealing** Permanent loss of marine habitats to land or freshwater habitats or man-made constructions. Associated activities: land claim, foundations of offshore installations.
Physical damage
Changes in siltation Settling out of sediments suspended in the water column, accumulation or erosion of fine sediments on the seafloor (smothering). Associated activities: offshore installations, land claim, coastal defence, extraction of aggregates, dredging. **Abrasion** Penetration or disturbance of sediments where there is limited or no loss of substrate from the system. Associated activities: bottom trawling, anchoring. **Selective extraction – removal of substratum** Removal of substratum where the exposed sediment is of the same type. Changes of habitat structure are temporary and / or reversible; re-colonisation by a similar benthic community is possible after the extraction event. Associated activities: extraction of aggregates, dredging.

3.4.2.1 Identification of activities and pressures in the German Exclusive Economic Zone

The assessment of indicator 6.1.2 requires a conceptual understanding of the potential impacts on benthic habitat structure and suitability caused by physical pressures. In this section human activities occurring in the German EEZ are described in terms of their geographical distribution and their physical or mechanical impact, which is considered to result from the spatial and temporal footprint of the associated pressures. Table 3-2 summarises the information from this chapter and links human activities in the German EEZ with the definitions of physical pressures from Table 3-1.

Bottom trawling

Demersal trawling takes place in large parts of the Baltic and North Sea. A survey regarding the effects of bottom trawls on the benthic fauna showed that only small areas of the North Sea are not regularly fished (Schroeder et al. 2008). Fishing activities are solely restricted in the three nautical mile zone in the Baltic Sea and the 'plaice box' in the North Sea, which is closed for larger beam trawlers (BSH 2009a, BSH 2009b). Fishing gears employed in the North Sea are mainly otter trawls in the northern part and large and heavily-rigged beam trawls in the southern part of the EEZ. The highest fishing intensity with 10 to 15 events per year was registered in coastal waters of the North Sea up to a distance of 25 km to the coast. Coastal fisheries are mostly carried out by small beam trawl vessels (< 300 hp) targeting shrimp, plaice and sole. Major parts of the German EEZ of the North Sea are currently fished once a year, while the maximum fishing intensity is approximately five events per year (Schroeder et al. 2008). In the Baltic Sea, fisheries with towed gear is less intense and mainly carried out by otter trawls, both inshore and offshore (Janssen et al. 2008, Pedersen et al. 2010).

Fishing with towed bottom gears causes physical disturbance of the sea bottom and therefore adversely affects benthic habitats and communities. Effects include the reduction of habitat complexity, alterations in sediment characteristics and removal of structuring features. The passage of the fishing gear over the seafloor disturbs the upper bottom layers thereby causing a re-suspension of sediments, re-mineralisation of nutrients and contaminants and re-sorting of sediment particles. Habitat structures are altered in terms of a homogenisation of the seabed, e.g. by flattening of sand ripples, removal of rocks or structuring organisms such as biogenic reefs, epibenthic fauna or burrows and mounds (Kaiser et al. 2002). Fining of sediments has been observed in areas with a high intensity of fishing with bottom-tending gears and may be a long-term consequence of the resuspension and settling of sediments following fishing events (BSH 2009a). Generally, effects in more dynamic habitats such as unconsolidated sediments in shallow waters are less severe than those occurring in structurally complex habitats (e.g. seagrass meadows, biogenic reefs) and habitats relatively undisturbed by natural perturbations (Kaiser et al. 2002).

The degree of mechanical disturbance of the seafloor has been observed to differ apart from sediment properties and natural disturbances also due to the fishing gear used. While otter trawling creates irregular features in the form of furrows on the sea-bed, beam trawling mainly leads to a flattening of bottom topography. The net opening of an otter trawler is maintained by trawl doors which cause furrows generally ranging from 1 to 5 cm but may reach up to 20 cm deep depending on the door weight and the substrate properties. Trawl door marks may disappear after several months in highly dynamic ecosystems but may also last up to five years in sheltered areas (FAO 2004). Additionally, large amounts of sediment are resuspended during otter trawling (ICES 2003).

Beam trawlers are equipped with tickler chains which are specifically designed to disturb the seabed surface along the whole width of the gear and penetrate the upper few centimetres of the sediment. The width of a beam trawl ranges from 4 to 12 m. Observations on the persistence of beam trawl marks range from tracks disappeared after a few days in tidally exposed areas to several months or more in sheltered areas (FAO 2004).

Offshore wind farms

The production of offshore wind energy is currently one of the most important in terms of area utilisation, especially in the EEZ of the North Sea. At present 28 wind farms are authorised in the North Sea and three in the Baltic Sea. With the test field 'alpha ventus' and 'Bard Offshore 1'in the North Sea and 'Baltic 1' in the territorial waters of the Baltic Sea three wind farms are already in operation. Several others are currently under construction in the German North Sea (BSH 2013a). Applications for many more wind farms are being assessed by the regulatory authorities. In the EEZ of the North Sea, the offshore wind farms approved and applied for so far will occupy an area of more than 15% of the total surface area (Ammermann 2011).

An offshore wind farm generally comprises of different components affecting the sea-bed: foundations of the piles (e.g. monopiles, tripods or gravity base) and the converter platform, power cables to connect the piles and the converter platform and scour protection in form of rock or concrete mattresses. Additional installations such as substations may be needed (OSPAR 2006).

Physical impacts on the seabed arise from the construction phase and the physical presence of the installations. Construction works in form of dredging activities, piling or drilling and cable-laying operations will disturb the seafloor by mobilising sediments and temporary causing increased turbidity. The permanent submarine installations are accompanied by a loss of marine soft-bottom habitats due to the introduction of artificial hard substrates. In dynamic ecosystems scouring may impact an additional area (OSPAR 2006). Usually scour pits can be considered to be limited to within ten times the diameter of the obstacle. Cumulative effects of scouring around piles could not be observed. If scour protection is applied, materials are placed around the tower in a radius of around 25 m (Meissner & Sordyl 2006). Furthermore, the erection of a wind farm may influence local hydrographical regime and sediment transport processes (OSPAR 2006).

Other permanent offshore installations

Other offshore installations mean structures beside those erected in relation to offshore wind farms. These contain platforms for the exploitation of gas and marine research. The extraction of gas is carried out at present only to a small extent in the German North Sea with one active gas rig located near the Dogger Bank. Additionally, two gas compressor platforms and several research platforms are operational in the North and Baltic Seas. Further research platforms are planned (BSH 2013b).

The main pressures affecting the seabed arise from the placement and physical presence of sub-marine structures. Effects on the marine environment are comparable to those exerted by wind farms and mainly include permanent habitat alterations by the foundation of structures and scour protection as well as temporary effects of construction works.

Cables and pipelines

Currently the German North Sea is crossed by six gas pipelines connecting gas rigs to each other and the mainland. With the Nord Stream pipeline there exists at present one gas pipeline in the Baltic Sea, two more are at the planning stage. Pipelines are usually laid directly on the sea floor without further coverage. Especially in shallow waters the pipeline may be placed in a trench to ensure its stability and mechanical protection. In this case a trench is dug where the pipeline is laid in and afterwards the trench will be filled back. Alternatively, the pipeline can be secured by concrete mats or gravel. Usually pipelines are enclosed by a concrete casing and have a diameter of approximately 1.2 m (Herberg et al. 2007).

In addition to the gas pipelines a series of submarine cables is planned or already exists. Cables can be distinguished into data or telecommunication cables and power cables. In the German North Sea there are currently eight data cables in operation, in the Baltic Sea seven. With the NorNed cable between the Netherlands and Norway currently only one transit power cable in the German North Sea is in operation. In the Baltic Sea two transit power cables exist which connect Germany with Denmark and Sweden. Additionally, in the North Sea the first high-voltage power cable to link offshore wind farms with the coast is already in operation, many more will be established in the near future (BSH 2013b).

Submarine cables are usually placed in a depth of approximately 1 m in the sea floor. Where cables cannot be buried, e.g. in areas of exposed bedrock or at intersections with other cables or pipelines, they are laid directly on the seabed and may be covered by a protective structure like rock armour (OSPAR 2008).

The installation of cables and pipelines results in physical disturbance of the seabed and associated impacts such as damage or displacement of benthic organisms, increased turbidity and alteration of sediment properties. The presence of pipelines, cables if not buried and protection structures represents the introduction of artificial hard substrates in prevalent soft-bottom habitats. Near-bottom currents may be influenced by pipelines or protection structures and thus alter sediment characteristics (OSPAR 2008). The footprint of cables and pipelines is dependent on the length, diameter and whether or not it is trenched.

Extraction of sand and gravel

In the German North Sea there are currently four areas licensed for the extraction of sand and gravel with a total area of 1.350 km². The area currently in use accounts for approximately 250 km² (BMU 2008). Large parts of the extraction sites are located in Natura 2000 sites where priority habitats such as reefs and permanently submerged sandbanks are present (BSH 2013b). In the North Sea, the area actually extracted in 2005 was 2.8 km², while in 2006 the area was extended to 6.6 km² (Schroeder et al. 2008). In the EEZ of the Baltic Sea currently no aggregate extraction takes place. The amount of material extracted in the North Sea varied between 1.4 and 36.2 million tonnes in the years 2005 to 2009. The maximum values resulted from the construction of the Jade-Weser Port in 2008/09 and were taken from the territorial waters (BLMP 2012).

Extraction takes place by means of suction dredging either with the vessel remaining stationary or while driving. In Germany, most aggregate dredging is carried out by trailer suction dredging. This creates a series of longitudinal tracks, generally 2-3 m wide and up to 50 cm deep, as the drag head passes over the seabed. Sediment is mobilised and brought into suspension as the drag head disturbs the sediment surface and with the overflow of excess water back in the water column. Sometimes screening of the sediment takes place while dredging, e.g. particles of a certain size, mostly fine sand, are sorted out and returned to the water (Hill et al. 2011).

Anchor dredging, where the vessel remains stationary to extract deep deposits, is less common. In this way rounded pits of around 10 m depth and with a diameter of 10-50 m are produced. Although the disturbed area is much smaller compared to trailer dredging, morphological changes are much more severe (Hill et al. 2011).

The main impacts on the physical environment caused by aggregate extraction are alterations of the seabed topography, changes in sediment composition and mobilisation of particulate matter. High intensity dredging may result in a strongly disturbed topography with deep tracks and furrows remaining for several years (ICES 2009). A lowering of the seabed by up to 2-3 m may be a con-sequence of repeated dredging in the same area. Such changes in seabed topography may in turn lead to an altered hydrodynamic and sedimentation regime. Extraction sites are often characterised by a higher proportion of sediments with a small grain size. Changes in sediment composition may be caused by screening when finer particles are returned to the seabed, by overspill of water containing small sand particles or by the infilling of dredge tracks and furrows. Increased turbidity plumes of suspended material generate from the dredging activity on the seafloor, the overflow and screening, thereby extending the area subject to changes in sediment composition. Depending on local conditions and extraction method sediments may as well become coarser, e.g. by selective extraction of sand or when gravel deposits are being exposed beneath the surface layer of the seabed. The footprint of aggregate extraction activities can be assumed to cover an area of up to 2-3 km around the extraction site, depending on sediment type (ICES 2009, Hill et al 2011).

Table 3-2: Summary of human activities and associated physical pressures on sea floor integrity in the German EEZ.

Activity	Geographic distribution	Pressure	Description of pressure
Bottom trawling	throughout the North and Baltic Sea	abrasion	alteration of seabed topography: reduction in habitat complexity, changes in sediment characteristics, removal of physical and biological structures
Offshore wind farms	three wind farms in operation in Baltic and North Sea, 28 authorised, around 90 planned, mostly in the North Sea	sealing	physical presence of foundation: loss of marine habitat
		smothering	physical presence of structures (foundations, scour protection), introduction of artificial hard substrate, scouring
		changes in siltation	during construction: increased turbidity, resuspension of sediments, in operation: changes in sediment transport
Other permanent offshore installations	one gas rig in the North Sea, several research and gas compressor platforms in the North and Baltic Seas	sealing	physical presence of foundation: loss of marine habitat
		smothering	physical presence of structures (foundations, scour protection), introduction of artificial hard substrate, scouring
		changes in siltation	during construction: increased turbidity, resuspension of sediments, in operation: changes in sediment transport
Pipelines	six gas pipelines in the North Sea	smothering	physical presence of pipeline when not trenched, protection structures, introduction of artificial hard substrate
		changes in siltation	during pipeline-laying: increased turbidity, resuspension of sediments, in operation: changes in sediment transport when pipeline is not trenched
Cables	several telecommunication and power cables in the North and Baltic Sea, more power cables planned for wind farms	smothering	physical presence when cable is not buried, in the case of protection structures: physical presence, introduction of artificial hard substrate
		changes in siltation	burial of cable: increased turbidity, resuspension of sediments
Extraction of sand and gravel	several areas licensed for extraction in the North Sea	smothering	changes in sediment composition (sediments mostly become finer)
		selective extraction	removal of substrate, altered seabed topography: presence of tracks and furrows, lowering of seabed
		changes in siltation	during dredging activity: increased turbidity, resuspension of sediments

3.4.2.2 Spatial and temporal extent of pressures

Assessing the intensity of pressures involves information on both the spatial and temporal footprint of the related activities. Determination of the spatial extent should include data on the precise location of activities, e.g. the site of an offshore wind farm, combined with information on the area affected like the extent to which seabed is disturbed by smothering around a pile foundation.

Estimates for the spatial extent of pressures given in this section are based on literature describing the area subject to physical disturbance (e.g. Eastwood et al. 2007, de Vries et al. 2011, DEFRA 2012).

Sealing

Offshore wind farms: Sealing by offshore wind farms results from the placement of foundations for the wind turbines. Specifications for the base diameters of the different foundations show some variation. Monopiles have been chosen for most of the installed offshore wind farms to date. In OSPAR (2006) the diameter of a monopile is set at 4 to 6 m with the indication that towers of 5 m appear to be the dominant size. Approvals for the offshore wind farms planned in Germany usually estimate a diameter of 5 m for the area sealed by monopiles, some wind farms designs may possess even larger monopiles with 6 m diameter (e.g. wind farm 'Innogy') (BSH 2013a). Tripods (three legs) and jackets (four legs) are anchored by driven or drilled piles, typically ranging from 0.8 to 2.5 m in diameter. These types of foundations are used with larger turbines and may be located in deeper waters (EWEA 2009). Gravity based structures have also been used on several projects. Information on the diameter of gravity-based foundations varies from 15 m (Meissner & Sordyl 2006) to 30 m (OSPAR 2006). Gravity based structures may also vary in shape, they may be circular or rectangular. Based on this data, to estimate the spatial footprint of sealing caused by the foundations of wind turbines the area of 20 m² per foundation is suggested. This would correspond to a monopile with a diameter of 5 m or a jacket with piles of 2.5 m. Tripods will generally have a smaller footprint while gravity-based structures are usually significantly larger.

Other permanent offshore installations: Platforms for the extraction of oil and gas are usually founded on jacket structures with four or six piles. Research platforms may be jackets such as FINO 1 or monopiles like FINO 2 and 3. The average area sealed by different types of platforms is estimated at 15 m² as the general footprint of jackets and monopiles used for research platforms is believed to be smaller than for the foundations of wind turbines.

Smothering

Offshore wind farms / other permanent offshore installations: Scour protection is applied around monopiles and gravity-based foundations and usually has a radius of 10 m for monopiles. Recent studies on scour development of off-shore wind farms indicate the effects of scour are locally restricted to the near vicinity of the piles (Orejas et al. 2005, Meissner & Sordyl 2006). Changes in sediment dynamics around 'alpha ventus' have been observed in a maximum distance of 60 m from the structure (Lambers-Huesmann & Zeiler 2011). Surveys at wind farms in the UK found scour pits around individual monopile foundations in highly mobile sediments developed to 100 m in diameter while at other, more stable sites scour pits reached only a diameter of 10 m (CEFAS 2006, DECC 2008). According to METOC Plc (2000), the area around a structure prone to local scour is usually expected to be approximately ten times the diameter of the structure. Around the FINO research platforms changes in sediment structure could be observed up to around 40 m in the direction of the main current (Orejas et al. 2005). Buffers used for the spatial assessment of offshore wind farms range from a diameter of 50 m (DEFRA 2012) to 100 m (Eastwood et al. 2007). Based on the studies conducted in the German offshore area and due to the fact that changes in sediment properties do not occur circular around a pile, a diameter of 50 m or an approximate area of 2000 m² is proposed for the spatial footprint of offshore wind turbines and other platforms. Physical loss (sealing and smothering) caused by an offshore wind farm with 80 turbines would thus add up to the total area loss of 0.16 km² (average size of an offshore wind farm: 40-50 km²).

Cables and pipelines: Some of the pipelines in German waters are completely trenched (e.g. the pipeline connecting the gas rig in the EEZ with the Dutch NOGAT pipeline) or at least in shallow waters. Furthermore, pipelines laid on mobile sediments may bury themselves and thus will not exert any pressure on the seabed (BSH 2009a). Therefore, it is proposed to estimate the spatial footprint of pipelines by the mean diameter of 1.2 m and only in the EEZ where pipelines are usually not buried in sediment. The large majority of cables are buried, so that impacts are short-term only during construction. Physical loss occurs when the cable has to be protected by rock armour at locations with hard substrate or at intersections. The area thus altered is believed to be negligibly small and may not be properly assessed with the available geospatial data.

Extraction of sand and gravel: The effect of aggregate extraction on habitat structures depends on the method and intensity of dredging, the level of screening and sediment type (Hill et al. 2011). The pressure associated with extraction of sand and gravel could thus be ‘smothering’ (sediment composition and consequently habitat type changes) or ‘selective extraction’ (exposed sediment is of the same type). It is assumed that based on national legislation and by means of Environmental Impact Assessments significant changes of habitat types by dredging are prevented. Therefore, the pressure associated with aggregate extraction is proposed to be ‘selective extraction’.

Selective extraction

Extraction of sand and gravel: In the UK waters mineral mining activities are routinely monitored by an electronic monitoring system which automatically records at 30 s intervals. Dredging locations are then spatially aggregated into 50 x 50 m blocks and categorised from low to high intensity which is expressed as hours dredged (Eastwood et al. 2007). These data can be used to represent the direct spatial extent of aggregate dredging (Eastwood et al. 2007, DEFRA 2010). It would be essential to obtain equally exact data for the German areas licensed for the extraction of sand and gravel, otherwise the spatial footprint of extraction cannot be assessed.

Abrasion

Bottom trawling: The most reliable source of positional data for fishing vessels and the one with the highest resolution is the EC vessel monitoring system (VMS). Since January 2012 this includes all vessels in excess of 12 m operating in European waters. Resolution and accuracy obtained by VMS data far exceed that of the ICES rectangle-based data formerly used to provide information on spatial and temporal trends in fishing effort (Lee et al. 2010).

Several methods have been developed and applied to estimate the spatial footprint of fishing effort. Main differences are the distinction between fishing and steaming according to the recorded speed and the method of converting VMS data points to an area describing fishing effort. The Bundesanstalt für Landwirtschaft und Ernährung (BLE) provides data processed by the von-Thünen institute in Hamburg so there is little influence on the methods used for calculating the spatial extent of fishing effort.

Changes in siltation

Offshore wind farms: Under the pressure 'changes in siltation' all impacts occurring during construction activities are subsumed. These include the disturbance, resuspension, erosion and accumulation of sediments caused by cable laying and foundation installation as well as by ship movement and anchoring. The extent of these activities is mostly very localized and depends on sediment type, grain size distribution and the hydrodynamic regime and thus can vary greatly between sites (OSPAR 2006). Even though the individual impacts are small-scale, it is proposed to define the wind farm area as a whole as impacted by changes in siltation, similar to the approach by HELCOM (2012).

Other permanent offshore installations: The construction of platforms for the exploration of oil or gas or for research purposes involves disturbances of the seabed as described above for wind farms. The extent of impacts is near-field and largely site- and project-specific. As a generalisation it is suggested to attribute a buffer of 100 m around the installation as the area impacted by construction activities.

Cables and pipelines: Laying of cables and pipelines leads to seabed disturbance and associated impacts of increased turbidity and alteration of sediments. The area affected by sediment plumes and smothering is generally limited to the near-field area along the construction corridor and depends on the method and device used and the amount of excavated and dumped sediment. Direct disturbance of the seabed occurs within 1-2 m on both sides of the trench. Impact modelling observed sediment deposition in a maximally 90-120 m wide cable corridor. Water quality effects may be noticed as far as 1 km; however, it is assumed that suspended sediment concentrations which occur during cable burial do not exceed naturally induced turbidity by tides, waves or currents (OSPAR 2008). Thus, it is proposed to calculate with a buffer of 100 m for the placement of cables and pipelines.

Extraction of sand and gravel: Increased turbidity due to sediment plumes can be detected in an area of up to 3 km around the extraction site, depending on sediment properties (ICES 2009, Hill et al. 2011). The dispersal of suspended material can be estimated by using particle transport models (Eastwood et al. 2007). However, as a uniform particle size distribution is assumed across all sites, it is believed that this model simulates a precision which may be misleading. HELCOM (2012) adds a buffer of 2000 m to the geospatial data on extraction sites which seems to be an appropriate mean value for the accumulation of fine sediments. If data on the exact location of the extracted area could be obtained, it is therefore suggested to apply a buffer of 2000 m around the extraction site to cover changes in siltation.

Table 3-3: Spatial considerations for intensity of pressures occurring in the German EEZ.

Pressure	Activity	Spatial footprint
Sealing	offshore wind farms	average size foundation = 20 m²
	other permanent offshore installations	average size of foundation = 15 m²
Smothering	offshore wind farms, other permanent offshore installations	average size of substrate alteration around foundation = 2000 m²
	surface pipelines	length of pipeline with diameter of 1.2 m
Selective extraction	extraction of sand and gravel	actual dredged area
Abrasion	bottom trawling	grid with fishing activity (0/1) for each cell (VMS cells or ICES rectangle)
Changes in siltation	construction of offshore wind farm	area of offshore wind farm
	construction of other permanent offshore installations	100 m around installation
	construction of cables and pipelines	100 m wide corridor
	extraction of sand and gravel	2000 m around actual area dredged

The temporal extent describes the frequency or duration of a pressure, e.g. the number of trawling events per year. A classification of five categories is applied for each pressure, ranging from rare to persistent (Table 3-4). The scale is based on expert judgement and should reflect the actual frequency of pressures in the German EEZ. A pressure occurring more than three times per year is assumed to be persistent. For example, even in more tolerant habitats like sublittoral sand, bottom trawling four times a year results in a permanent disturbance without the possibility of recovery. Pressures associated with physical loss (sealing and smothering) are persistent and can therefore only be allocated to the highest category, which is equivalent to the highest intensity possible for physical damage. The reporting period is chosen as reference period since this is assumed to aid in reflecting effects of management measures.

Table 3-4: Scale for temporal extent of physical pressures.

Rank	Definition
rare	1 event per reporting period
occasional	> 1 -< 6 events per reporting period
regular	1 event per year
frequent	> 1-3 events per year
persistent	> 3 events per year / permanent installation

Data on distribution and temporal extent of physical pressures are used to create pressure maps by means of GIS layers. The spatial scale is dependent on the nature of data available for the assessment. Data with the highest possible resolution are preferred, e.g. for fisheries information from the EC vessel monitoring system (VMS). Each physical pressure is displayed on a separate map. Figure 3-1 summarises the necessary components and steps for the generation of pressure maps.

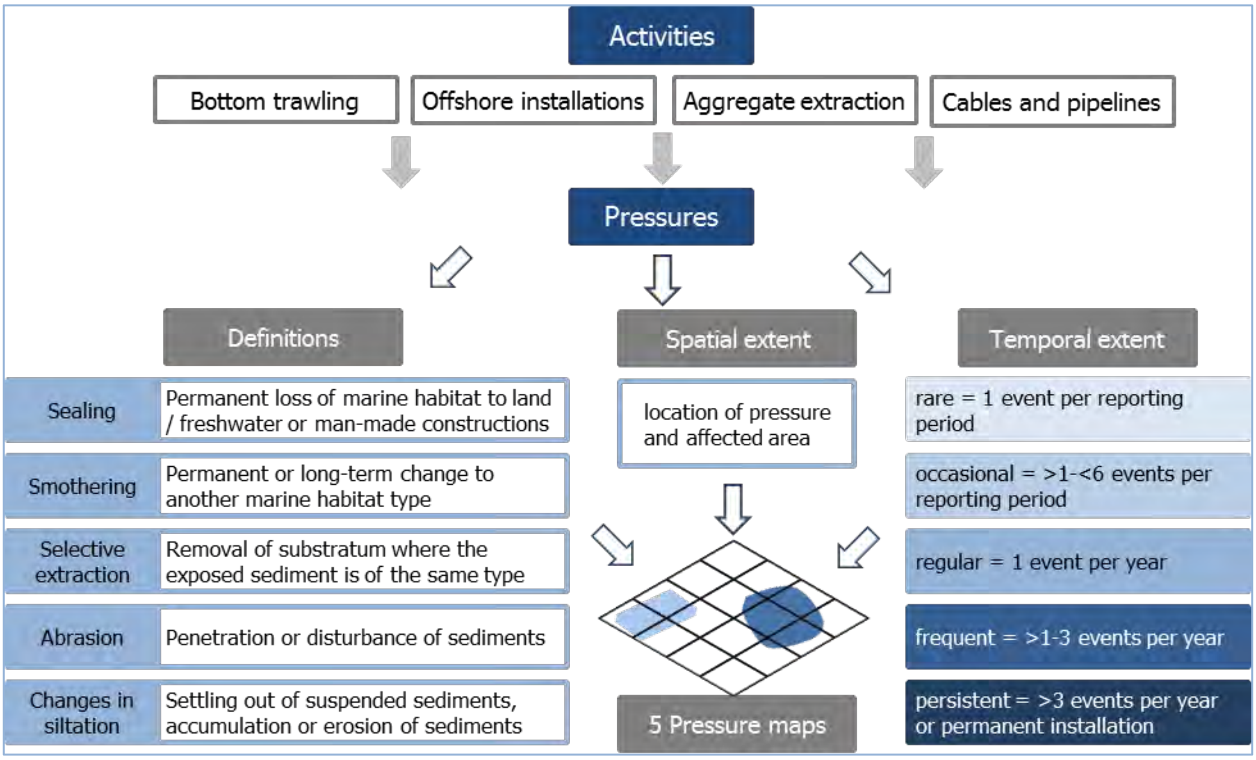

Figure 3-1: Human activities in the German EEZ and characterisation of associated pressures (own illustration).

3.4.3 Benthic habitats

3.4.3.1 Definition of habitat types

Annex III, table 1 of the MSFD provides an indicative list of habitat types:

- Predominant habitat types - The predominant seabed and water column habitat type(s) with a description of the characteristic physical and chemical features, such as depth, water temperature regime, currents and other water movements, salinity, structure and substrata composition of the seabed,
- Special habitat types - Identification and mapping of special habitat types, especially those recognised or identified under Community legislation (the Habitats Directive and the Birds Directive) or international conventions as being of special scientific or biodiversity interest,
- Habitat types meriting special reference - Habitats in areas which by virtue of their characteristics, location or strategic importance merit a particular reference. This may include areas subject to intense or specific pressures or areas which merit a specific protection regime.

Predominant habitats

The Commission Staff Working Paper (EC 2011) provides further instructions on definitions of habitat types. Predominant seabed habitat types are closely linked to level 3 of the EUNIS habitat classification scheme.

Habitats are classified according to their depth (littoral, shallow, shelf, bathyal and abyssal) and their substrate. Substrates are differentiated into rock and biogenic reef and sediment habitats (coarse, sand, mud, mixed). Sublittoral sediments in the German EEZ of the North Sea classified according to EUNIS level 3 are as follows:

- A5.1 Sublittoral coarse sediment
- A5.2 Sublittoral sand
- A5.3 Sublittoral mud
- A5.4 Sublittoral mixed sediments

Special habitats

Special or listed habitat types refer to those identified under several regulatory frameworks such as the EU legislation or international conventions (EC 2011). Habitat types in German waters belonging to this category are therefore priority habitats of the Habitats Directive, protected biotopes according to § 30 BNatSchG (Federal Nature Conservation Act), the OSPAR list of threatened and/or declining species and habitats and the HELCOM red list of marine and coastal biotopes and biotope complexes. The following set of habitat types occurs in German coastal and marine waters:

- seagrass beds
- macrophyte meadows and beds
- *Mytilus edulis* beds
- sea-pen and burrowing megafauna communities
- *Sabellaria spinulosa* reefs
- shell gravel bottoms
- gravel bottoms with *Ophelia* species
- species-rich habitats on coarse sands, gravel or shell debris
- reefs
- sandbanks

Habitats in particular areas

Habitats in particular areas can include areas subject to specific or multiple pressures and are therefore likely to entail risks to marine biodiversity, marine ecosystems, human health or legitimate uses of the sea, or areas already designated or which should be designated due to various forms of spatial and management protection. Currently, particular habitats have neither been identified by the European Commission nor by the Regional Seas Conventions. In order to be consistent with other national environmental policies and to account for the ecological importance of protected areas it is proposed to consider these as habitats in particular areas on a national basis. With regard to benthic habitats this would be the designated Natura 2000 sites in the North and Baltic Seas. Habitats in particular areas will not be separately assessed, as Natura 2000 sites consist of both special and predominant habitats. Instead of that, a specific GES target should be proposed for these particular habitats in order to intensify national efforts for conservation of designated sites (see chapter 3.7).

3.4.3.2 Sublittoral habitats in the German EEZ of the North Sea

According to Figge (1981), sediments in the German Bight are classified in several major areas: The Pleistocene Elbe valley, extending from the inner German Bight to the east of the Doggerbank, and the bordering plains west of this valley are characterized by fine sands with noticeable contents of silt and clay (5-50 %) and a comparatively even relief. Sediments of the Borkum Reef Ground west of the Pleistocene Elbe valley are more heterogeneous. The predominant medium and coarse sands are interspersed with gravel and small stones. With increasing water depth sediments change to medium and fine sands with a silt fraction of up to 10 %. The area east of the Pleistocene Elbe valley (Sylter Outer Reef, Amrum Outer Ground) is marked by a conspicuously heterogeneous distribution of marine sediments. Between typical relict sediments with coarse sands, gravel and stones fine and medium sands accumulate. The density of stones is generally higher compared to the Borkum Reef Ground. The predominant sediments of the Doggerbank are fine sands, partly mixed with shell debris and a minor fraction of silt and clay (BSH 2009a).

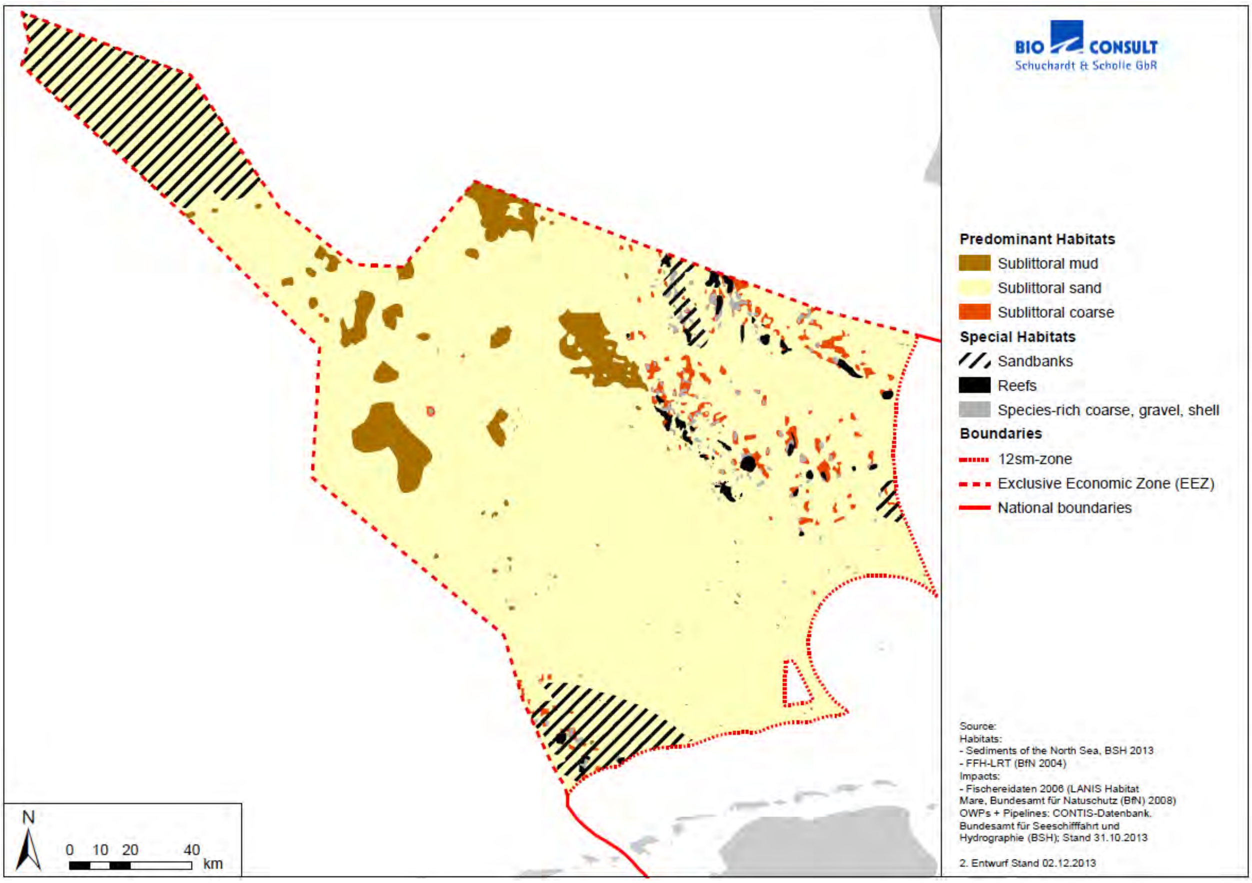

Figure 3-2: Distribution of predominant and special habitat types in the German EEZ of the North Sea.

Figure 3-2 shows the distribution of sediments in the German EEZ and the Natura 2000 habitat types. Based on this map, the following sublittoral predominant and special habitats have been identified in the German North Sea:

Predominant habitat types:

- Sublittoral sand
  This habitat type is widely distributed in the German Bight and contains all sediments with fine and muddy sands (silt and clay < 20 %).
- Sublittoral mud
  Larger areas of fine and sandy mud can be found in the central part of the German EEZ. The fraction of silt and clay exceeds 20 %.
- Sublittoral coarse sediment
  Small areas in the Borkum Reef Ground, Amrum Outer Ground and Sylter Outer Reef with medium to coarse sands.

Special habitat types:

- Reefs
  Geogenic reefs as defined by the Interpretation Manual of European Union Habitats (EC 2013). Biogenic reefs have not yet been designated in the German North Sea.
- Sandbanks
  Sandbanks in the German EEZ are located at the Doggerbank, the Borkum Reef Ground and in the Sylter Outer Reef. They are defined according to the Interpretation Manual of European Union Habitats (EC 2013).
- Species-rich habitats on coarse sands, gravel or shell debris
  Small areas with mixed or unmixed sediments of coarse sands, gravel and shell debris.

3.4.3.3 Assessment of habitat sensitivity

For a particular pressure to have an impact on a habitat or community, these have to demonstrate a level of sensitivity to that pressure. In principle, sensitivity of ecosystem components is determined by two aspects: the ability to withstand disturbance or stress (resistance or tolerance) and the ability and time needed to recover from a perturbation and return to the previous state (resilience or recoverability). Highly sensitive species or habitats are therefore those which possess both low resistance and resilience (Environment Agency 2010). Basically, approaches to assess resistance and recovery time of habitats either rely on experts to allocate sensitivity categories to habitats based on given criteria (e.g. Halpern 2008, OSPAR 2009, HELCOM 2010, Andersen et al. 2011) or refer to evidence base or biological traits of selected species (e.g. McMath et al. 2000, Tyler-Walters et al. 2001). Expert judgement is however also required to choose species which are considered to be characteristic or important for the structure and function of the habitat. A more holistic approach which not only takes account of species sensitivities but also physico-chemical features such as substrate characteristics is delivered by Tillin et al. (2010). As there is a large number of existing approaches for assessing habitat sensitivities which are already approved and accepted, it is proposed to refer to the knowledge of these previous studies. Particular focus has been given to the MarLIN approach (Tyler-Walters et al. 2001), as it is mainly based on available evidence and includes a large database on benthic species features. The expanded concept by Tillin et al. (2010) has also been especially considered in the following suggestions for the sensitivity assessment of predominant and special habitats.

In general, sensitivity assessments focus mainly on the biological components of habitats. As benthic species play a crucial role in creating physical structures of the habitat (e.g. burrows or pits), it is considered that faunal sensitivity has to be a major part of any assessment of sensitivity to morphological impacts. At the same time the impact on the physical habitat, the modification following disturbance and the ability to recover from damage is regarded as important in order to assess sensitivity of the habitat as a whole. If habitat suitability is affected by the pressure, then recovery of the benthic community may not take place or may be delayed.

The proposed approach to define the sensitivity of habitats combines the assessment of both habitat structure and important species. In principal, the sensitivity assessment is closely related to the MarLIN approach described by Tyler-Walters et al. (2001): assessment of the resistance of a habitat or species in relation to a defined intensity of each pressure, assessment of the recoverability of the habitat or species and the combination of resistance and recoverability to derive an overall sensitivity rank for the particular habitat or species in relation to each pressure.

Habitat sensitivity by Tyler-Walters et al. (2001) is the result of the individual sensitivities of key or characteristic species. In the concept presented here an additional step is included: the physical impact on habitats is assessed with regard to the resistance in relation to a specific pressure and the recoverability following the disturbance. This sensitivity of physical habitat properties is combined with the sensitivities derived for representative species to obtain an overall sensitivity rank for the habitat. For the assessment of habitat sensitivity, it is assumed that habitats are in an optimum reference state, i.e. habitat alterations due to previous anthropogenic activities are not considered.

Selection of characteristic species for the sensitivity assessment

Characteristic species used in the sensitivity assessment should be species which significantly influence the ecology of a particular habitat type. These could be species which provide a distinct habitat that supports an associated community, or are important for community functioning by interactions with other species, or species which are used for the definition of a habitat. The loss or degradation of one of these species would severely affect the viability, structure and function of the habitat and may result in the loss of the habitat or a changed classification. For example, the loss of *Sabellaria spinulosa* would lead to the loss of the habitat '*Sabellaria* reef'. The sea urchin *Echinus esculentus* is important for structure and function in geogenic reef communities due to its grazing activities. Most of the characteristic species used for the assessment are those that aid to classify a habitat type. As far as available, the characteristic species identified by Rachor & Nehmer (2003) for the classification of benthic communities in the south-eastern North Sea were adopted (see also chapter 3.9). The criteria applied by Rachor & Nehmer (2003, see also Rachor 2007) for the selection of characteristic species include dominance, presence, faithfulness in dominance and abundance and the contribution of discriminating species in a dissimilarity analysis. Other sources for the selection of characteristic species were Nehls et al. (2008) and BfN (2011). The selection of characteristic species for the sensitivity assessment is assumed as a preliminarily approach for the initial application of the methodology. For future assessments it is proposed to mainly refer to results of an ongoing habitat mapping project, which also should provide information on characteristic species of benthic habitats or to use the description of the reference state.

Resistance of the physical habitat and characteristic species

The resistance or tolerance of the physical properties and the characteristic species of a habitat should reflect the susceptibility to damage or loss as a result of a pressure on the seabed. The likely tolerance of the species or habitat is estimated with respect to a specified magnitude and duration of change in order to provide a standard level against which to assess resistance. Benchmarks for physical disturbance indicated by Tyler-Walters et al. (2001) largely correspond to the pressure definitions given in Table 3-1. The following definitions are used by the MarLIN approach and are adopted for the indicator concept:

- Substratum loss (= selective extraction): All of substratum occupied by the species or biotope under consideration is removed. A single event is assumed for sensitivity assessment. Once the activity or event has stopped (or between regular events) suitable substratum remains or is deposited. Species or community recovery assumes that the substratum within the habitat preferences of the original species or community is present.
- Physical disturbance or abrasion (= abrasion): Force equivalent to a standard scallop dredge landing on or being dragged across the seabed. A single event is assumed for assessment.
- Smothering (= changes in siltation): All of the population of a species or an area of a biotope is smothered by sediment to a depth of 5 cm above the substratum for one month.

The resistance of the physical habitat and the characteristic species is classified in four ranks, based on tolerance scales by Tyler-Walters et al. (2001) and IOW (2009) (Table 3-5).

Table 3-5: Scale for resistance of the physical habitat and characteristic species (adapted from Tyler-Walters et al. 2001).

Rank	Physical habitat	Characteristic species
low	Structure and function of physical habitat characteristics are altered completely or to a large extent.	The species population is likely to be killed / destroyed by single event of anthropogenic pressure.
intermediate	Significant alterations of physical habitat characteristics; essential structure and function are maintained.	Some individuals of a species population may be killed / destroyed by single event and the viability of a species population will be reduced.
high	Minor alterations of physical seabed characteristics, low impact on structure and function.	A species population is unlikely to be killed / destroyed by single event. However, the viability of a species population will be reduced.
tolerant	No negative effect detectable or positive effects on structure and function of physical habitat characteristics.	No negative effect detectable or positive effects on survival or viability of a species.

Recoverability of the physical habitat and characteristic species

Recoverability describes the ability of a habitat or species population to restore from damage sustained as a result of a physical impact on the seabed. Recoverability of organisms is especially dependent on the ability of the species to regenerate, regrow, recruit or recolonize and the extent of damage incurred. Recovery is only possible when the impact has stopped or has been removed.

Information on the potential impact of physical disturbance and the response of specific habitats and species is based on available evidence or expert judgement. Precedence is given to direct evidence of impacts such as information from targeted studies or experiments that looked at the effect of the specific factor on the habitat, the species or similar species. As a main source for the assessment of species resistance and recoverability, the MarLIN web site (MarLIN 2013) was used that provides detailed information on the sensitivity of selected species. Where information on characteristic species is not available, the relevant biological traits are inferred from similar species or congeners. As an additional source of information on species beside the MarLIN web site serves the 'Genus Trait Handbook' (MES 2008) or similar references. Tyler-Walter et al. (2001) also present simple decision trees to aid the resistance and recoverability assessment based on the available key information for the species like mobility, environmental position or reproductive biology. These decision trees provide a systematic and transparent approach to assessment and are described in full by Tyler-Walters et al. (2001).

The recoverability of the physical habitat or species is assessed against a five-step scale which has been adopted from Tyler-Walters et al. (2001) and Tillin et al. (2010) (Table 3-6).

Table 3-6: Scale for recoverability of the physical habitat and characteristic species (adapted from Tyler-Walters et al. 2001).

Rank	Definition
very low	full recovery not possible or will take over 25 years
low	full recovery within 10-25 years
moderate	full recovery within 2-10 years
high	full recovery within 1-2 years
very high	full recovery within 1 year

Overall habitat sensitivity

A decision matrix is used to automate the combination of intolerance and recoverability and to obtain sensitivity categories for the physical habitat and the characteristic species. The matrix has been adapted from Tyler-Walters et al. (2001) (Table 3-7).

Table 3-7: Matrix for the sensitivity of the physical habitat and characteristic species (adapted from Tyler-Walters et al. 2001).

<table>
<tr><th colspan="2" rowspan="2">Sensitivity</th><th colspan="5">Recoverability</th></tr>
<tr><th>very low
(>25 yr.)</th><th>low
(>10-25 yr.)</th><th>moderate
(>2-10 yr.)</th><th>high
(1-2 yr.)</th><th>very high
(<1 yr.)</th></tr>
<tr><td rowspan="4">Resistance</td><td>low</td><td>very high</td><td>high</td><td>intermediate</td><td>intermediate</td><td>low</td></tr>
<tr><td>intermediate</td><td>high</td><td>high</td><td>intermediate</td><td>low</td><td>low</td></tr>
<tr><td>high</td><td>intermediate</td><td>intermediate</td><td>low</td><td>low</td><td>very low</td></tr>
<tr><td>tolerant</td><td>not sensitive</td><td>not sensitive</td><td>not sensitive</td><td>not sensitive</td><td>not sensitive</td></tr>
</table>

The overall sensitivity is derived from the sensitivity ranks of the physical habitat and the sensitivity of characteristic species. The highest (i.e. most sensitive) rank assigned to either habitat structure or species determines the overall habitat sensitivity. For example, if the habitat structure is judged to have an intermediate sensitivity but the characteristic species are highly sensitive, then the overall sensitivity of the habitat is reported as high. Figure 3-3 illustrates the methodology to assess habitat sensitivity and to generate pressure-specific sensitivity maps.

The physical pressures ‘smothering’ and ‘sealing’ are defined by a loss of substratum and therefore a loss of the habitat is implied. The habitat is not expected to recover unless the area is actively restored or any permanent structures are removed. Sealing and smothering are in addition associated with an impact which destroys habitat structures as well as benthic organisms. Therefore, resistance is classified as low and recoverability as very low (>25 years) which means that all habitats are ranked as possessing a very high sensitivity towards the pressures sealing and smothering.

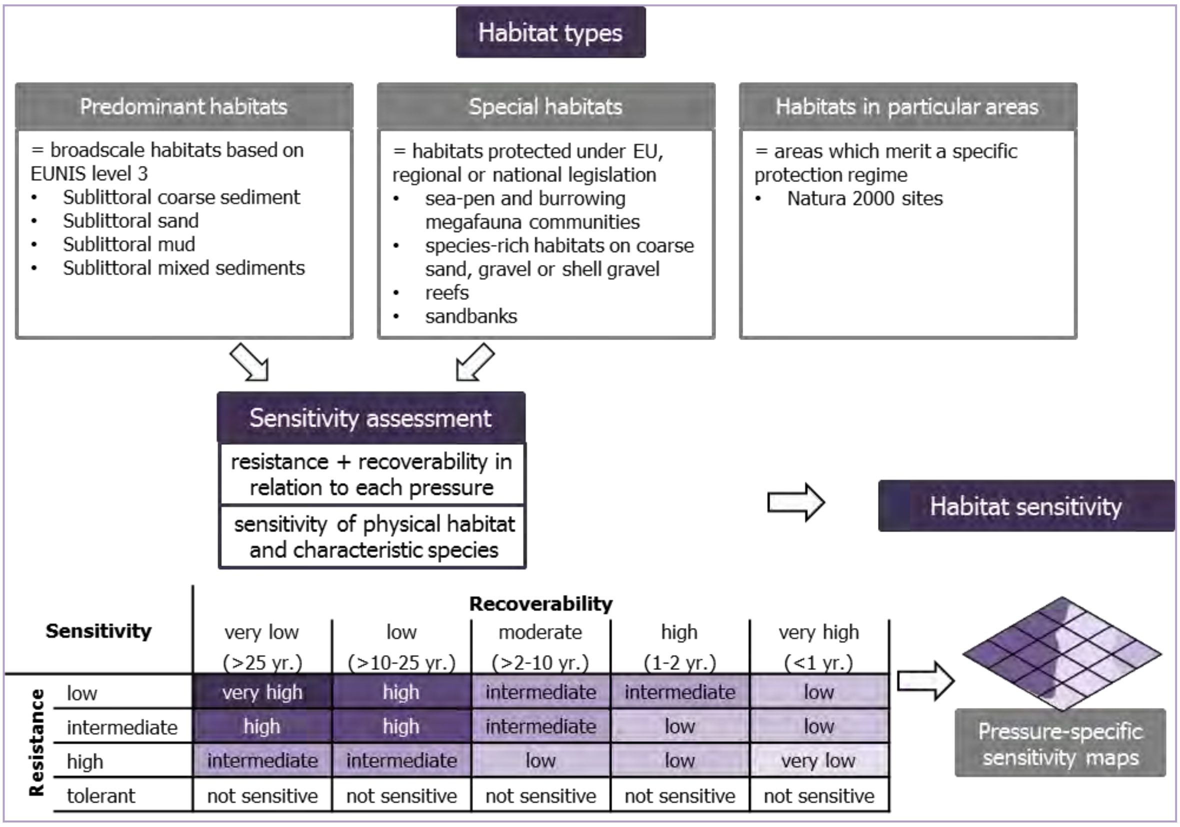

Figure 3-3: Sensitivity assessment of benthic habitats (own illustration).

In order to assess the proportion of benthic habitats perturbed by human activities, data on the distribution and extent of predominant and special habitats is required. For the application of the methodology to the German EEZ, a map with sediment distribution and the Natura 2000 habitat types is preliminarily used (Figure 3-2).

This habitat map is the best available map according to the current status of knowledge and ongoing discussions. However, some habitats cannot be identified according to this map, e.g. 'sublittoral mixed sediment' or 'sea-pen and burrowing megafauna communities'. Therefore, in future assessments the habitat map shall be regularly updated with the latest state of research. It is also recommended to further refine predominant habitat types to EUNIS level 5 or 6 as far as possible. With the presently defined habitats on EUNIS level 3, a sensitivity assessment by means of characteristic species is difficult to achieve. EUNIS level 3 habitats are solely classified according to abiotic conditions like water depths and sediment type. At that level it is also not possible to identify habitats with high natural disturbance, e.g. areas with high current or tidal energy which may be more resistant towards physical pressures. An extensive habitat mapping project covering the German EEZ of the North Sea is currently under progress and should in the future provide information on habitat types and the associated characteristic species. For the present sensitivity assessment, the characteristic species of benthic assemblages defined by Rachor & Nehmer (2003) are used as a first approach to support the assessment of physical habitat properties with biological aspects (see chapter 3.9).

3.4.4 Physical impacts on habitats

The degree of physical impact on a habitat is a product of its sensitivity and the exposure to a specific pressure. An impact assessment thus requires the linkage of sensitivity information with pressure data. A matrix combining pressure intensity in terms of the temporal extent and habitat sensitivity supports the classification in nine categories of physical impact (Table 3-8). A percentage value is assigned to each rank which should provide an approximation of the relative impact on the habitat with regard to e.g. habitat structure, species richness, abundance or biomass. Due to the different nature of the pressures 'selective extraction', 'abrasion' and 'changes in siltation', for each of these physical damage pressures a separate impact matrix is provided in order to include a weighting factor in the impact assessment. 'Sealing' and 'smothering' are persistent pressures which are associated with an impact that destroys habitat structures as well as benthic organisms. The habitat is not expected to recover, thus sealing and smothering always result in a very high impact or total loss of habitat (100%).

Table 3-8: Impact matrix combining habitat sensitivity and temporal extent of pressure.

Impact		Habitat sensitivity				
		very low	low	intermediate	high	very high
Temporal extent of pressure	rare	very low	very low-low	low	low-medium	medium
	occasional	very low-low	low	low-medium	medium	medium-high
	regular	low	low-medium	medium	medium-high	high
	frequent	low-medium	medium	medium-high	high	high-very high
	persistent	medium	medium-high	high	high-very high	very high

Pressure-impact relationships may be described by various types of functions, e.g. linear relation or logarithm function, and depend on the habitat or the life strategy of species. As a first approach to set up an impact matrix, the modelling results of Schroeder et al. (2008) were used as a basis. Schroeder et al. (2008) modelled fishery-induced mortality rates of selected benthic species with different ecotypes (r- and K-selected species of in- and epifauna) for the fishing gears beam and otter trawl (Figure 3-4).

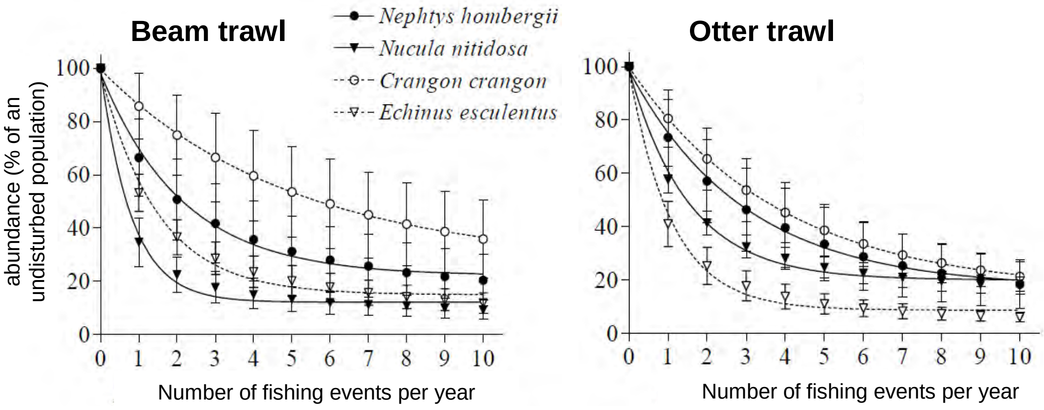

Figure 3-4: Percentage decrease in abundance of the benthic species *Nephtys hombergii*, *Nucula nitidosa*, *Crangon crangon* and *Echinus esculentus* induced by beam and otter trawling with different intensities per year (Schroeder et al. 2008).

For the development of an impact matrix, the decrease in abundance was averaged over the different species and gears to obtain a logarithmic curve for the physical impact of bottom trawling (Figure 3-5).

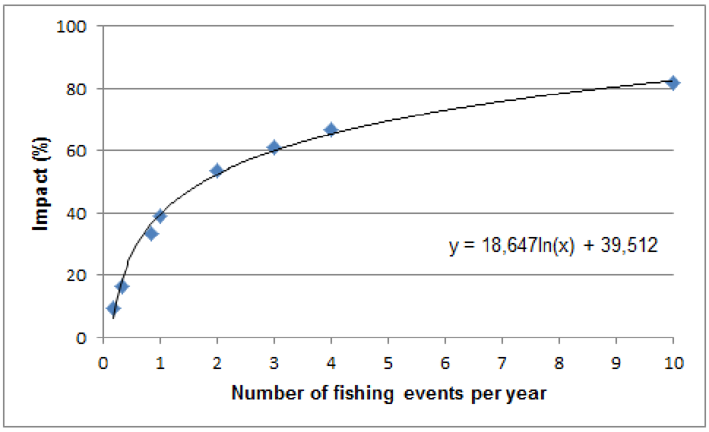

Figure 3-5: Estimated physical impact on benthic habitats by bottom trawling, based on decrease in abundance modelled by Schroeder et al. (2008).

In a second step the percentage values derived from the function were applied to the impact matrix combining sensitivity and temporal extent of pressure. Habitat sensitivity was set at intermediate with the respective temporal fishing intensities and then extrapolated to the very low and very high categories. For the impact matrices of the pressures ‘changes in siltation’ and ‘selective extraction’ weighting factors of 0.5 and 1.5 respectively were applied (Table 3-9).

Table 3-9: Values for relative impact on benthic habitats for the pressures ‘selective extraction’, ‘abrasion’ and ‘changes in siltation’.

Rank of impact	Selective extraction	Abrasion	Changes in siltation
very low	1%	0.5%	0.25%
very low – low	3%	2%	1%
low	9%	6%	3%
low – medium	44%	29%	15%
medium	59%	40%	20%
medium – high	85%	57%	28%
high	98%	65%	33%
high – very high	100%	80%	40%
very high	100%	100%	50%

3.4.5 Cumulative physical impacts on habitats

In order to determine the cumulative physical impact on a particular habitat, the five impact maps have to be summarised. Multiple pressures affecting a given location may vary in their cumulative impact. Several possible responses of habitats are discussed: Where pressure A causes the response ‘a’ from the habitat and pressure B the response ‘b’, then the cumulative effect under A + B conditions may be additive (a+b), antagonistic (<a+b) or synergistic (>a+b) (Crain et al. 2008). Most approaches to assess cumulative impacts assume additive effects for lack of knowledge on actual responses of benthic habitats. It is proposed to follow this practice as the physical pressures regarded here are assumed to affect habitat structure and suitability in a similar mode. This means that percentages for overlapping physical impacts are added up with 100 % (total loss) as maximum value. The cumulative physical impact is calculated from the proportion of area impacted (A, [%]) for each habitat and the corresponding value for impact intensity (I, [%]) as derived from the impact matrices. The cumulative impact (CI, [%]) for each habitat results from the sum of individual values for the relative impact on habitat:

$$CI = \sum I \times A / 100 \ [\%]$$

High values of cumulative impact indicate either pressures with considerable temporal and spatial extent or habitats with high sensitivity towards the occurring pressures. The cumulative impact value may range from 0% which would be a habitat completely without impacts to 100% meaning the total loss of the habitat.

This method provides the advantage of easily comparing the different impacts of the pressures physical loss (reduction in extent) and physical damage (impairment of condition) and results in a single percentage value of physical degradation for each habitat. Habitats and areas which are especially at risk by multiple pressures should be easily identified by this approach. Figure 3-6 briefly outlines the methodology to generate one map for the cumulative physical impact.

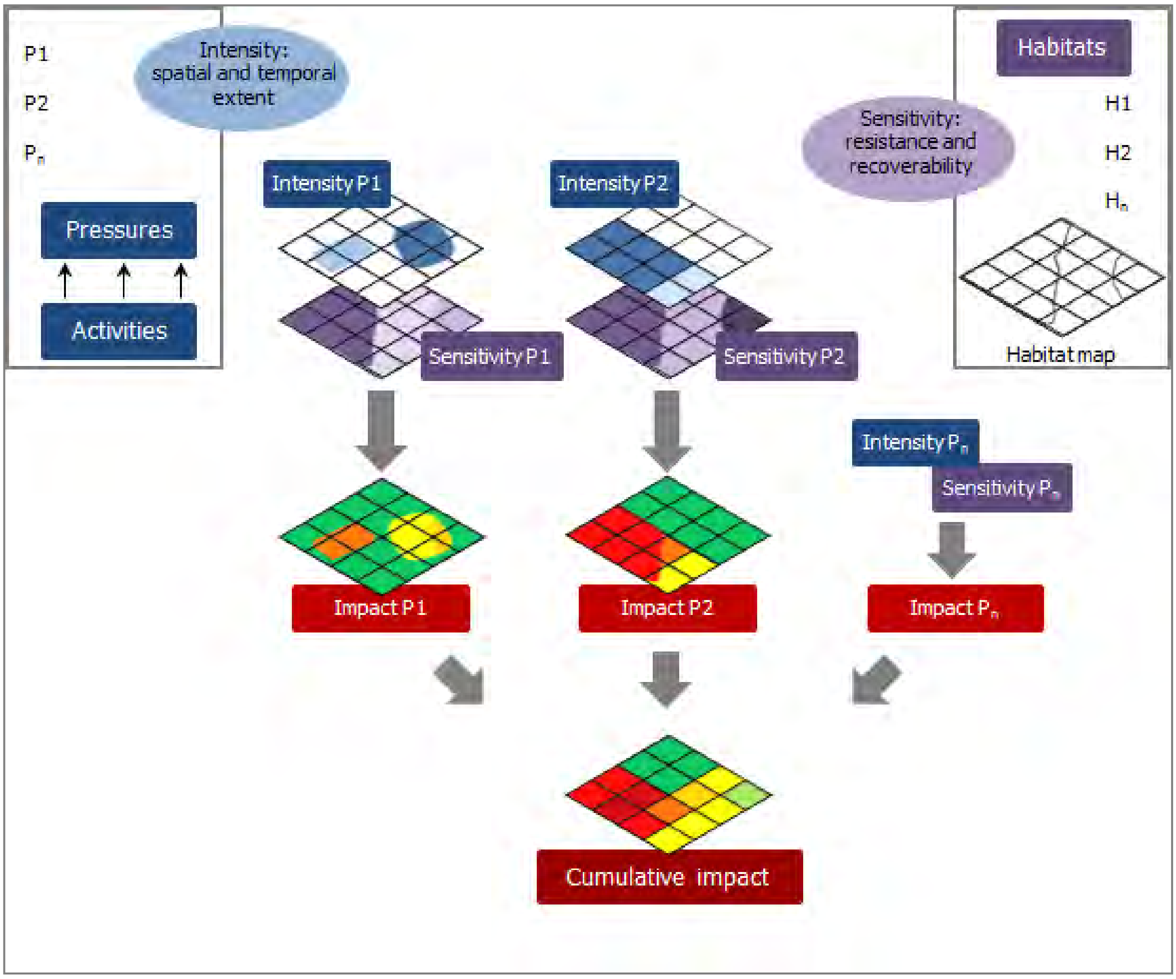

Figure 3-6: Assessment of cumulative physical impact by combining pressure intensity and habitat sensitivity (own illustration).

3.5 Application of assessment concept

3.5.1 Technical data

A first application of the proposed assessment concept was carried out for the German Exclusive Economic Zone (EEZ) of the North Sea. Anthropogenic activities considered are bottom trawling, permanent offshore installations, aggregate extraction and pipelines. Data formats and sources for human activities are described in Table 3-10. The habitat map used for the sensitivity assessment is based on the distribution of sediments in the German EEZ and the Natura 2000 habitat types (Figure 3-2). Data preparation and analysis was done with ArcGIS version 9.3.1.

Table 3-10: Data on human activities used for application of the assessment concept.

Activity	Data format	Data status	Data source
Bottom trawling	VMS data points, all fishing vessels > 15 m gear types: beam trawl <300 PS beam trawl >300 PS heavily rigged beam trawl > 300 PS otter / pair trawl area fished = temporal fishing effort x fishing speed x width of gear grid 100 x 100 m	2006	LANIS Habitat Mare, BfN method described in Schroeder et al. (2008)
Aggregate extraction	area in use for extraction	10/2013	CONTIS database, BSH
	area extracted	2005/2006	Schroeder et al. (2008)
Offshore wind farms	offshore wind farms in operation / under construction	10/2013	CONTIS database, BSH
Other offshore installations	installations for extractions of gas / research	10/2013	CONTIS database, BSH
Pipelines	pipelines in operation	10/2013	CONTIS database, BSH

3.5.2 Activities and pressures

Physical loss: sealing and smothering

The pressures 'sealing' and 'smothering' were combined as these pressures mostly arise from the same human activities and have the same temporal extent, i.e. they are persistent. Activities which are relevant with regard to physical loss are the foundations of offshore installations, scouring and scour protection around offshore installations and pipelines. The area impacted by physical loss is given in Table 3-11.

Table 3-11: Total area impacted by 'sealing' and 'smothering'.

Activity	Area impacted [km²]
Offshore wind farms	0.184
Research	0.006
Extraction of gas	0.004
Pipelines	1.052

Physical damage: selective extraction

Aggregate extraction is currently the only activity in the EEZ causing the pressure 'selective extraction'. At present there are three areas in use for sand and gravel extraction which are located in the Sylter Outer

Reef. Data on actual areas dredged are difficult to obtain. For this first application it was only possible to use approximate data of extracted areas in 2005 and 2006 as transferred from Schroeder et al. (2008) (Figure 3-7). The description of the temporal extent of dredging activities is likewise only an approximation due to lack of data.

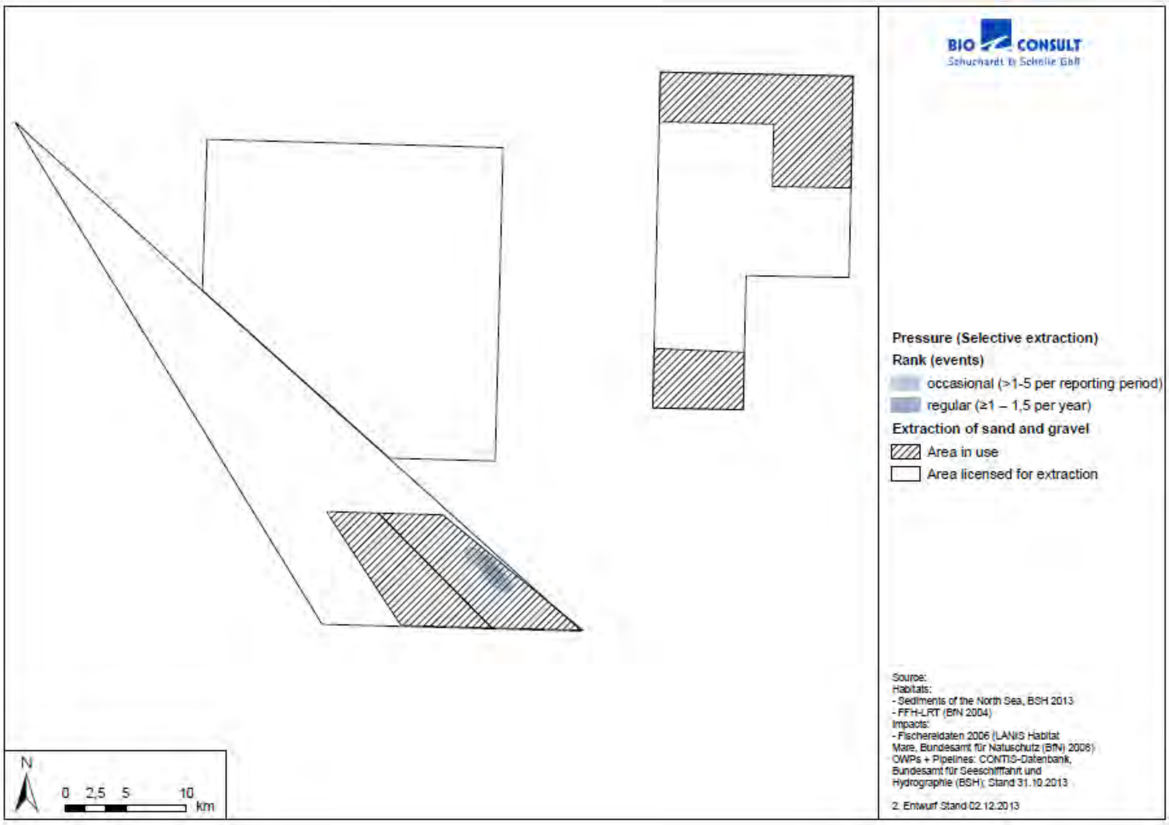

Figure 3-7: Pressure map for 'selective extraction' (detail of EEZ).

Physical damage: Abrasion

The pressure 'abrasion' in the German EEZ is caused by bottom trawling. Figure 3-8 shows the distribution and intensity of bottom trawling by beam and otter trawls in 2006.

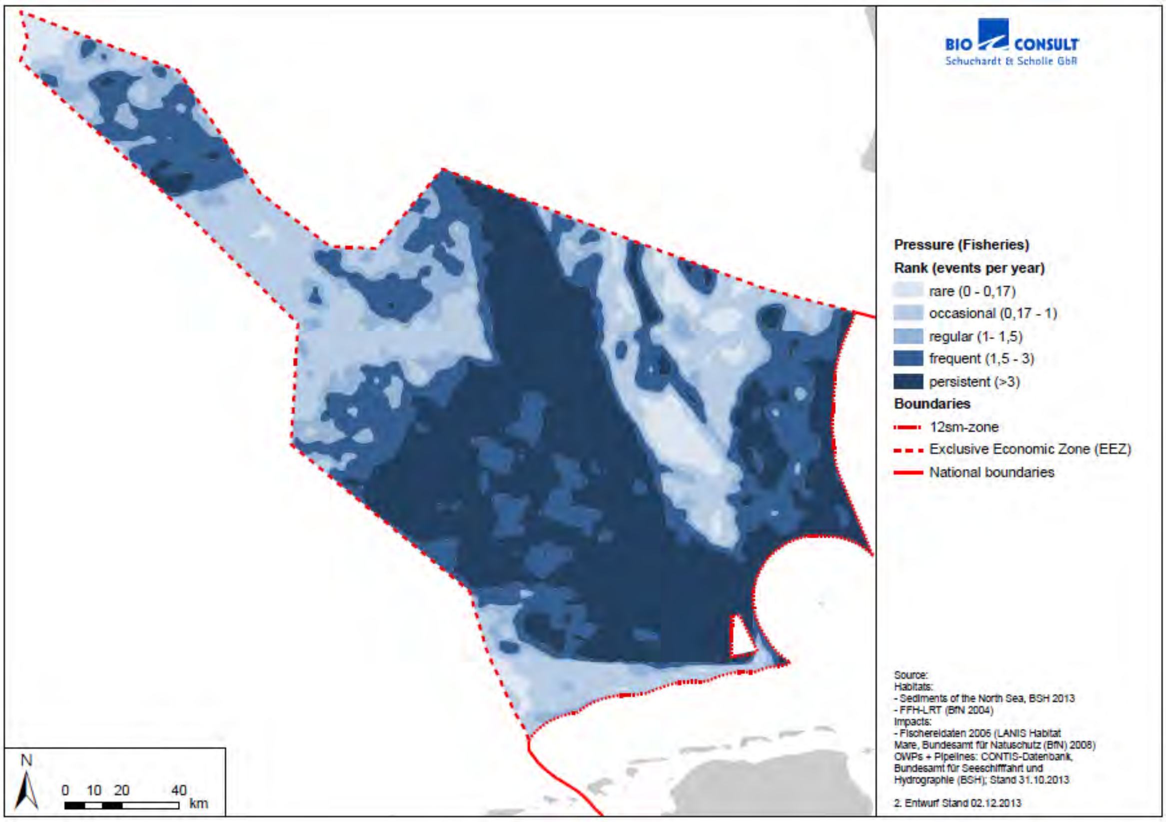

Figure 3-8: Pressure map for 'abrasion'.

Physical damage: Changes in siltation

'Changes in siltation' is a physical pressure associated with construction activities and dredging. Several construction works are currently ongoing in the German EEZ, mainly for offshore wind farms. Pressures resulting from the construction of power cables are not yet included in this assessment. For the extraction of sand and gravel the data from 2005 and 2006 were used with the uncertainties described for the pressure 'selective extraction' (Figure 3-9).

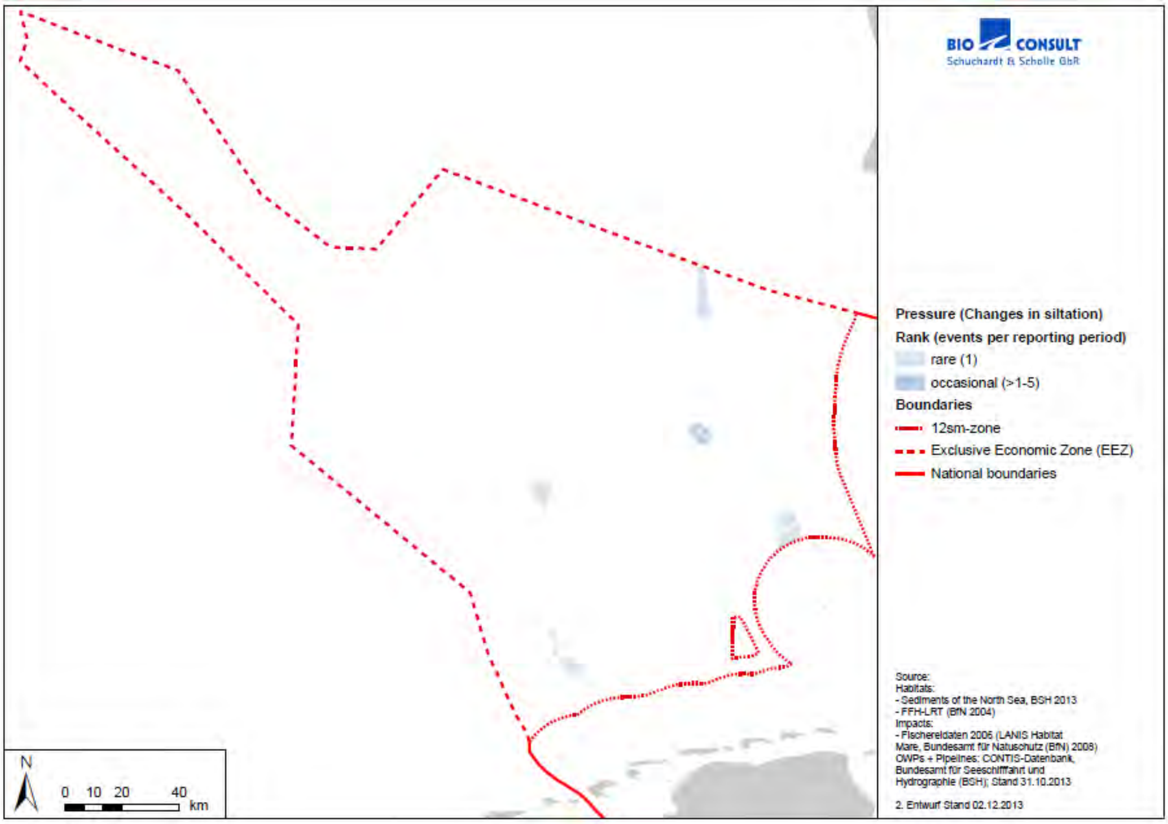

Figure 3-9: Pressure map for 'changes in siltation'.

3.5.3 Habitat sensitivity

Sensitivity maps were generated for each of the physical pressures according to the methodology described in chapter 3.4.3.3. The detailed assessments of the benthic habitats in the German EEZ are presented in chapter 3.9. Table 3-12 summarises the sensitivity ranks determined for the predominant and special habitats in the EEZ. The sandbank Doggerbank and further sandbanks on the Sylter Outer Reef and Borkum Reef Ground are listed separately as these habitats differ in their characteristic benthic communities. As an example, Figure 3-10 shows the sensitivity of benthic habitats towards the pressure 'abrasion'.

Table 3-12: Summary of sensitivity ranks for benthic habitats in the German North Sea towards the physical loss and damage pressures.

	Sealing	Smothering	Selective extraction	Abrasion	Changes in siltation
Sublittoral sand	very high	very high	intermediate	low	very low
Sublittoral mud	very high	very high	not relevant	low	very low
Sublittoral coarse	very high	very high	intermediate	intermediate	low
Sandbanks (Doggerbank)	very high	very high	intermediate	intermediate	low
Other Sandbanks	very high	very high	intermediate	low	very low
Reefs	very high	very high	very high	high	intermediate
Species-rich coarse/gravel/shell	very high	very high	high	intermediate	low

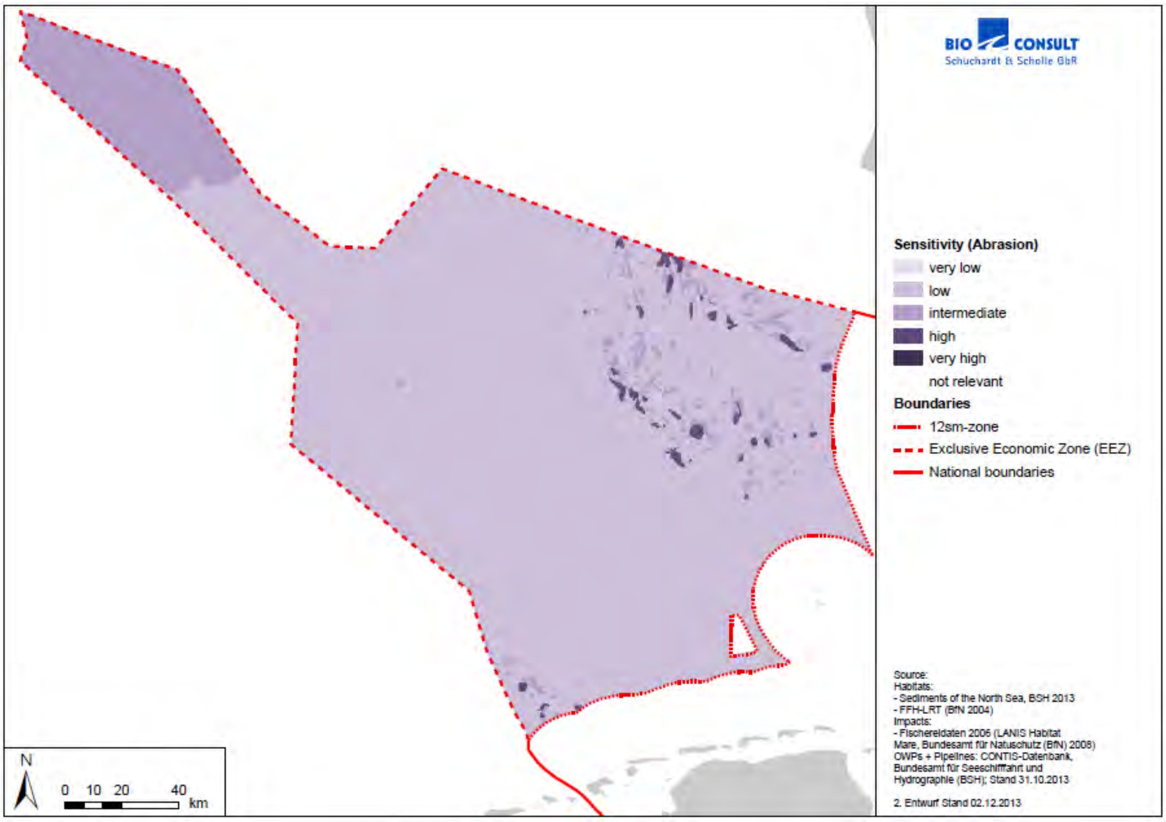

Figure 3-10: Habitat sensitivity towards the pressure 'abrasion'.

3.5.4 Physical impact on benthic habitats

With the information from the pressure maps and the related habitat sensitivity maps combined, the potential impact of each of the pressures on benthic habitats is visualized. Figure 3-11 shows the impact map for the pressure 'abrasion' as an example. In Table 3-14 and Table 3-15 the calculated absolute and relative area of benthic habitats impacted by each of the physical pressures is given.

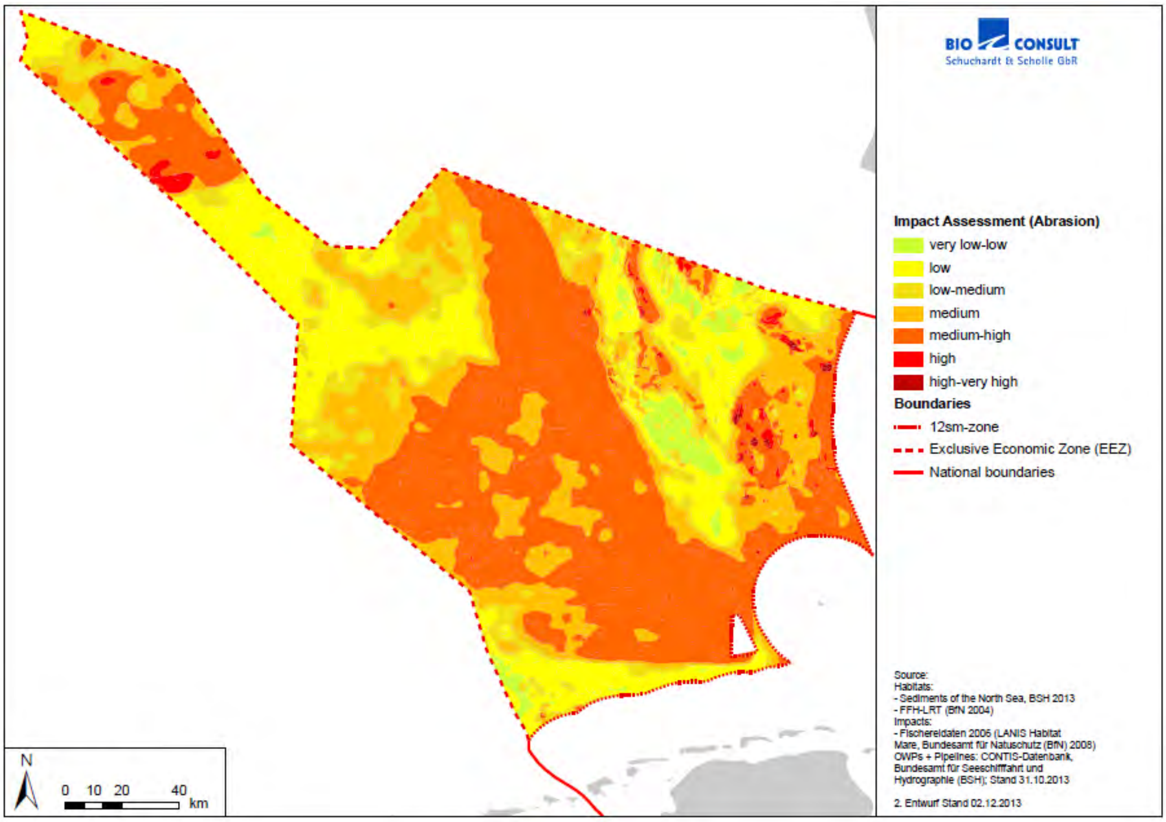

Figure 3-11: Impact on benthic habitats for the pressure 'abrasion'.

Table 3-13 summarises the total area of the German EEZ impacted by each of the physical pressures. In terms of area, 'abrasion' caused by bottom trawling is the main pressure which covers nearly the complete seabed of the EEZ (98.9 %). Areas without abrasion are solely the construction sites of offshore wind farms as well as operational wind farms. Areas subject to physical loss currently account for less than 0.01 % of the total area. The pressure 'changes in siltation' affects 1 % of the EEZ with the predominant activity being the construction of offshore wind farms. Selective extraction in 2005 / 2006 was restricted to an area of 0.02 % of the EEZ.

Table 3-13: Total area impacted by physical pressures in the German EEZ of the North Sea.

Pressure	Area impacted [km²]	Area impacted [%]
Sealing / smothering	1.2	<0.1
Selective extraction	6.3	<0.1
Abrasion	28142.8	98.9
Changes in siltation	283.3	1.0

Table 3-14: Area impacted (in km²) of benthic habitats in the German EEZ of the North Sea.

Habitat	Pressure	Area impacted [km²]									Total area impacted [km²]
		very low	very low – low	low	low – medium	medium	medium – high	high	high – very high	very high	
Sublittoral sand	Sealing / smothering	-	-	-	-	-	-	-	-	0.9	0.9
	Selective extraction	-	-	-	0.8	-	-	-	-	-	0.8
	Abrasion	-	712.4	4488.4	2350.0	5091.6	9802.3	-	-	-	22444.7
	Changes in siltation	189.7	4.2	-	-	-	-	-	-	-	193.9
Sublittoral mud	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Selective extraction	-	-	-	-	-	-	-	-	-	-
	Abrasion	-	0.7	231.5	112.4	509.2	570.2	-	-	-	1423.9
	Changes in siltation	0.2	-	-	-	-	-	-	-	-	0.2
Sublittoral coarse sediment	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Selective extraction	-	-	-	-	-	-	-	-	-	-
	Abrasion	-	-	41.7	161.0	63.8	169.9	105.2	-	-	541.6
	Changes in siltation	-	9.8	5.1	-	-	-	-	-	-	14.9
Sandbanks (Doggerbank)	Sealing / smothering	-	-	-	-	-	-	-	-	0.3	0.3
	Selective extraction	-	-	-	-	-	-	-	-	-	-
	Abrasion	-	-	193.0	361.3	530.7	1036.8	119.9	-	-	2241.7
	Changes in siltation	-	-	-	-	-	-	-	-	-	-
Other Sandbanks	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Selective extraction	-	-	-	-	-	-	-	-	-	-
	Abrasion	-	85.7	556.6	102.2	121.7	189.6	-	-	-	1055.8
	Changes in siltation	44.8	-	-	-	-	-	-	-	-	44.8
Reefs	Sealing / smothering	-	-	-	-	-	-	-	-	-	<0.1
	Selective extraction	-	-	-	-	-	3.1	2.4	-	-	5.5
	Abrasion	-	-	-	97.5	62.2	15.6	29.9	36.5	-	241.7
	Changes in siltation	-	-	7.0	12.1	-	-	-	-	-	19.1
Species-rich coarse /gravel /shell	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Selective extraction	-	-	-	-	-	-	-	-	-	-
	Abrasion	-	-	42.4	72.0	17.1	36.3	25.7	-	-	193.4
	Changes in siltation	-	9.3	1.1	-	-	-	-	-	-	10.4

Table 3-15: Area impacted (in %) of benthic habitats in the German EEZ of the North Sea.

Habitat	Pressure	Area impacted [%]									Total area impacted [%]
		very low	very low – low	low	low – medium	medium	medium – high	high	high – very high	very high	
Sublittoral sand	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Selective extraction	-	-	-	<0.1	-	-	-	-	-	<0.1
	Abrasion	-	3.1	19.8	10.4	22.4	43.2	-	-	-	98.9
	Changes in siltation	0.8	<0.1	-	-	-	-	-	-	-	0.9
Sublittoral mud	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Selective extraction	-	-	-	-	-	-	-	-	-	-
	Abrasion	-	0.05	16.2	7.9	35.6	39.9	-	-	-	99.7
	Changes in siltation	0.02	-	-	-	-	-	-	-	-	<0.1
Sublittoral coarse sediment	Sealing / smothering	-	-	-	-	-	-	-	-	0.1	0.1
	Selective extraction	-	-	-	-	-	-	-	-	-	-
	Abrasion	-	-	7.6	29.3	11.6	31.0	19.2	-	-	98.7
	Changes in siltation	-	1.8	0.9	-	-	-	-	-	-	2.7
Sandbanks (Doggerbank)	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Selective extraction	-	-	-	-	-	-	-	-	-	-
	Abrasion	-	-	8.6	16.1	23.7	46.3	5.3	-	-	100.0
	Changes in siltation	-	-	-	-	-	-	-	-	-	-
Other Sandbanks	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Selective extraction	-	-	-	-	-	-	-	-	-	-
	Abrasion	-	7.8	50.5	9.3	11.0	17.2	-	-	-	95.7
	Changes in siltation	4.1	-	-	-	-	-	-	-	-	4.1
Reefs	Sealing / smothering	-	-	-	-	-	-	-	-	-	-
	Selective extraction	-	-	-	-	-	1.3	1.0	-	-	2.3
	Abrasion	-	-	-	40.0	25.5	6.4	12.3	15.0	-	99.1
	Changes in siltation	-	-	2.9	4.9	-	-	-	-	-	7.8
Species-rich coarse /gravel /shell	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Selective extraction	-	-	-	-	-	-	-	-	-	-
	Abrasion	-	-	21.3	36.1	8.6	18.2	12.9	-	-	97.0
	Changes in siltation	-	4.6	0.6	-	-	-	-	-	-	5.2

3.5.5 Cumulative physical impact

The separate impact maps finally result in one cumulative impact map (Figure 3-12). The dominant physical pressure in the German EEZ is 'abrasion' caused by bottom trawling. Impacts which interfere with each other are areas with aggregate extraction and bottom trawling as well as pipelines and bottom trawling. Other human uses are mutually exclusive, for example construction works and bottom trawling or operational wind farms, where fishing is excluded. However, for the resulting cumulative impact it must be noted that fishing data are from 2006, where no wind farms were under construction. With several OWF areas excluded from trawling, it is possible that fishing effort has shifted to other areas and has actually increased elsewhere.

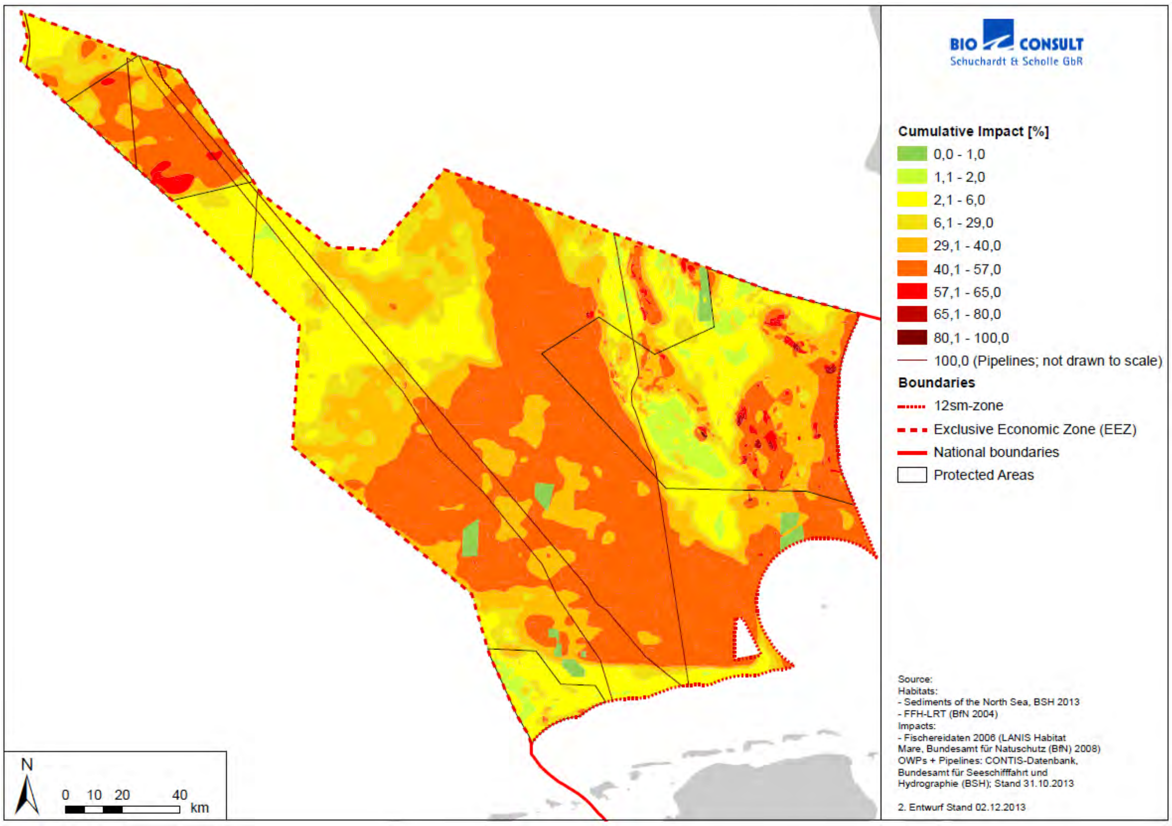

Figure 3-12: Cumulative physical impact on benthic habitats.

The cumulative physical impact has been calculated from the proportion of area impacted (A, [%]) for each habitat and the corresponding intensity of impact (I, [%]) as derived from the impact matrices (see chapter 3.4.4). The cumulative impact (CI, [%]) for each habitat results from the sum of individual values for the relative impact on habitat:

$$CI = \sum I \times A / 100 \, [\%]$$

Table 3-16 gives an example for the calculation of the cumulative impact on the predominant habitat 'sublittoral mud'.

Table 3-16: Calculation of cumulative impact as exemplified by the predominant habitat 'sublittoral mud'.

Pressure	Rank of intensity	Intensity of impact (I [%])	Area impacted (A [%])	Relative impact on habitat (I x A /100 [%])
Changes in siltation	very low	0.25	0.02	<0.01
Abrasion	very low - low	2	0.05	<0.01
Abrasion	low	6	16.20	0.97
Abrasion	low - medium	29	7.87	2.28
Abrasion	medium	40	35.64	14.26
Abrasion	medium - high	57	39.91	22.75
Sealing / smothering	very high	100	<0.01	<0.01
Cumulative impact (∑ Relative impact)				**40.26**

The resulting cumulative impact values are presented in Table 3-17. The calculated cumulative impact ranges from 20.1 % for sandbanks on the Borkum Reef Ground / Sylter Outer Reef to 47.6 % for reef habitats. The impact values mainly arise from high impacts of bottom trawling. Major parts of the benthic habitats are fished more than once a year, e.g. 65 % of the widespread sand habitats are subject to trawling more than 1.5 times per year. The comparatively low cumulative impact value for 'other sandbanks' originates from the lower fishing pressure on the Borkum Reef Ground, where half of the sandbank area is trawled less than once a year. The high impact value for reefs is mainly caused by the high sensitivity towards 'abrasion' determined for this habitat.

Table 3-17: Calculated cumulative impact of physical loss and damage on benthic habitats.

Habitat	Cumulative impact
Sublittoral sand	37.9 %
Sublittoral mud	40.3 %
Sublittoral coarse sediment	43.8 %
Sandbanks (Doggerbank)	44.5 %
Other sandbanks	20.1 %
Reefs	47.6 %
Species-rich coarse/gravel/shell	34.0 %

3.5.6 Physical impacts on marine protected areas

The physical impact of the individual pressures has been calculated for benthic habitats in marine protected areas as well (Table 3-18, Table 3-19).

Sylter Outer Reef

The cumulative impact on benthic habitats in the Sylter Outer Reef ranges from 31.0 % for the predominant habitat 'sublittoral sand' to 56.1 % for 'sublittoral mud' (Table 3-20). High impact values were also calculated for 'sublittoral coarse sediment' (46.7 %), 'reefs' (51.5 %) and 'sandbanks' (56.0 %). The wide range of cumulative impact values corresponds to varying fishing intensity in the Sylter Outer Reef. While large parts of the Natura 2000 site were fished with low intensity, other areas were subject to persistent fishing pressure of up to five times per year.

Borkum Reef Ground

The only physical pressure affecting benthic habitats at the Natura 2000 site Borkum Reef Ground is 'abrasion' caused by bottom trawling. In 2006, fishing intensity was comparatively low with generally less than once per year. With the exception of reef habitats, the cumulative impact values for habitats in the Borkum Reef Ground were likewise relatively low, varying from 6.0 % to 23.7 % (Table 3-20). The habitat 'sandbank' which covers the major part of the protected site (77.4 %) holds a cumulative impact of 8.7 %. Due to the high sensitivity rank of reefs towards 'abrasion', the cumulative impact of this habitat type amounts to 40.1 %.

Doggerbank

The total area of the Doggerbank is subject to 'abrasion' by bottom trawling and is additionally crossed by three gas pipelines. The cumulative impact of the main habitat 'sandbank' (95.6 % of total area) at the Doggerbank accounts for 51.2 % (Table 3-20). The impact values for 'sublittoral sand' amounts to 34.4 % and for 'sublittoral mud' 20.2 %. However, muddy habitats cover only 0.02 % of the total area.

Table 3-18: Area impacted (in km²) of habitats in marine protected areas in the German EEZ of the North Sea.

Habitat	Pressure	Area impacted [km²]									Total area impacted [km²]
		very low	very low – low	low	low – medium	medium	medium – high	high	high – very high	very high	
Sylter Outer Reef											
Sublittoral sand	Sealing / smothering	-	-	-	-	-	-	-	-	0.1	0.1
	Selective extraction	-	-	-	0.8	-	-	-	-	-	0.8
	Abrasion	-	488.9	1125.4	399.0	1212.6	1235.6	-	-	-	4461.5
	Changes in siltation	6.8	4.2	-	-	-	-	-	-	-	11.1
Sublittoral mud	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Abrasion	-	0.6	1.6	-	0.6	132.4	-	-	-	135.1
Sublittoral coarse sediment	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Abrasion	-	-	15.6	108.6	47.0	127.7	84.2	-	-	383.1
	Changes in siltation	-	2.5	5.1	-	-	-	-	-	-	7.6
Sandbanks	Abrasion	-	-	-	-	4.8	75.1	-	-	-	79.9
	Selective extraction	-	-	-	-	-	3.1	2.4	-	-	5.5
Reefs	Abrasion	-	-	-	56.2	43.6	4.3	19.6	33.0	-	156.7
	Changes in siltation	-	-	4.7	12.1	-	-	-	-	-	16.8
Species-rich coarse /gravel /shell	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Abrasion	-	-	19.9	37.2	8.6	18.7	12.6	-	-	96.9
	Changes in siltation	-	3.2	1.1	-	-	-	-	-	-	4.4
Borkum Reef Ground											
Sublittoral sand	Abrasion	-	15.8	44.0	2.5	8.6	-	-	-	-	70.8
Sublittoral mud	Abrasion	-	-	1.3	-	-	-	-	-	-	1.3
Sublitt. coarse	Abrasion	-	-	13.0	22.4	5.2	-	-	-	-	40.6
Sandbanks	Abrasion	-	80.2	319.6	57.6	6.7	-	-	-	-	464.1
Reefs	Abrasion	-	-	-	10.0	6.1	6.6	-	-	-	22.7
coarse /gravel	Abrasion	-	-	4.8	12.1	1.4	-	-	-	-	18.3
Doggerbank											
Sublittoral sand	Sealing / smothering	-	-	-	-	-	-	-	-	0.2	0.2
	Abrasion	-	-	5.7	26.9	36.5	4.6	-	-	-	73.8
Sublittoral mud	Abrasion	-	-	0.1	0.2	-	-	-	-	-	0.3
Sandbanks	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Abrasion	-	-	-	147.5	367.1	974.4	119.9	-	-	1608.9

Table 3-19: Area impacted (in %) of habitats in marine protected areas in the German EEZ of the North Sea.

Habitat	Pressure	Area impacted [%]									Total area impacted [%]
		very low	very low – low	low	low – medium	medium	medium – high	high	high – very high	very high	
Sylter Outer Reef											
Sublittoral sand	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Selective extraction	-	-	-	<0.1	-	-	-	-	-	<0.1
	Abrasion	-	11.0	25.2	8.9	27.2	27.7	-	-	-	100.0
	Changes in siltation	0.2	0.1	-	-	-	-	-	-	-	0.2
Sublittoral mud	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Abrasion	-	0.4	1.1	-	0.4	98.0	-	-	-	100.0
Sublittoral coarse sediment	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Abrasion	-	-	4.1	28.3	12.3	33.3	22.0	-	-	100.0
	Changes in siltation	-	0.7	1.3	-	-	-	-	-	-	2.0
Sandbanks	Abrasion	-	-	-	-	6.0	94.0	-	-	-	100.0
	Selective extraction	-	-	-	-	-	2.0	1.6	-	-	3.5
Reefs	Abrasion	-	-	-	35.9	27.8	2.8	12.5	21.0	-	100.0
	Changes in siltation	-	-	3.0	7.7	-	-	-	-	-	10.7
Species-rich coarse /gravel /shell	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Abrasion	-	-	20.6	38.4	8.8	19.3	13.0	-	-	100.0
	Changes in siltation	-	3.3	1.2	-	-	-	-	-	-	4.5
Borkum Reef Ground											
Sublittoral sand	Abrasion	-	22.3	62.1	3.5	12.2	-	-	-	-	100.0
Sublittoral mud	Abrasion	-	-	100.0	-	-	-	-	-	-	100.0
Sublitt. coarse	Abrasion	-	-	31.9	55.2	12.9	-	-	-	-	100.0
Sandbanks	Abrasion	-	17.3	68.9	12.4	1.4	-	-	-	-	100.0
Reefs	Abrasion	-	-	-	43.9	27.0	29.1	-	-	-	100.0
coarse /gravel	Abrasion	-	-	26.4	66.2	7.4	-	-	-	-	100.0
Doggerbank											
Sublittoral sand	Sealing / smothering	-	-	-	-	-	-	-	-	0.3	0.3
	Abrasion	-	-	7.8	36.5	49.5	6.2	-	-	-	100.0
Sublittoral mud	Abrasion	-	-	38.4	61.6	-	-	-	-	-	100.0
Sandbanks	Sealing / smothering	-	-	-	-	-	-	-	-	<0.1	<0.1
	Abrasion	-	-	-	9.2	22.8	60.6	7.5	-	-	100.0

Table 3-20: Calculated cumulative impact of physical loss and damage on benthic habitats in marine protected areas.

Habitat	Proportion of total protected area (%)	Cumulative impact (%)
Sylter Outer Reef		
Sublittoral sand	85.5	31.0
Sublittoral mud	2.6	56.1
Sublittoral coarse sediment	7.3	46.7
Sandbanks	1.5	56.0
Reefs	3.0	51.5
Species-rich coarse/gravel/shell	1.9	35.4
Borkum Reef Ground		
Sublittoral sand	11.8	10.0
Sublittoral mud	0.2	6.0
Sublittoral coarse sediment	6.8	23.1
Sandbanks	77.4	8.7
Reefs	3.8	40.1
Species-rich coarse/gravel/shell	3.0	23.7
Doggerbank		
Sublittoral sand	4.4	34.4
Sublittoral mud	<0.1	20.2
Sandbanks	95.6	51.2

3.6 Setting baselines

Within the MSFD the baseline is defined as a state or condition against which the Good Environmental Status can be assessed. Therefore, determining the baseline state is an essential precondition to the development of reasonable GES targets and the subsequent assessment of the present state of habitats in relation to these targets. In the proposed concept the determination of a baseline or reference state is also important for habitat sensitivity as the assessment should be based on habitats in an optimum state. Several methods for setting baselines for marine benthic habitats are described in literature and shall at first briefly be presented here:

Reference state: Baselines can be set as a state or condition in which impacts from anthropogenic pressures are absent or negligible. This approach was used to set reference conditions for the Water Framework Directive. There are basically three options to accomplish this approach.

- Existing reference state: Current information on species and habitats from areas where human pressures are considered to be negligible are used as a baseline. It has the advantage of setting a baseline under current physiographic, geographic and climatic conditions. However, there may be limited availability of such genuinely unimpacted sites (OSPAR 2012).

- Historical reference state: Baseline is set by means of historical data which describe habitats at a time when human impacts were negligible or absent. Information may be obtained by a variety of sources such as historical accounts, old maps or fishing records. This provides a moderately scientifically robust basis, depending on the quality and quantity of available data and requiring expert interpretation. Obtaining the required data may be highly time and resource consuming. Furthermore, historical information does not include the influence of climate change and other natural or anthropogenic dynamics which cannot be reversed, so additional data is needed for the determination of the reference state. Regarding the distribution and extent of habitats, historical data sets may be particularly important as some habitats may have substantially decreased, e.g. seagrass beds (Hill et al. 2012).
- Modelling of reference state: Reference state can be determined by modelling a theoretical unimpacted state under present climatic conditions. The scientific robustness of this approach depends on the quality and quantity of historical and / or current data. At present modelling capabilities are not regarded as sufficient for defining reference conditions, however, this may change with future developments and modelling may be useful to support other baseline-setting approaches (Hill et al. 2012).

<u>Past state:</u> Baselines can be set as a state in the past, based on a time-series dataset for a specific habitat. Expert judgement is involved to determine the period in the time series which is assumed to reflect the least impacted conditions, e.g. the first data record. However, this may already represent some degree of deterioration from unimpacted state and may also run the risk of the ‘shifting baseline syndrome’, where each generation redefines the standard for a healthy environment (OSPAR 2012).

<u>Current state:</u> Baselines can be set as the date when a particular environmental directive or policy comes into force or the first assessment of state. This approach was used in the context of the Habitats Directive. In order to account for the possibly already impacted current state of the environment, the target associated with this baseline is typically directed towards the prevention of any further deterioration or even towards the improvement of current state (OSPAR 2012).

Advice on setting baselines for the MSFD indicators related to benthic habitats is given in OSPAR (2012) and Hill et al. (2012). The use of reference state as a baseline for seabed habitats is widely acknowledged and recommended, as baselines at past or current state often represent an already impacted state. The reference state, that is the state when pressures on benthic habitats are absent or negligible, may be generated by current or historical data or by modelling. It is proposed to use the existing reference state as a baseline, i.e. the current extent of habitats and a condition where impacts from anthropogenic pressures are absent or negligible. This reference state reflects current physiographic, geographic and climatic conditions and should be also used to determine habitat sensitivities. However, it is acknowledged that the existing reference state may be difficult to determine as benthic habitats have been subject to human activities for a long time. Thus, it may prove necessary to take account of historical data for some aspects of the reference state. For example, especially the predominant habitats of the south-eastern North Sea are presently dominated by short-lived opportunistic species due to extensive fishing pressure, while in the past larger species with a longer life span were characteristic for some benthic habitats. For future assessments, the consideration of other species for reference state and consequently for the sensitivity assessment may affect results of habitat sensitivity which should be evaluated carefully. Expert judgement or spatial modelling may also aid in setting reference conditions. However, this approach does not consider habitats which have deteriorated in range and extent due to human impacts, e.g. European oyster beds or *Sabellaria spinulosa* reefs. For these habitats the historical extent and reference conditions in combination with expert judgement have to be considered.

3.7 Setting GES targets

3.7.1 Existing approaches for setting environmental targets

Targets for the Good Environmental Status represent boundaries or thresholds between the acceptable and unacceptable status of the marine environment (GES or below GES). Generally, targets are established with reference to a baseline. They can either be set directional (improvement towards a more desirable state), at the baseline itself or as a deviation from the baseline. The latter is recommended and is in line with the requirements of the MSFD since it allows for setting a detailed quantitative target (OSPAR 2012).

Existing targets related to the extent of habitats can be found within the Habitats Directive and under the OSPAR Convention. The Habitats Directive sets individual targets to achieve the Favourable Conservation Status for the parameters range, area, structure and function (condition) and future prospects. The same target values are applied to all habitat types of community interest. The overall assessment of a particular habitat is determined by the worst class allocated to the four parameters. Regarding the parameter area, the target for the Favourable Conservation Status is set at: stable (loss and expansion in balance) or increasing AND not smaller than the 'favourable reference area' (EC 2005). Generally, member states have adopted a tolerance level between 1 % and 5 % for the ‘stable area’. The baseline ‘favourable reference area’ is commonly defined as the state when the Habitats Directive came into force in 1994. Regarding specific structures and functions (i.e. condition of the habitat), a quantitative value is only used to describe the ‘unfavourable-bad’ status. This status is achieved when more than 25% of the habitat area is damaged.

Within the national implementation of the Habitats Directive, the consideration of the area impacted exclusively in relation to the overall size of the respective habitat type was not regarded as sufficient to determine adverse effects, as in the case of extensive habitats large areas could be lost without being considered as adversely affected. Therefore, absolute threshold values were specifically assigned to the Natura 2000 habitat types in combination with relative thresholds of 1 %, 0.5 % and 0.1 %, e.g. an acceptable area loss of up to 5 ha for reefs and sandbanks if the relative loss does not exceed 0.1 %. The relative thresholds are indicated supplementary in order to account for the protection of smaller occurrences of individual habitat types within a Natura 2000 site (Lambrecht & Trautner 2007).

The ‘Texel-Faial criteria’ were designed to assess threatened and/or declining habitats in the OSPAR area. One of these criteria relates to the decline of habitats in extent or quality. Where a habitat has declined by 15 % or more of its former natural distribution, it is defined as ‘significantly declined’. The decline may be historic, recent or current. OSPAR proposes to apply this 15 % threshold as a target for the distribution and extent of predominant habitats (OSPAR 2012). In the current versions of the OSPAR indicators ‘area of loss’ and ‘physical damage’ the 15 % value is suggested for predominant habitats while for special habitats the target should be ‘stable or increasing and not smaller than baseline value’ with a 5 % tolerance. However, concern was expressed by the experts that the proposed targets may be unacceptably high and further discussion will be necessary (OSPAR 2013a, b). Concerns about the 15 % target have also been raised by other experts assigned with the implementation of the MSFD. CEFAS (2012) criticises that the 15 % represents a threshold beyond which the habitat is related to the list of threatened and/or declining habitats in order to prioritise their conservation. The threshold was not developed to establish a target for an acceptable impact with regard to the requirements of the MSFD.

In the HELCOM indicator 'Cumulative impact on benthic habitats' a GES target of 'at most 25% of the habitat area being significantly impacted' is used (Korpinen et al. 2013). This value is derived from the Habitats Directive, where a habitat which has over 25% of the area significantly impacted regarding its specific structures and functions is classified to 'unfavourable – bad status'. However, the authors point out that the GES target for the MSFD should use a stricter threshold than the one for 'unfavourable - bad status'. Despite the comparatively high target value applied by Korpinen et al. (2013), with one exception all of the benthic habitats in the German part of the Baltic Sea and bordering areas (Danish and German Straits and Bights, Arkona and Bornholm Basin) failed the GES target in the impact analysis.

Within the Water Framework Directive (WFD) baselines were determined as reference conditions and the related target ('good ecological state') as deviations from the baselines. This methodical approach is in line with the recommendations by OSPAR and the EC for implementing the MSFD, however, as the WFD mainly addresses Biological Quality Elements, an adoption of targets is more relevant for the condition indicators than for indicators describing the parameters extent or distribution.

3.7.2 Considerations for GES targets

According to the MSFD, the Good Environmental Status of Descriptor 6 is achieved when 'seafloor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected' (EC 2008). As it is currently not possible to set a scientifically robust target for sustainable human use, it is acknowledged that GES targets for indicator 6.1.2 have to be a product of expert judgement and decisions at a policy level. In accordance with the majority of experts it is recommended to establish a quantitative target which defines the proportion of a habitat that is acceptable to be below GES without endangering the structure and functioning of the habitat as a whole.

It is recommended that the GES target for indicator 6.1.2 should

- be appropriate to cover impacts both from loss of habitat area and on condition of the habitat,
- differentiate between predominant and special habitat types,
- be in line with national targets for protected habitat types (Natura 2000 habitats),
- especially consider habitats with severe historical decline (e.g. European oyster beds),
- take account of the spatial distribution of impacts.

The assessment concept of indicator 6.1.2 combines pressures associated with physical loss and physical damage. Concerns have been raised that with this procedure habitat losses may not be adequately recognized. At present, area loss in the German EEZ caused by sealing and smothering accounts for only a very small part of the seabed (<0.01%), however, this may change in future and for particular habitat types. Thus, for now it is not deemed necessary to differentiate between physical loss and damage, whereas for future assessments it is suggested to critically review the need for an additional GES target to limit habitat loss.

Different GES targets should apply for predominant and special habitat types, as the latter are already designated for special protection. Special habitat types are among others priority habitats of the Habitats Directive. GES targets for the MSFD have to be in accordance with other national conservation objectives, therefore targets developed for the Habitat Directive must be applied also to corresponding MSFD habitat types. However, the Habitats Directive defines no quantitative target for an acceptable impact on habitat condition. Thresholds for acceptable loss can be found in Lambrecht & Trautner (2007), but these thresholds describe a significance level for new projects rather than existing impacts, e.g. from bottom trawling.

Habitats which have suffered major deterioration in the past have to be assessed with other criteria and targets. The objective should initially be an increase in habitat area towards historical extent. This trend target is proposed in order to support restoration projects which are currently planned for e.g. European oyster beds. If these projects are successfully realised, the target should be reviewed for the following reporting period and may possibly be changed into a quantitative target value. If restoration of a historically declined habitat type leads to area losses of other habitats, this should not be assessed as a negative effect as the reason for habitat loss is not destructive human use.

The spatial distribution of human activities and pressures should allow for areas with no or relatively low impacts, which could serve as sources for repopulation of damaged habitat areas. Especially regarding fisheries, the distribution of intensities is crucial and should be well planned for management actions. The exclusion of fisheries from an area may lead to an increase in fishing intensity in areas with previously low fishing effort. More positive effects may be expected if trawling activities are shifted to areas with an already high fishing intensity (Schroeder et al. 2008). A target considering the spatial distribution of impacts is currently not proposed, however, it should be ensured that GES targets allow for an adequate habitat area with no or low impacts.

It is recommended to regularly review GES targets (e.g. with every reporting period) in order to take account of changes in prevailing conditions or human pressures and improvement in scientific evidence and management experience. It may also be useful to set interim targets as a first step to reverse the trend of degradation of seabed habitats to one of recovery. A possible interim target could be the 'reduction of habitat area with persistent pressures', this would e.g. restrict the areas which are fished more than three times per year. Another possible interim target could be a two-tier quantitative target for high impacts (ranks medium-high to very high) and overall impacts, e.g. '10% of habitat area may be subject to high impacts with no more than 25% subject to total impacts'.

In contrast to other approaches such as Korpinen et al. (2013), it is recommended for the assessment concept of indicator 6.1.2 not to differentiate between significant impacts on benthic habitats and impacts of less severity. It is reckoned that all impacts should be considered, as a habitat with an extensive but low impact may be in a worse state compared to a habitat with a very small but very high impact (e.g. wind turbine foundation). If the GES target for physical loss and damage would be set at 'no more than 15 % of habitat area below GES' for predominant habitats like suggested by OSPAR, this would mean that in this value all impacts from very low to very high are included (while in Korpinen et al. (2013) only the two highest impact classes are considered). A habitat with a low sensitivity towards abrasion (which is the main pressure in the German EEZ) could thus suffer a persistent fishing intensity on a quarter of the total habitat area or a yearly trawling event on half of the total area and still achieve GES – provided that the remaining habitat area is completely free from human impacts. With the assessment of the total impact it is necessary to have major areas with relatively low impact, otherwise GES cannot be achieved. The proposed assessment method also provides various possibilities to implement management actions to achieve GES: areas with high impacts can be compensated by areas with no / very low uses or impacts could be restricted on the whole habitat area.

Against this background it is assumed that a GES target of 15 % for acceptable impact (i.e. cumulative impact) on predominant habitats could be used as a reliable objective towards ensuring sustainably used and healthy benthic ecosystems. This index-based target value may be coupled with an area-related target considering habitat area with no or low pressures. For special habitats, the target regarding the cumulative impact value should be less than 15 % in order to account for the importance of special protection. The indicator also provides the opportunity to set a GES target for marine protected areas (habitats in particular areas), which include both predominant and special habitats. A GES target for protected areas should be closely linked to conservation objectives and management actions. This could be a designated area without any human uses.

At this point, GES targets for the assessment of indicator 6.1.2 shall not yet be determined. Rather several options for targets shall be summarised for further discussions. Possible GES targets could be:

- a quantitative target based on the cumulative impact value, e.g. the cumulative impact value on predominant habitats must not exceed 15 %
- a dual quantitative target for area with no or low impact and for cumulative impacts, e.g. area with no / low impact must be at least 15 % and the cumulative impact value must not exceed 15 % for predominant habitats
- a stricter quantitative target for special habitats, e.g. the cumulative impact value on special habitats must not exceed 5 %
- a trend target for habitat types with major historical decline, e.g. area of habitat must increase towards historical extent
- a target considering habitats in particular areas (protected areas), e.g. 15 % of each habitat in Natura 2000 sites is without impacts from human activities

In order to achieve GES for indicator 6.1.2 and the German benthic ecosystems as a whole, all habitat types have to reach their individual target. This suggestion is based on the one out – all out principle which was first introduced by the Water Framework Directive. The one out – all out rule follows the precautionary principle and ensures that the seabed as a whole is in a Good Environmental Status.

3.8 Further development of the assessment concept

With the present report a methodology for the national assessment of indicator 6.1.2 is proposed and successfully applied with current data of the German EEZ of the North Sea. In addition, suggestions have been made for setting of baselines and GES targets. The assessment concept is already at an advanced stage so as to allow for a good estimation of physical impacts on benthic habitats. It is acknowledged that the proposed modelling concept is a pragmatic approach which includes in some parts several assumptions and uncertainties. In order to improve the results of future assessments the following enhancements of the concept are suggested:

- Improvement of sensitivity assessment with results of currently ongoing habitat mapping project by the Bundesamt für Naturschutz (BfN),
- validation of the assessment concept with levels of confidence,
- analysis of possible linking between indicator 6.1.2 and indicators associated with criteria 6.2 'condition of benthic habitats',
- development of a reference state for benthic habitats in the German North Sea: the sensitivity assessment of habitats should ideally be based on the reference state,
- calibration of assessment concept: either by integration of 'condition indicators' or by directly monitoring different levels of known human impact and
- modification and application of the assessment concept for coastal waters.

For further assessments it should as well be tried to improve data base, especially on fishing pressure and aggregate extraction. In order to achieve a conclusive result for the assessment of GES, data on human activities should cover the corresponding reporting period of six years.

In spite of these unresolved issues, the proposed methodology presents a major step for assessing cumulative physical impacts on benthic habitats. The concept provides a simple, cost-effective and informative method which is easily applicable to other marine regions. The approach also enables to determine if the Good Environmental Status is achieved and offers the knowledge base to implement management actions.

3.9 Annex: Sensitivity assessment of benthic habitats

3.9.1 Characteristic species for the sensitivity assessment

Rachor & Nehmer (2003) identify seven large-scale benthic communities in the German EEZ of the North Sea, which are mainly discriminated according to prevailing substrate. Figure 3-13 shows the spatial distribution of the benthic associations described by Rachor & Nehmer (2003).

Rachor & Nehmer (2003) differentiate between several benthic associations in sandy habitats, but their analyses do not describe a separate community in muddy habitats. Characteristic species for the habitat type 'sublittoral sand' can thus be found in the *Nucula-nitidosa*-, the *Amphiura-filiformis*- and the *Tellina-fabula*-association. A further benthic community on sand, the *Bathyporeia-Tellina*-association, settles exclusively on the sandbank Doggerbank and therefore the corresponding characteristic species can be used to assess sensitivity in this area. The areas defined as 'sublittoral mud' are mainly settled by the *Amphiura-filiformis*-association, so this community is used for the biological sensitivity assessment of mud habitats. Coarse sediments in the south-eastern North Sea are settled by the *Goniadella-Spisula*-association. Rachor & Nehmer (2003) differentiate two variations of this association in the EEZ, which correspond to the definition of 'sublittoral coarse sediment' (*Goniadella-Spisula*-association on medium and coarse sands) and the special habitat type 'species-rich habitats on coarse sands, gravel or shell debris' (*Goniadella-Spisula*-association on coarse sands and gravel). Table 3-21 lists predominant and special habitat types and the corresponding benthic communities and associated characteristic species identified by Rachor & Nehmer (2003), which are preliminarily used for the sensitivity assessment. For the criteria applied for the selection of characteristic species see also Rachor (2007). Rachor & Nehmer (2003) did not identify characteristic species for reef habitats, therefore the species list proposed by Nehls et al. (2008) is used. For the habitat 'species-rich habitats on coarse sands, gravel and shell debris' it is also referred to the mapping guidelines by the BfN (2011). Further information on the selection of characteristic species can be found in the respective chapters.

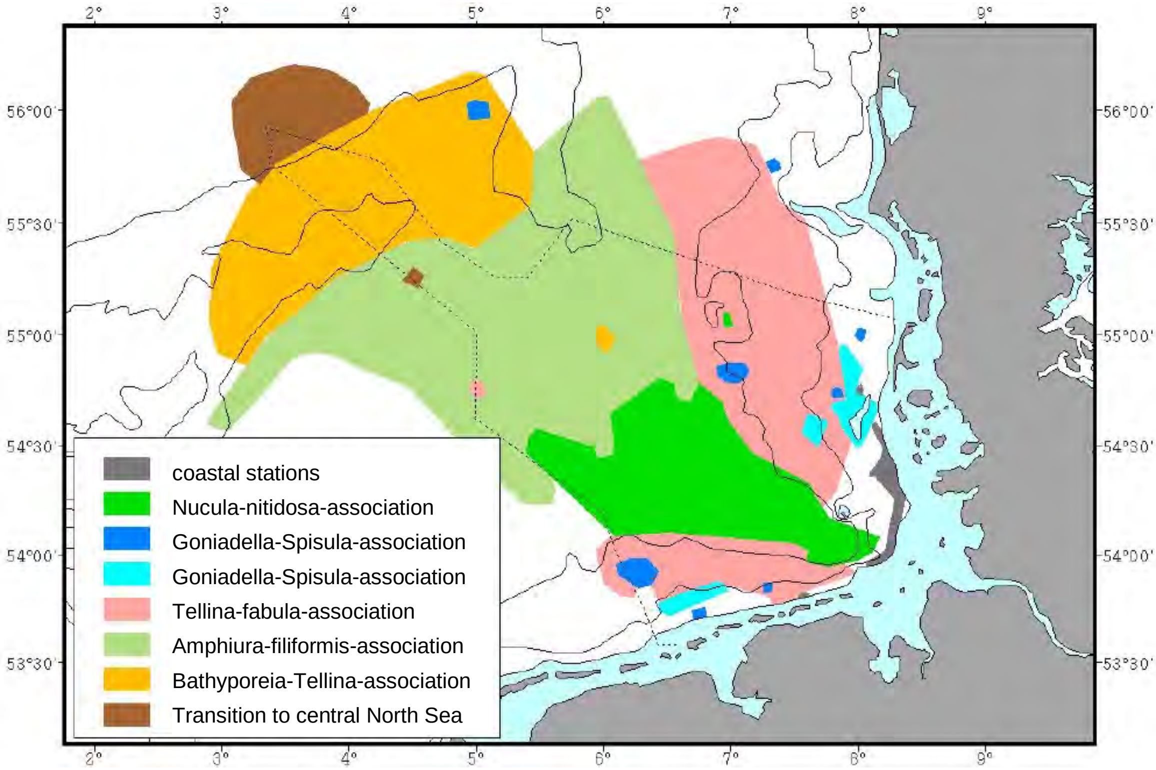

Figure 3-13: Spatial distribution of benthic assemblages in the German North Sea according to Rachor & Nehmer (2003).

Table 3-21: Habitat types in the German North Sea, the corresponding benthic associations according to Rachor & Nehmer (2003) and the characteristic species used for the sensitivity assessment

Habitat type	Benthic association	Characteristic species
Sublittoral sand	*Tellina-fabula*	*Magelona johnstoni* *Tellina fabula* *Urothoe poseidonis* *Bathyporeia guilliamsoniana*
	Nucula-nitidosa	*Nucula nitidosa* *Abra alba* *Scalibregma inflatum*
	Amphiura-filiformis	*Amphiura filiformis* *Mysella bidentata* *Harpinia antennaria* *Corbula gibba*
Sublittoral mud	*Amphiura-filiformis*	*Amphiura filiformis* *Mysella bidentata* *Harpinia antennaria*
Sublittoral coarse sedi-ment	*Goniadella-Spisula*	*Aonides paucibranchiata* *Ophelia limacina* *Thracia* spp.
	Goniadella-Spisula on coarse and medium sands	*Goodallia triangularis* *Spisula solida* *Angulus tenuis*
Sandbanks	*Bathyporeia-Tellina* (Doggerbank)	*Amphiura brachiata* *Spiophanes bombyx* *Lanice conchilega* *Bathyporeia spp.* *Cerianthus lloydii* *Tellina fabula* *Spio decoratus*
	Tellina-fabula (Borkum Reef Ground, Sylter Outer Reef)	*Magelona johnstoni* *Tellina fabula* *Urothoe poseidonis* *Bathyporeia guilliamsoniana*
Reefs	-	*Leucosolenia botryoides* *Alcyonium digitatum* *Pomatoceros triquiter* *Flustra foliacea* *Balanus crenatus* *Pholas dactylus* *Cancer pagurus* *Echinus esculentus* *Ciona intestinalis*
Species-rich habitats on coarse sands, gravel or shell debris	*Goniadella-Spisula* on coarse sands and gravel	*Aonides paucibranchiata* *Branchiostoma lanceolatum* *Echinocyamus pusillus* *Spisula elliptica* *Pisione remota*

3.9.2 Sublittoral sand

According to Rachor & Nehmer (2003), the sublittoral sand habitats in the German EEZ are inhabited by several benthic associations. Species used for the assessment are characteristic species identified for the *Tellina-fabula-*, the *Nucula-nitidosa-* and the *Amphiura-filiformis-*association. These associations are treated separately, however, the overall sensitivity rank does not differ between the communities.

3.9.2.1 Selective extraction

Table 3-22: Sensitivity of sublittoral sand towards the pressure ‘selective extraction’.

Selective extraction		Resistance	Recoverability	Sensitivity
Physical habitat		low	high	intermediate
Tellina fabula- association	*Tellina fabula*	low	moderate	intermediate
	Magelona johnstoni	low	high	intermediate
	Urothoe poseidonis	low	high	intermediate
	Bathyporeia guillamsioniana	low	high	intermediate
Habitat sensitivity sublittoral sand + *Tellina-fabula*-association				**intermediate**
Nucula-nitidosa--association	*Nucula nitidosa*	low	moderate	intermediate
	Abra alba	low	high	intermediate
	Scalibregma inflatum	low	high	intermediate
Habitat sensitivity sublittoral sand + *Nucula-nitidosa*-association				**intermediate**
Amphiura-filiformis-association	*Amphiura filiformis*	low	moderate	intermediate
	Mysella bidentata	low	high	intermediate
	Harpinia antennaria	low	not assessed	not assessed
	Corbula gibba	low	high	intermediate
Habitat sensitivity sublittoral sand + *Amphiura-filiformis*-association				**intermediate**

Physical habitat – explanatory notes

The extraction of sediment implies the complete removal of substrate by creating longitudinal tracks of generally 2-3 m width and up to 50 cm depth (trailer suction dredging) or rounded pits of around 10 m depth and with a diameter of 10-50 m (anchor dredging). Severe alterations of seabed topography and possibly also changes in sediment composition occur, therefore resistance to selective extraction is rated as low.

Physical seabed structures are supposed to have recovered when dredge tracks have disappeared and the original sediment composition is restored. Research on seabed recovery mostly focuses on observation of dredge furrows, while the recovery of sediment composition may take far longer but is less intense investigated. The disappearance of furrows may take place due to infilling where there is naturally high sediment transport or from dredging overflow. Existing furrows may also collapse or changed hydrodynamics may further erode dredge tracks. The infilling of furrows by fine sediment particles is associated with a decrease in sediment size and an increase in sediment instability and may thus prolong recovery time. Typical conditions for a fast recovery (months – 1 year) following extraction are high energy environments, fine sediments including sand, already disturbed communities and dominance of r-selected species, whereas slow recovery (years – decades) is predicted in moderate to low energy environments, with coarse sands, stable communities and a dominance of K-selected species. Additional factors influencing physical recovery are the method and intensity of dredging, the total area dredged and the extent of changes in sediment composition (Hill et al. 2011).

In the southern North Sea where tidal currents are generally strong, sand with a grain size up to 2 mm is mobile across the area during spring tides and may aid in the infilling of dredge tracks (Hill et al. 2011). Typical time-scales for the regeneration of dredge furrows in sandy substrates are in the range of months. In the German Baltic Sea in a shallow area of 8-10 m depth with fine to medium sands, furrows created by trailer suction dredging were observed to refill within months. In contrast, at another extraction site in the German Baltic Sea with fine sands in water depths between 14 and 21 m dredge tracks were still visible after ten years. At an extraction site west of Sylt stationary dredging was deployed creating pits of around 10 m depth and up to 2000 m in diameter. Bathymetric investigations revealed that only 10 % of the pits were refilled after cessation of dredging (ICES 2009).

Regarding the recoverability of sandy habitats in the areas licensed for extraction in the German North Sea considerable uncertainties remain. As investigation reports of the areas currently in use which could support the assessment are not available, the recovery time of sublittoral sand is preliminarily judged as high (1-2 years). The assessment is understood as precautionary, due to the sediment properties and the presumably moderate energy at the seabed, recovery of at least dredge tracks may as well be faster.

Characteristic species (*Tellina-fabula*-association) – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

The majority of species in the sublittoral sand is infaunal and would therefore be removed along with the substratum. Only some epifaunal and swimming species may be able to avoid the impact. The characteristic amphipods *Urothoe poseidonis* and *Bathyporeia guillamsioniana* settle the uppermost centimetres of sandy sediment and are thus also removed. Resident populations would be lost, so resistance for all characteristic species is assessed as low.

The bivalve *Tellina fabula* spawns at least once a year and has a protracted breeding period. The number of gametes is likely to be high with a larval phase of at least one month. The species therefore has high dispersal potential; however, post settlement development is not particularly rapid and the species may take two or more years to mature. Experimental data suggest that *Tellina fabula* would colonize available sediments in the year following environmental perturbation, but that a breeding population may take two or more years to establish. It is expected that full recovery would occur within five years and so recoverability is assessed as moderate.

The polychaete genus *Magelona* spp. displays characteristics typical of an r-selected species, i.e. rapid reproduction, short life span and high dispersal. The larval dispersal phase would potentially allow the species to colonize remote habitats. It is expected that populations of *Magelona* spp. would recover within two or three years and certainly within five years. Recoverability is therefore assessed as moderate.

Urothoe poseidonis is a small amphipod with moderate mobility which lives on the sediment surface and in shallow burrows. Sexual maturity is achieved at five month and a large number of reproductions with about 15 eggs per brood occur in a 15-day cycle during the breeding season between April and October. The genus thus has a relatively high fecundity and subsequent growth rate but a very limited dispersal potential (MES 2008). Recovery time is judged as high.

Repopulation of defaunated sediments by the amphipod *Bathyporeia* spp. is likely to be rapid. The genus is likely to have a high to very high capacity for recovery from many factors of disturbance. It is a short-lived genus which reaches maturity after six months and produces two generations within a year. There is no opportunity for larval dispersal as they are brooded, but adults are highly mobile in the water column and thus recovery potential is very high (MES 2008).

Characteristic species (*Nucula-nitidosa*-association) - explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

The majority of species in sublittoral coarse sediment is infaunal and would therefore be removed along with the substratum. Only some epifaunal and swimming species may be able to avoid the impact. Resident populations of the benthic endofauna would be lost, so resistance for all characteristic species is assessed as low.

The life-span of the bivalve *Nucula nitidosa* ranges from 7-10 years with 2-3 years to reach sexual maturity. *Nucula nitidosa* reproduces in high numbers, but has a limited dispersal potential as larvae settle in the vicinity of the adults. Long-distance dispersal is potentially poor. If a population is removed from an area, it may take a long time for the area to be recolonized, depending on the local hydrography. Recoverability is assessed as moderate.

Abra alba spawns at least twice a year over a protracted breeding period, during which time an average sized animal of 11 mm can produce between 15000 to 17000 eggs. Such egg production ensures successful replacement of the population, despite high larval mortality which is characteristic of planktonic development. Timing of spawning and settlement suggests that the larval planktonic phase lasts at least a month, in which time the larvae may be transported over a considerable distance. In addition to dispersal via the plankton, dispersal of post-settlement juveniles may occur via byssus drifting and probably bedload transport. Experimental data suggest that *Abra alba* would colonize available sediments within the year following environmental perturbation. Summer settled recruits may grow very rapidly and spawn in the autumn, whilst autumn recruits experience delayed growth and may not reach maturity until the following spring/summer. In the worst instance, a breeding population may take up to two years to fully establish and so recoverability has been assessed to be high.

Little is known of the longevity, egg size or fecundity of *Scalibregma inflatum*. The sexes are separate and there is one spawning between October-December after which the adults die. The reproductive epitoke stage is pelagic for a short time but there is no true larval stage (MES 2008). It is estimated that *Scalibregma inflatum* has a high recoverability.

Characteristic species (*Amphiura-filiformis*-association) - explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Breeding of *Amphiura filiformis* is annual and in the UK one period of recruitment occurs in the autumn. The larvae of this species can disperse over considerable distances due to their long planktonic existence. Adults, although mobile, are not highly active. Some immigration of adults from nearby populations may be possible. However, it can take approximately 5-6 years for *Amphiura filiformis* to grow to maturity so population structure may not return to original levels for at least this length of time. Several studies observed high mortality rates of new settling *Amphiura filiformis* and low rates of recruitment. Therefore, it seems likely that after removal of all or most of the population recovery will be determined by the presence of suitable hydrodynamic forces providing new larvae. Once settled the population is likely to take longer than five years to return to maturity and so recoverability has been suggested to be moderate.

The bivalve *Mysella bidentata* has a generation time of one year, a relatively high fecundity and a planktonic larval phase. It is estimated that recoverability is high.

Information on the amphipod *Harpinia antennaria* is currently not sufficient to assess sensitivity.

The life span for individuals of *Corbula gibba* is about 1-2 years. It has a rapid growth rate in the first few months of its life and the ability to survive in a wide range of environmental conditions and the capacity to achieve high population densities. *Corbula gibba* is known to be a pioneer species in recolonization of defaunated seabeds. The settling time of larvae is variable and may change depending on location and may take several months. In Danish waters there were high mortalities of newly settled individuals during the first month of settling. Overall it is likely that this species has good powers of population recovery. A population that is reduced in extent or abundance could potentially recover within a few years, depending on recruitment. Its ability to recolonize defaunated area suggests that the population would recover in a relatively short period of time even if the population was removed. Recoverability is judged to be high.

3.9.2.2 Abrasion

Table 3-23: Sensitivity of sublittoral sand towards the pressure ‘abrasion’.

Abrasion		Resistance	Recoverability	Sensitivity
Physical habitat		intermediate	very high	low
Tellina fabula-association	*Tellina fabula*	intermediate	high	low
	Magelona johnstoni	intermediate	high	low
	Urothoe poseidonis	intermediate	very high	low
	Bathyporeia guillamsioniana	tolerant	not relevant	not sensitive
Habitat sensitivity sublittoral sand + *Tellina-fabula*-association				**low**
Nucula-nitidosa-association	*Nucula nitidosa*	intermediate	high	low
	Abra alba	intermediate	very high	low
	Scalibregma inflatum	intermediate	high	low
Habitat sensitivity sublittoral sand + *Nucula-nitidosa*-association				**low**
Amphiura-filiformis-association	*Amphiura filiformis*	high	very high	very low
	Mysella bidentata	intermediate	high	low
	Harpinia antennaria	not assessed	not assessed	-
	Corbula gibba	intermediate	high	low
Habitat sensitivity sublittoral sand + *Amphiura-filiformis*-association				**low**

Physical habitat – explanatory notes

Impacts of fishing gears on sandy habitats include the removal of habitat complexity by flattening of biogenic structures or sand ripples, the penetration of sediment and smothering by resuspended sediment. Otter trawls generally disturb the upper 1-5 cm while beam trawls scour the sediment down to 8 cm (FAO 2004). Resistance towards abrasion is assessed as intermediate.

Physical restoration has been observed to be rapid (days to few months) in sandy habitats (Environment Agency 2010). In a study comparing the responses of various sediment types to physical disturbance, Dernie et al. (2003) found that clean sand communities had the most rapid recovery rate. Schwinghamer et al. (1996) examined the effect of otter trawls on habitats with fine and medium grained sand in the Grand Banks after trawling had stopped. The tracks left by the trawl doors were visible for at least ten weeks but not visible or only faintly visible after one year. Recoverability is therefore suggested to be very high.

Characteristic species (*Tellina-fabula*-association) – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Despite their robust body form, bivalves are vulnerable to physical abrasion. *Tellina fabula* is a shallow burrower with a fragile shell and may be damaged by an impact with fishing gear so resistance is recorded as intermediate. As presumably not the whole population is affected, recoverability is assessed as high.

Magelona spp. is a small polychaete which exposes its palps at the surface while feeding. The species lives infaunally in sandy sediment, usually within a few centimetres of the sediment surface. Physical disturbance, such as dredging or dragging an anchor, would be likely to penetrate the upper few centimetres of the sediment and cause physical damage to *Magelona* spp. Resistance is therefore recorded as intermediate. Due to the rapid reproduction, short life span and high dispersal potential of *Magelona*, recoverability is recorded as high.

The amphipod *Urothoe poseidonis* burrows in the upper centimetres of sediment. It has a moderate mobility and may therefore be affected by fishing gears. Resistance is assessed as intermediate. The genus has a relatively high fecundity and subsequent growth rate so that potential recovery time is judged as very high (MES 2008).

Bathyporeia spp. are highly mobile amphipod species so that they are unlikely to be damaged by abrasion. Therefore, *Bathyporeia guillamsioniana* has been assessed as tolerant.

Characteristic species (*Nucula-nitidosa*-association) – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Fishing for demersal species will disturb the surface layer of sediment and any protruding or shallow burrowing species. Even though the bivalve *Nucula nitidosa* has a small thick shell, it is probably vulnerable to physical damage from e.g. otter boards but its small size relative to the meshes of commercial trawls may ensure survival of at least a moderate proportion of disturbed individuals that pass through the nets. A manipulative field experiment in a fine muddy habitat reported a decline in the population density of *Nucula nitidosa* after five months of trawling disturbance, which remained significantly lower than the reference control area after ten months. Therefore, resistance has been assessed as intermediate as mortality may occur, and recoverability has been assessed as high. The life-span of *Nucula nitidosa* ranges from 7-10 years with 2-3 years to reach sexual maturity. *Nucula nitidosa* reproduces in high numbers, but has a limited dispersal potential as larvae settle in the vicinity of the adults. Overall, *Nucula nitidosa* is likely to exhibit good local, within-population recruitment. Therefore, if the extent of abundance of a population is reduced, recoverability is likely to be high.

The bivalve *Abra alba* is a shallow burrower with a fragile shell and may be damaged by physical impact. Bergmann & Santbrink (2000) reported between <0.5%and 18% mortality of *Abra alba* due to trawling in the southern North Sea, depending on the type of trawl (12 m or 6 m beam trawl or otter trawl). They included *Abra alba* amongst their list of bivalve species most vulnerable to trawling. Therefore, resistance has been assessed to be intermediate. The life history characteristics of *Abra alba* and its widespread distribution contribute to its powers of recoverability. *Abra alba* spawns at least twice a year over a protracted breeding period, during which time an average sized animal of 11 mm can produce between 15000 to 17000 eggs. Such egg production ensures successful replacement of the population, despite high larval mortality which is characteristic of planktonic development. Timing of spawning and settlement suggests that the larval planktonic phase lasts at least a month, in which time the larvae may be transported over a considerable distance. In addition to dispersal via the plankton, dispersal of post-settlement juveniles may occur via byssus drifting and probably bedload transport. Recoverability is likely to be very high in instances where a proportion of the adult population survives.

Scalibregma inflatum is a small to medium sized polychaete worm which burrows in sediment. Infaunal polychaetes with little mobility are likely to be damaged by abrasion and suffer some degree of mortality. Resistance is judged as intermediate. Little is known of the longevity, egg size or fecundity of this species. The sexes are separate and there is one spawning between October-December after which the adults die. The reproductive epitoke stage is pelagic for a short time but there is no true larval stage (MES 2008). Providing that part of the population survives, *Scalibregma inflatum* is likely to have a high recoverability.

Characteristic species (*Amphiura-filiformis*-association) - explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Brittlestars have fragile arms which are likely to be damaged by abrasion. *Amphiura filiformis* burrows in the sediment and extends only its arms when feeding. Literature reviews suggest that *Amphiura* spp. may be less susceptible to beam trawl damage than other species like echinoids or tube dwelling amphipods and polychaetes. Brittlestars can tolerate considerable damage to arms and even the disk without suffering mortality and are capable of arm and even some disk regeneration. Resistance to abrasion is therefore recorded as high. Individuals can still function whilst regenerating a limb so recovery will be rapid.

Due to their small size, the bivalve *Mysella bidentata* may escape damage from trawling although they may experience increased predation before re-burrowing. *Mysella bidentata* is often preferentially found in the structured irrigated burrows of host species such as *Amphiura filiformis* and if the top layers of sediment are ploughed this structure will be lost. Resistance has been assessed as intermediate. Recovery is likely to be high.

Information on the amphipod *Harpinia antennaria* is currently not sufficient to assess sensitivity.

The small solid shells of *Corbula gibba* may be vulnerable to physical damage (from e.g. otter boards) However, the size of *Corbula gibba* relative to the meshes of commercial trawls may ensure survival of a moderate proportion of disturbed individuals that pass through them. Specimens exposed on the sediment surface would be at risk of predation. Experimental trawling studies resulted in varying mortality rates. Therefore, a resistance of intermediate is recorded with a high recovery level.

3.9.2.3 Changes in siltation

Table 3-24: Sensitivity of sublittoral sand towards the pressure 'changes in siltation'.

Changes in siltation		Resistance	Recoverability	Sensitivity
Physical habitat		high	very high	very low
Tellina fabula-association	*Tellina fabula*	high	very high	very low
	Magelona johnstoni	high	very high	very low
	Urothoe poseidonis	high	very high	very low
	Bathyporeia guillamsioniana	high	very high	very low
Habitat sensitivity sublittoral sand + *Tellina-fabula*-association				**very low**
Nucula-nitidosa--association	*Nucula nitidosa*	high	very high	very low
	Abra alba	high	very high	very low
	Scalibregma inflatum	tolerant	not relevant	-
Habitat sensitivity sublittoral sand + *Nucula-nitidosa*-association				**very low**
Amphiura-filiformis-association	*Amphiura filiformis*	high	very high	very low
	Mysella bidentata	high	very high	very low
	Harpinia antennaria	not assessed	not assessed	-
	Corbula gibba	high	very high	very low
Habitat sensitivity sublittoral sand + *Amphiura-filiformis*-association				**very low**

Physical habitat – explanatory notes

Sediment plumes generated by construction works or aggregate extraction may cause changes in habitat structure such as infilling of small pits by fine sediments, siltation within crevices or development of migratory sand ripples (Hill et al. 2011). Finer sediment particles remain in suspension longer than larger particulates and can disperse over a wider area. Suspended fine and medium sands require a few hours for resettlement whereas silty sediments may remain in suspension for a few days (OSPAR 2008). In habitats with strong seabed transport recovery may be fast as fine sediments are rapidly mobilized. Resistance of sublittoral sand habitats is therefore regarded as high and recovery time as very high.

Characteristic species (*Tellina-fabula*-association) – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Tellina fabula is a shallow burrower in sandy sediments. It requires its inhalant siphon to be above the sediment surface for feeding and respiration. Smothering with 5 cm of sediment would temporarily halt feeding and respiration and requires the species to relocate to its preferred depth. *Tellina fabula* is an active burrower and would be expected to relocate with no mortality. However, growth and reproduction may be compromised and so resistance is assessed as high. Growth and reproduction would return to normal following relocation so recoverability is immediate.

Magelona spp. lives infaunally in fine sand and moves by burrowing. It deposit feeds at the surface by extending contractile palps from its burrow. An additional 5 cm layer of sediment would result in a temporary cessation of feeding activity, and therefore growth and reproduction are likely to be compromised. However, *Magelona* would be expected to quickly relocate to its favoured depth, with no mortality, and hence a high resistance is recorded. Once the animals have relocated to the surface, feeding activity should return to normal and therefore recoverability is suggested to be immediate.

Urothoe poseidonis is an amphipod burrowing in sediment which is likely to be able to accommodate deposition of sediment (MES 2008). The population may still suffer from reduced viability, so tolerance is assessed as high. Recoverability after smothering is assumed to be rapid.

The amphipod *Bathyporeia* spp. would probably be unaffected by an additional covering of sediment of a texture within its habitat preference, although there may be an energetic cost incurred by the additional burrowing activity required to attain a near-surface position for feeding and to swim. *Bathyporeia* spp. is likely to be more intolerant of smothering by both coarser and finer particles through which burrowing is likely to be hindered. Consequently, the resistance of *Bathyporeia guillamsoniana* to an increase in sedimentation has been assessed to be high. The species is likely to have a very high capacity for recovery.

Characteristic species (*Nucula-nitidosa*-association) - explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

The bivalve *Nucula nitidosa* can tolerate anaerobic conditions for several days and is able to thrive in poorly aerated sediments. It is therefore suggested that this ability to tolerate anaerobic conditions and their mobility allows them to survive when covered by sediments. Therefore, a high resistance has been recorded. Recoverability is assumed to be very high.

Abra alba is a shallow burrower in muddy sediments. It requires its inhalant siphon to be above the sediment surface for feeding and respiration. Sudden smothering with 5 cm of sediment would temporarily halt feeding and respiration and requires the species to relocate to its preferred depth. As an active burrower *Abra alba* would be expected to relocate with no mortality. However, growth and reproduction may be compromised owing to energetic expenditure and so resistance has been assessed to be high. Growth and reproduction would return to normal following relocation so recoverability is recorded as very high.

The polychaete *Scalibregma inflatum* burrows in sediment and is a sub-surface deposit feeder exploiting detritus (MES 2008). Therefore, the species is suggested to be tolerant of smothering.

Characteristic species (*Amphiura-filiformis*-association) - explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Amphiura filiformis is an infaunal species which can burrow and lives up to a depth of 4 cm within the sediment. Therefore, smothering by sediment of 5 cm is unlikely to have great effect although feeding and hence viability of the population may be reduced if the sediment is particularly fine and mobile. Since only sub-lethal effects are likely resistance is considered to be high. Recovery is likely to be rapid as individuals move up through the sediment to resume their position for feeding and any fine particles are removed.

The suspension feeding bivalve *Mysella bidentata* is capable of burrowing and unlikely to be significantly affected by the addition of 5 cm of sediment, providing the sediment was of similar consistency to the existing sediment. As the viability of the population may be reduced due to temporary cessation of feeding activity and additional energetic costs of relocation, resistance is assessed as high. Recoverability is likely to be rapid.

Information on the amphipod *Harpinia antennaria* is currently not sufficient to assess sensitivity.

Corbula gibba is a burrower in shallow muddy or sandy sediments and uses a byssus thread to attach to pieces of shell or rock in the sediment. It uses its short inhalant siphon above the sediment for feeding and respiration. If smothered *Corbula gibba* would most likely burrow up through the new sediment. *Corbula gibba* is also considered to be generally tolerant of prolonged oxygen deprivation. Laboratory studies on *Corbula gibba* have shown that they can survive up to 57 days in near anoxic conditions. However, sudden smothering of the sediment would halt feeding. Therefore, resistance has been assessed as high with an immediate recoverability level.

3.9.3 Sublittoral mud

3.9.3.1 Selective extraction

Selective extraction is a pressure currently not relevant in the sublittoral mud habitats of the North Sea EEZ. The human activity associated with selective extraction in offshore areas is aggregate extraction, which affects only sand and gravel habitats.

3.9.3.2 Abrasion

Table 3-25: Sensitivity of sublittoral mud towards the pressure 'abrasion'.

Abrasion		Resistance	Recoverability	Sensitivity
Physical habitat		intermediate	high	low
Characteristic species	*Amphiura filiformis*	high	very high	very low
	Mysella bidentata	intermediate	high	low
	Harpinia antennaria	not assessed	not assessed	-
	Corbula gibba	intermediate	high	low
Habitat sensitivity				**low**

Physical habitat – explanatory notes

Towed demersal gears have been shown to alter the sedimentary characteristics of subtidal muddy sand/mud habitats by penetration of the sediment. Trawling alters the physical environment of the benthos by creating furrows or scar from trawl doors, scouring and flattening the seabed with ground rope and weights, and redistributing sediment and other material (Environment Agency 2010). Trawl doors may cause furrows of up to 20 cm deep depending on the door weight and the hardness of the sediment (FAO 2004). The resistance of sublittoral mud towards abrasion has therefore been assessed as intermediate. Trawl marks are likely to last longer in sheltered areas with fine sediments. Pits at muddier sites generally take longer to infill (and thus had less negative infilling rates) than those in sandier sites. Muddy sands were found to be very vulnerable to the impacts of fishing activities, with recovery times predicted to take from several months to years (Environment Agency 2010). The same trawl track could be identified for almost five years in a sandy mud area in Kiel Bay that is not exposed to tidal currents (FAO 2004). This long recovery time is due to the fact that mud habitats are mediated by a combination of physical, chemical and biological processes (compared to sand habitats that are dominated by physical processes) (Environment Agency 2010). Due to the prevailing hydrographical conditions in the muddy areas of the German EEZ, recoverability of mud habitats is estimated as high.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Brittlestars have fragile arms which are likely to be damaged by abrasion. *Amphiura filiformis* burrows in the sediment and extends only its arms when feeding. Literature reviews suggest that *Amphiura* spp. may be less susceptible to beam trawl damage than other species like echinoids or tube dwelling amphipods and polychaetes. Brittlestars can tolerate considerable damage to arms and even the disk without suffering mortality and are capable of arm and even some disk regeneration. Resistance to abrasion is therefore recorded as high. Individuals can still function whilst regenerating a limb so recovery will be rapid.

Due to their small size, the bivalve *Mysella bidentata* may escape damage from trawling although they may experience increased predation before re-burrowing. *Mysella bidentata* is often preferentially found in the structured irrigated burrows of host species such as *Amphiura filiformis* and if the top layers of sediment are ploughed this structure will be lost. Resistance has been assessed as intermediate. Recovery is likely to be high.

Information on the amphipod *Harpinia antennaria* is currently not sufficient to assess sensitivity.

The small solid shells of *Corbula gibba* may be vulnerable to physical damage (from e.g. otter boards) However, the size of *Corbula gibba* relative to the meshes of commercial trawls may ensure survival of a moderate proportion of disturbed individuals that pass through them. Specimens exposed on the sediment surface would be at risk of predation. Experimental trawling studies resulted in varying mortality rates. Therefore, a resistance of intermediate is recorded with a high recovery level.

3.9.3.3 Changes in siltation

Table 3-26: Sensitivity of sublittoral sand towards the pressure ‘changes in siltation’.

Changes in siltation		Resistance	Recoverability	Sensitivity
Physical habitat		high	very high	very low
Characteristic species	*Amphiura filiformis*	high	very high	very low
	Mysella bidentata	high	very high	very low
	Harpinia antennaria	not assessed	not assessed	-
	Corbula gibba	high	very high	very low
Habitat sensitivity				**very low**

Physical habitat – explanatory notes

Increased sedimentation mostly involves fine sediment particles which are similar to substrate size in sublittoral mud habitats. Therefore, effects on habitat structure and benthic communities are assumed to be only small-scale. Resistance is judged to be high and recoverability is assumed to be very high.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Amphiura filiformis is an infaunal species which can burrow and lives up to a depth of 4 cm within the sediment. Therefore, smothering by sediment of 5 cm is unlikely to have great effect although feeding and hence viability of the population may be reduced if the sediment is particularly fine and mobile. Since only sub-lethal effects are likely resistance is considered to be high. Recovery is likely to be rapid as individuals move up through the sediment to resume their position for feeding and any fine particles are removed.

The suspension feeding bivalve *Mysella bidentata* is capable of burrowing and unlikely to be significantly affected by the addition of 5 cm of sediment, providing the sediment was of similar consistency to the existing sediment. As the viability of the population may be reduced due to temporary cessation of feeding activity and additional energetic costs of relocation, resistance is assessed as high. Recoverability is likely to be rapid.

Information on the amphipod *Harpinia antennaria* is currently not sufficient to assess sensitivity.

Corbula gibba is a burrower in shallow muddy or sandy sediments and uses a byssus thread to attach to pieces of shell or rock in the sediment. It uses its short inhalant siphon above the sediment for feeding and respiration. If smothered *Corbula gibba* would most likely burrow up through the new sediment. *Corbula gibba* is also considered to be generally tolerant of prolonged oxygen deprivation. Laboratory studies on *Corbula gibba* have shown that they can survive up to 57 days in near anoxic conditions. However, sudden smothering of the sediment would halt feeding. Therefore, resistance has been assessed as high with an immediate recoverability level.

3.9.4 Sublittoral coarse sediment

3.9.4.1 Selective extraction

Table 3-27: Sensitivity of sublittoral coarse sediment towards the pressure 'selective extraction'.

Selective extraction		Resistance	Recoverability	Sensitivity
Physical habitat		low	moderate	intermediate
Characteristic species	*Aonides paucibranchiata*	low	moderate	intermediate
	Ophelia limacina	low	moderate	intermediate
	Thracia spp.	low	moderate	intermediate
	Goodallia triangularis	not assessed	not assessed	not assessed
	Spisula solida	low	high	intermediate
	Angulus tenuis	low	moderate	intermediate
Habitat sensitivity				**intermediate**

Physical habitat – explanatory notes

The extraction of sediment implies the complete removal of substrate by creating longitudinal tracks of generally 2-3 m width and up to 50 cm depth (trailer suction dredging) or rounded pits of around 10 m depth and with a diameter of 10-50 m (anchor dredging). Severe alterations of seabed topography and possibly also changes in sediment composition occur, therefore resistance to selective extraction is rated as low.

Physical seabed structures are supposed to have recovered when dredge tracks have disappeared and the original sediment composition is restored. Research on seabed recovery mostly focuses on observation of dredge furrows, while the recovery of sediment composition may take far longer but is less intense investigated. Recovery takes the longest period of time at dredge sites characterised by coarse sediments (Hill et al. 2011). Observations from studies conducted in sandy gravel sediments reveal that the morphological behaviour of dredged tracks and pits varies significantly. In an area exposed to long-period waves, dredge tracks 0.3 – 0.5 m deep, in a gravelly substrate at a depth of 38 m, were found to disappear completely within eight months. In contrast, at an experimental dredged gravel site off Norfolk, UK, in 25 m of water, dredge tracks appeared to have been completely eroded well within three years of the cessation of dredging. Erosion of dredge tracks in areas of moderate wave exposure and tidal currents have been observed to take from three to more than seven years in gravelly sediments. In the latter case, however, infill resulted mainly from sand in transport. Especially in coarse sediments, the refill material may be finer grained than the material on the surrounding seabed, which could lead to a permanent change in benthic communities (Herrmann & Krause 1998). In the southern North Sea where tidal currents are generally strong, sand with a grain size up to 2 mm is mobile across the area during spring tides (Hill et al. 2011). Therefore, it is assumed that recovery of coarse sediments after cessation of dredging is principally possible, but may take a few years. Recoverability is thus estimated as moderate.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

The majority of species in sublittoral coarse sediment is infaunal and would therefore be removed along with the substratum. Only some epifaunal and swimming species may be able to avoid the impact. Resident populations of the benthic endofauna would be lost, so resistance for all characteristic species is assessed as low.

Aonides paucibranchiata is a small-sized polychaete with limited mobility. The fecundity and dispersal potential of this genus is low (larval duration 2-10 days), so recolonisation from sources outside a disturbed area is likely to be slow. Recoverability is estimated to be moderate (MES 2008).

The life-span of *Ophelia limacina* is 6-10 years and adults mature at 1-2 years. The sexes are separate and eggs are fertilised externally after spawning in July-August. The duration of the larval stage is 2-10 days with settlement occurring between June and November. Little is known of the fecundity of this genus, but the relatively short planktonic phase and long life-span of the adult suggests an intermediate potential for recolonisation and subsequent recovery of biomass (MES 2008).

It is not possible to estimate the regeneration and dispersal potential of *Thracia* spp., but the genus is long-lived (>10 years) and slow-growing and probably has a relatively low recoverability following disturbance (MES 2008). It is estimated that recovery will last more than two years but will be completed within ten years. Therefore, recoverability is judged to be moderate.

Information on the bivalve *Goodallia triangularis* is currently not sufficient to assess sensitivity.

The bivalve *Spisula solida* can live up to ten years. Individuals are sexually mature at 1 year, regardless of their size. The sexes of *Spisula* are separate and both show a synchrony in gametogenic development and spawning. Gametogenesis starts in September when temperatures decrease and spawning begins in February. Larvae can remain in the water column for several weeks, allowing fairly wide dispersal. The potential recovery of this bivalve is high and is often recorded amongst the first colonizers of sediments disturbed by dredging.

Little information is available on biological traits of *Angulus tenuis*, therefore sensitivity of the closely related species *Angulus (Tellina) fabula* is used as reference: The bivalve *Tellina fabula* spawns at least once a year and has a protracted breeding period. The number of gametes is likely to be high with a larval phase of at least one month. The species therefore has high dispersal potential; however, post settlement development is not particularly rapid and the species may take two or more years to mature. Experimental data suggest that *Tellina fabula* would colonize available sediments in the year following environmental perturbation, but that a breeding population may take two or more years to establish. It is expected that full recovery would occur within five years and so recoverability is assessed as moderate.

3.9.4.2 Abrasion

Table 3-28: Sensitivity of sublittoral coarse sediment towards the pressure ‘abrasion’.

Abrasion		Resistance	Recoverability	Sensitivity
Physical habitat		intermediate	very high	low
Characteristic species	*Aonides paucibranchiata*	intermediate	high	low
	Ophelia limacina	intermediate	moderate	intermediate
	Thracia spp.	intermediate	high	low
	Goodallia triangularis	not assessed	not assessed	not assessed
	Spisula solida	intermediate	high	low
	Angulus tenuis	intermediate	high	low
Habitat sensitivity				**intermediate**

Physical habitat – explanatory notes

Impacts of fishing gears on habitats with coarse sands include the smoothing of the seafloor by flattening of biogenic structures or sand ripples, the penetration of sediment, smothering by resuspended sediment and displaced or overturned gravel (Environment Agency 2010). Otter trawls generally disturb the upper 1-5 cm while beam trawls scour the sediment down to 8 cm (FAO 2004). Resistance towards abrasion is assessed as intermediate.

Recovery time in gravel habitats has been predicted to be in the order of ten years, while physical restoration of sandy habitats has been observed to be rapid (days to few months) (Environment Agency 2010). The visible dredge marks from towed gear have been shown to be relatively short lived, lasting no more than a year in coarse sediments. As the habitat regard here predominantly consists of medium to coarse sands, recoverability is judged to be very high.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Little is known about the life history of the polychaete worm *Aonides paucibranchiata* but its size and morphology suggest that it is likely to be vulnerable to physical disturbance. Infaunal polychaetes with little mobility are likely to be damaged by abrasion and suffer some degree of mortality. Resistance is judged as intermediate. As a short-lived animal with small body size, it is likely to recover adult biomass relatively quickly following colonisation by juveniles (MES 2008). Providing that part of the population survives, *Aonides paucibranchiata* is likely to have a high recoverability.

As an infaunal surface deposit feeder, part of the *Ophelia limacina* population is likely to be damaged or killed by a trawling event. Resistance is therefore assumed to be intermediate. The relatively short planktonic phase and therefore low dispersal potential and long life-span of the adult suggests the recoverability to be moderate.

The bivalve genus *Thracia* spp. burrows deeply in coarse sands and fine gravels and may thus escape the passing of a trawl. However, some individuals, especially juveniles, may suffer damage and may even be killed. Therefore, resistance is estimated as intermediate. It is not possible to estimate the regeneration and dispersal potential of *Thracia* spp., but the genus is long-lived (>10 years) and slow-growing and probably has a relatively low recoverability following disturbance (MES 2008). As it is assumed that only a small part of the population is damaged, recoverability is estimated to be high.

Information on the bivalve *Goodallia triangularis* is currently not sufficient to assess sensitivity.

Fishing for demersal species will disturb the surface layer of sediment and any protruding or shallow burrowing species. Experimental trawls showed that 93% of the uncaught *Spisula solida* were undamaged, as they were well protected by their thick shells, and only 1% died. The impacts caused by a fishing dredge significantly increased the number of exposed *Spisula solida* clams and the abundance of potential predators. The impact of the dredge increased the time needed for *Spisula solida* to rebury, which rendered them vulnerable to predation for longer periods. Resistance has been assessed as intermediate as mortality may occur and recoverability has been assessed as high.

Little information is available on biological traits of *Angulus tenuis*, therefore sensitivity of the closely related species *Angulus (Tellina) fabula* is used as reference: Despite their robust body form, bivalves are vulnerable to physical abrasion. *Tellina fabula* is a shallow burrower with a fragile shell and may be damaged by an impact with fishing gear so resistance is recorded as intermediate. As presumably not the whole population is affected, recoverability is assessed as high.

3.9.4.3 Changes in siltation

Table 3-29: Sensitivity of sublittoral coarse sediment towards the pressure 'changes in siltation'.

Changes in siltation		Resistance	Recoverability	Sensitivity
Physical habitat		intermediate	very high	low
Characteristic species	*Aonides paucibranchiata*	high	very high	very low
	Ophelia limacina	high	very high	very low
	Thracia spp.	intermediate	high	low
	Goodallia triangularis	not assessed	not assessed	not assessed
	Spisula solida	intermediate	high	low
	Angulus tenuis	high	very high	very low
Habitat sensitivity				**low**

Physical habitat – explanatory notes

Sediment plumes generated by construction works or aggregate extraction may cause changes in habitat structure such as infilling of small pits by fine sediments, siltation within crevices or development of migratory sand ripples (Hill et al. 2011). Finer sediment particles remain in suspension longer than larger particulates and can disperse over a wider area. Suspended fine and medium sands require a few hours for resettlement whereas silty sediments may remain in suspension for a few days (OSPAR 2008). As suspended particles tend to be significantly finer than the prevailing coarse sands, changes in sediment composition are supposed to be more distinct than e.g. in mud habitats. Resistance of sublittoral coarse sediment is therefore regarded as intermediate. Recovery is dependent on seabed transport, wave and tidal energy. It is estimated to be very high in coarse sediments.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Aonides paucibranchiata is a small deposit feeding polychaete with limited mobility. The species lives in a loosely constructed tube or is free-living (MES 2008). An additional 5 cm layer of sediment would result in a temporary cessation of feeding activity, and therefore growth and reproduction are likely to be compromised. However, spionids would be expected to quickly relocate to its favoured depth, with no mortality, and hence a high resistance is recorded. Recoverability will probably be very high.

The polychaete *Ophelia limacina* reaches 3-10 cm in length and burrows in unconsolidated mixed to medium coarse sands where it is a deposit-feeder exploiting diatoms & detritus within the sediments. *Ophelia* has moderate mobility within the surface deposits, and is likely to be able to accommodate moderate deposition of sediment (MES 2008). Resistance is judged to be high and recoverability very high.

The bivalve *Thracia* spp. is a deep-burrowing form that lives in sand, gravel and mud where it lives as a suspension-feeder on phytoplankton and detritus in the water column. It has very limited mobility and therefore it cannot be excluded that some mortality of individuals may occur. Resistance is therefore precautionarily estimated as intermediate and recovery as high.

Information on the bivalve *Goodallia triangularis* is currently not sufficient to assess sensitivity.

Spisula solida is a fast burrowing bivalve and suspension feeder. If *Spisula solida* were covered by sediments it would be able to reposition itself within the sediment. Fahy et al. (2003) noted that in a clam bed in Ireland, where part of the bed has silted up, numbers of *Spisula solida* and the size of the clam patch were reduced. Therefore, resistance has been assessed as intermediate to reflect the reduction in the size of the clam bed and *Spisula* numbers. Recoverability is assessed as high.

Little information is available on biological traits of *Angulus tenuis*, therefore sensitivity of the closely related species *Angulus (Tellina) fabula* is used as reference: *Tellina fabula* is a shallow burrower in sandy sediments. It requires its inhalant siphon to be above the sediment surface for feeding and respiration. Smothering with 5 cm of sediment would temporarily halt feeding and respiration and requires the species to relocate to its preferred depth. *Tellina fabula* is an active burrower and would be expected to relocate with no mortality. However, growth and reproduction may be compromised and so resistance is assessed as high. Growth and reproduction would return to normal following relocation so recoverability is immediate.

3.9.5 Sandbanks

3.9.5.1 Definition of sandbanks

Sandbanks are elevated, elongated, rounded or irregular topographic features, permanently submerged and predominantly surrounded by deeper water. They consist mainly of sandy sediments, but larger grain sizes, including boulders and cobbles, or smaller grain sizes including mud may also be present on a sandbank (EC 2013)

Sandbanks of notable size in the German North Sea include the Dogger Bank and the smaller Amrum Outer Ground. The Borkum Reef Ground is an example of a sandbank with cobble fields and stony or gravelly areas constituting reef-like structures.

3.9.5.2 Characteristic species

In the German EEZ four sandbank areas have been identified. These vary according to prevailing hydrological and sediment conditions, thus producing different benthic communities. It is not possible to use a uniform list of characteristic species for all sandbank locations. Sandbanks in the Borkum Reef Ground and the Sylter Outer Reef predominantly consist of fine sands with the *Tellina fabula*-association with small areas of coarse sands and reefs. Therefore, it is proposed that for the sensitivity assessment ranks assigned to the predominant habitat ‘sublittorals sand’ with the *Tellina-fabula*-association are used.

The sandbank on the Doggerbank is characterized by fine sediments and the *Bathyporeia-Tellina*-association. Characterising species for this habitat according to Rachor & Nehmer (2003) are: *Spiophanes bombyx*, *Lanice conchilega*, *Bathyporeia elegans*, *Amphiura brachiata*, *Cerianthus loydii*, *Tellina fabula*, *Bathyporeia nana* and *Spio decorata.* The only species which fulfills all criteria for a characteristic species is the brittlestar *Amphiura brachiata*. In order to have a more comprehensive assessment for the habitat, the other characterizing species were also used for the determination of the sensitivity rank for the habitat.

3.9.5.3 Selective extraction

Table 3-30: Sensitivity of sandbanks (Doggerbank) towards the pressure 'selective extraction'.

Selective extraction		Resistance	Recoverability	Sensitivity
Physical habitat		low	high	intermediate
Characteristic species	*Spiophanes bombyx*	low	high	intermediate
	Lanice conchilega	low	high	intermediate
	Bathyporeia spp.	low	high	intermediate
	Amphiura brachiata	low	moderate	intermediate
	Cerianthus lloydii	low	moderate	intermediate
	Tellina fabula	low	moderate	intermediate
	Spio decoratus	low	high	intermediate
Habitat sensitivity				**intermediate**

Physical habitat – explanatory notes

The extraction of sediment implies the complete removal of substrate by creating longitudinal tracks of generally 2-3 m width and up to 50 cm depth (trailer suction dredging) or rounded pits of around 10 m depth and with a diameter of 10-50 m (anchor dredging). Severe alterations of seabed topography and possibly also changes in sediment composition occur, therefore resistance to selective extraction is rated as low.

Physical seabed structures are supposed to have recovered when dredge tracks have disappeared and the original sediment composition is restored. Research on seabed recovery mostly focuses on observation of dredge furrows, while the recovery of sediment composition may take far longer but is less intense investigated. The disappearance of furrows may take place due to infilling where there is naturally high sediment transport or from dredging overflow. Existing furrows may also collapse or changed hydrodynamics may further erode dredge tracks. The infilling of furrows by fine sediment particles is associated with a decrease in sediment size and an increase in sediment instability and may thus prolong recovery time. Typical conditions for a fast recovery (months – 1 year) following extraction are high energy environments, fine sediments including sand, already disturbed communities and dominance of r-selected species, whereas slow recovery (years – decades) is predicted in moderate to low energy environments, with coarse sands, stable communities and a dominance of K-selected species. Additional factors influencing physical recovery are the method and intensity of dredging, the total area dredged and the extent of changes in sediment composition (Hill et al. 2011).

In the southern North Sea where tidal currents are generally strong, sand with a grain size up to 2 mm is mobile across the area during spring tides and may aid in the infilling of dredge tracks (Hill et al. 2011). Typical time-scales for the regeneration of dredge furrows in sandy substrates are in the range of months. In the German Baltic Sea in a shallow area of 8-10 m depth with fine to medium sands, furrows created by trailer suction dredging were observed to refill within months. In contrast, at another extraction site in the German Baltic Sea with fine sands in water depths between 14 and 21 m dredge tracks were still visible after ten years. At an extraction site west of Sylt stationary dredging was deployed creating pits of around

10 m depth and up to 2000 m in diameter. Bathymetric investigations revealed that only 10 % of the pits were refilled after cessation of dredging (ICES 2009).

Regarding the recoverability of sandy habitats in the areas licensed for extraction in the German North Sea considerable uncertainties remain. As investigation reports of the areas currently in use which could support the assessment are not available, the recovery time of sandbanks is preliminarily judged as high (1-2 years). The assessment is understood as precautionary, due to the sediment properties and the presumably moderate energy at the seabed, recovery of at least dredge tracks may as well be faster.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

The majority of species in the sublittoral sand is infaunal and would therefore be removed along with the substratum. Only some epifaunal and swimming species may be able to avoid the impact. The characteristic amphipods *Bathyporeia elegans* and *B. nana* settle the uppermost centimetres of sandy sediment and are thus also removed. Resident populations would be lost, so resistance for all characteristic species is assessed as low.

The polychaete *Spiophanes bombyx* is regarded as a typical 'r' selecting species with a short life span, high dispersal potential and high reproductive rate. It is often found at the early successional stages of variable, unstable habitats that it is quick to colonize following perturbation. Its larval dispersal phase may allow the species to colonize remote habitats. Recoverability is therefore estimated as high.

The sand mason *Lanice conchilega* lives for about 1 year at which point reproduction occurs between April-June. The female releases around 160,000 eggs and these are fertilised at the sediment surface. The larva spends about 8 weeks in a planktotrophic phase during which time a proto-tube develops before the post-larva sinks to the seabed. It has a capacity to disperse over considerable distances and can be found in dense communities. The relatively short life-span suggests that restoration of the biomass is achieved within one year following initial recolonisation by the juveniles. This species has a high recoverability.

Repopulation of defaunated sediments by the amphipod *Bathyporeia* spp. is likely to be rapid. The genus is likely to have a high to very high capacity for recovery from many factors of disturbance. It is a short-lived genus which reaches maturity after six months and produces two generations within a year. There is no opportunity for larval dispersal as they are brooded, but adults are highly mobile in the water column and thus recovery potential is high (MES 2008).

The genus *Amphiura* is a relatively long-lived and slow-growing brittlestar with a life-span of 10 to 20 years. Breeding is annual and larvae can disperse over considerable distances due to their long planktonic existence. Adults, although mobile, are not highly active. Some immigration of adults from nearby populations may be possible. However, it can take approximately 5-6 years for *Amphiura* to grow to maturity so population structure may not return to original levels for at least this length of time. Therefore, it seems likely that after removal of all or most of the population recovery will be determined by the presence of suitable hydrodynamic forces providing new larvae. Once settled the population is likely to take longer than five years to return to maturity and so recoverability of *Amphiura brachiata* has been suggested to be moderate.

The tubiculous sea anemone *Cerianthus lloydii* is a long-lived anemone with a life-span of as much as 11-20 years. The age at sexual maturity and fecundity is unknown. Fertilisation is external fertilisation and the larvae are pelagic. The dispersal potential may therefore be high, although without information on the fecundity, it is not possible to estimate the recolonisation potential for this genus. The long life-span and slow growth of this anemone suggests that it has a low rate of restoration of the biomass following recolonisation. Recoverability is estimated as moderate.

The bivalve *Tellina fabula* spawns at least once a year and has a protracted breeding period. The number of gametes is likely to be high with a larval phase of at least one month. The species therefore has high dispersal potential; however, post settlement development is not particularly rapid and the species may take two or more years to mature. Experimental data suggest that *Tellina fabula* would colonize available sediments in the year following environmental perturbation, but that a breeding population may take two or more years to establish. It is expected that full recovery would occur within five years and so recoverability is assessed as moderate.

Spio is a short-lived genus with a life-span of about one year. Sexual maturity is achieved at 2-3 months. The sexes are separate and approximately 250 eggs are fertilised externally during two reproductive periods (April-June & August-September). The embryos are brooded in the tube and then released as lecithotrophic larvae that spend about 4 weeks in the plankton. Settlement is from June-August. The dispersal potential is high and the relatively short generation time and rapid growth rate suggests that restoration of the biomass is achieved soon after settlement. This genus has a high recoverability.

3.9.5.4 Abrasion

Table 3-31: Sensitivity of sandbanks (Doggerbank) towards the pressure ‘abrasion’.

Abrasion		Resistance	Recoverability	Sensitivity
Physical habitat		intermediate	very high	low
Characteristic species	*Spiophanes bombyx*	intermediate	very high	low
	Lanice conchilega	intermediate	very high	low
	Bathyporeia spp.	tolerant	not relevant	not sensitive
	Amphiura brachiata	high	very high	very low
	Cerianthus lloydii	intermediate	moderate	intermediate
	Tellina fabula	intermediate	high	low
	Spio decoratus	intermediate	very high	low
Habitat sensitivity				**intermediate**

Physical habitat – explanatory notes

Impacts of fishing gears on sandy habitats include the removal of habitat complexity by flattening of biogenic structures or sand ripples, the penetration of sediment and smothering by resuspended sediment. Otter trawls generally disturb the upper 1-5 cm while beam trawls scour the sediment down to 8 cm (FAO 2004). Resistance towards abrasion is assessed as intermediate.

Physical restoration has been observed to be rapid (days to few months) in sandy habitats (Environment Agency 2010). In a study comparing the responses of various sediment types to physical disturbance, Dernie et al. (2003) found that clean sand communities had the most rapid recovery rate. Schwinghamer et al. (1996) examined the effect of otter trawls on habitats with fine and medium grained sand in the Grand

Banks after trawling had stopped. The tracks left by the trawl doors were visible for at least ten weeks but not visible or only faintly visible after one year. Recoverability is therefore suggested to be very high.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Spiophanes bombyx is a soft bodied organism that exposes its palps at the surface while feeding. It lives infaunally in sandy sediment and any physical disturbance that penetrates the sediment, for example dredging or dragging an anchor, would lead to physical damage of *Spiophanes bombyx*. Bergman & Hup (1992) reported a 40-60% decrease in the total density of *Spiophanes bombyx* after 3 trawling events. Therefore, a resistance of intermediate has been recorded. Jennings & Kaiser (1995) suggested that the top few centimetres of the sediment were usually occupied by opportunistic species, such as spionids, capitellid polychaetes and amphipods, which were able to recolonize disturbed areas quickly. They further suggested that this surface community would probably recover within 6 -12 months. Therefore, a recoverability of very high has been recorded.

Lanice conchilega is a medium-large polychaete worm belonging to the Family Terebellidae. It reaches a length of 25-30cm and forms a characteristic tube of sand-grains ending at the head end in a tuft of sandy filaments that project from the surface of the sediment. It is likely that the species is damaged and killed by abrasion. Therefore, resistance is assessed as intermediate. Due to their high reproductive and larval dispersal potential, recoverability is estimated to be very high.

Bathyporeia spp. are highly mobile amphipod species so that they are unlikely to be damaged by abrasion. Therefore, resistance has been assessed as tolerant.

Brittlestars have fragile arms which are likely to be damaged by abrasion. *Amphiura* spp. burrows in the sediment and extends only its arms when feeding. Literature reviews suggest that *Amphiura* spp. may be less susceptible to beam trawl damage than other species like echinoids or tube dwelling amphipods and polychaetes. Brittlestars can tolerate considerable damage to arms and even the disk without suffering mortality and are capable of arm and even some disk regeneration. Resistance to abrasion is therefore recorded as high. Individuals can still function whilst regenerating a limb so recovery will be rapid.

Cerianthus lloydii is a brownish, tube-dwelling anemone up to 15 cm long. The mouth and tentacles project above the surface of the sand from the soft tube, which can be up to 40 cm long and is permanently buried. It is able to retract rapidly into the tube to avoid physical disturbance. Withdrawn burrowing anemones are likely to reappear and dislodged individuals reburrow. However, it cannot be ruled out that some individuals may be damaged by trawling. Damaged anemones may be subject to predation by fish or other animals. Therefore, resistance is assessed as intermediate. *Cerianthus lloydii* has a life-span of 11-20 years. The age at sexual maturity and fecundity is unknown. Fertilisation is external and the larvae are pelagic (MES 2008). The dispersal potential may therefore be high, although without information on the fecundity and due to the long life-span and slow growth of this anemone recoverability is assessed as moderate.

Despite their robust body form, bivalves are vulnerable to physical abrasion. *Tellina fabula* is a shallow burrower with a fragile shell and may be damaged by an impact with fishing gear so resistance is recorded as intermediate. As presumably not the whole population is affected, recoverability is assessed as high.

Spio spp. is a small polychaete with 2-5cm in body length and lives in burrows in sand where it feeds as a surface deposit-feeder on detritus and diatoms. Adult worms can burrow up to 10 cm down and may escape the disturbance. Juveniles can only burrow up to 2 cm into the sediment and are likely to be affected. A resistance of intermediate has therefore been recorded. It is reported that the total density of spionids actually increased with increased fishing disturbance, presumably due to their ability to colonize newly exposed substratum. Recoverability has been recorded as very high.

3.9.5.5 Changes in siltation

Table 3-32: Sensitivity of sandbanks (Doggerbank) towards the pressure 'changes in siltation'.

Changes in siltation		Resistance	Recoverability	Sensitivity
Physical habitat		high	very high	very low
Characteristic species	*Spiophanes bombyx*	high	very high	very low
	Lanice conchilega	intermediate	very high	low
	Bathyporeia spp.	high	very high	very low
	Amphiura brachiata	high	very high	very low
	Cerianthus lloydii	intermediate	high	low
	Tellina fabula	high	very high	very low
	Spio decoratus	high	very high	very low
Habitat sensitivity				**low**

Physical habitat – explanatory notes

Sediment plumes generated by construction works or aggregate extraction may cause changes in habitat structure such as infilling of small pits by fine sediments, siltation within crevices or development of migratory sand ripples (Hill et al. 2011). Finer sediment particles remain in suspension longer than larger particulates and can disperse over a wider area. Suspended fine and medium sands require a few hours for resettlement whereas silty sediments may remain in suspension for a few days (OSPAR 2008). In habitats with strong seabed transport recovery may be fast as fine sediments are rapidly mobilized. Resistance of sandbank habitats is therefore regarded as high and recovery time as very high.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Spiophanes bombyx lives in the sediment and uses sediment grains to make its tube. It is likely that *Spiophanes bombyx* will be able to move up through any extra sediment, therefore resistance has been recorded as high. Recovery is likely to be immediate.

Lanice conchilega forms a characteristic tube of sand-grains ending at the head end in a tuft of sandy filaments that project from the surface of the sediment. The worm feeds on particulate matter on the sediment surface captured by a crown of tentacles. *Lanice conchilega* is capable of movement only within the tube and is likely to be vulnerable to deposition of sediment (MES 2008). Resistance has been assessed as intermediate and recoverability as very high.

The amphipod *Bathyporeia* spp. would probably be unaffected by an additional covering of sediment of a texture within its habitat preference, although there may be an energetic cost incurred by the additional burrowing activity required to attain a near-surface position for feeding and to swim. *Bathyporeia* spp. is likely to be more intolerant of smothering by both coarser and finer particles through which burrowing is likely to be hindered. Consequently, the resistance of *Bathyporeia* spp*.* to an increase in sedimentation has been assessed to be high. The genus is likely to have a very high capacity for recovery.

Amphiura spp.is an infaunal genus which can burrow and lives up to a depth of 4 cm within the sediment. Therefore, smothering by sediment of 5 cm is unlikely to have a great effect although feeding and hence viability of the population may be reduced if the sediment is particularly fine and mobile. Since only sub-lethal effects are likely resistance is considered to be high. Recovery is likely to be rapid as individuals move up through the sediment to resume their position for feeding and any fine particles are removed.

Cerianthus lloydii occurs in muddy sediments, so is likely to be tolerant of some smothering by suspended sediment. With a maximum height of only 3 cm above the sediment, the species will be completely smothered by the benchmark level of 5 cm of sediment. *Cerianthus lloydii* may be able to move by a limited amount and to rise above the smothering material. However, it is also likely that some individuals may die and so resistance is reported to be intermediate. Recoverability is assumed to be high.

Tellina fabula is a shallow burrower in sandy sediments. It requires its inhalant siphon to be above the sediment surface for feeding and respiration. Smothering with 5 cm of sediment would temporarily halt feeding and respiration and requires the species to relocate to its preferred depth. *Tellina fabula* is an active burrower and would be expected to relocate with no mortality. However, growth and reproduction may be compromised and so resistance is assessed as high. Growth and reproduction would return to normal following relocation so recoverability is immediate.

Spio spp*.* lives in the sediment and uses sediment grains to make its tube. It is likely that *Spio* spp*.* will be able to move up through any extra sediment, therefore resistance has been recorded as high. Recoverability will probably be very high.

3.9.6 Reefs

3.9.6.1 Definition of reefs

Reefs can be either biogenic concretions or of geogenic origin. They are hard compact substrata on solid and soft bottoms, which arise from the sea floor in the sublittoral and littoral zone. Reefs may support a zonation of benthic communities of algae and animal species as well as concretions and corallogenic concretions (EC 2013).

Sites of outstanding ecological value in the North Sea include areas around the Borkum Reef Ground, the eastern flank of the Elbe glacial valley, and the Steingrund reef off Helgoland (Nehls et al. 2008). Biogenic reefs have not yet been designated in the German North Sea.

3.9.6.2 Characteristic species

According to Nehls et al. (2008) criteria for characteristic reef species are:

- presence >50 % at the stations of a subarea
- preference for hard substrate
- longevity

Characteristic species identified by Nehls et al. (2008) for the Borkum Reef Ground and the Sylter Outer Reef are:

Table 3-33: Characteristic species of reef habitats in the German North Sea (Nehls et al. 2008). Species selected for the sensitivity assessment are printed in bold.

Porifera	***Leucosolenia botryoides***
Cnidaria	*Metridium senile*
Cnidaria	***Alcyonium digitatum***
Cnidaria	*Alcyonium glomeratum*
Cnidaria	*Sertularia cupressina*
Polychaeta	***Pomatoceros triquiter***
Bryozoa	***Flustra foliacea***
Crustacea - Cirripedia	*Balanus balanus*
Crustacea - Cirripedia	***Balanus crenatus***
Crustacea - Cirripedia	*Balanus improvisus*
Bivalvia	***Pholas dactylus***
Crustacea – Amphipoda	*Caprella linearis*
Crustacea - Decapoda	*Galathea strigosa*
Crustacea - Decapoda	*Galathea squamosa*
Crustacea - Decapoda	***Cancer pagurus***
Echinodermata	***Echinus esculentus***
Ascidiacea	***Ciona intestinalis***
Ascidiacea	Ascidiella scabra

For the sensitivity assessment one species was chosen from each class, except for Crustacea, where a barnacle species and a decapod crustacean were selected.

3.9.6.3 Selective extraction

Some of the reef structures in the Sylter Outer Ground are situated within designated areas for aggregate extraction. Especially areas with gravel or smaller stones may be affected by the pressure selective extraction.

Table 3-34: Sensitivity of reefs towards the pressure 'selective extraction'.

Selective extraction		Resistance	Recoverability	Sensitivity
Physical habitat		low	very low	very high
Characteristic species	*Leucosolenia botryoides*	low	low	high
	Alcyonidium digitatum	low	moderate	intermediate
	Pomatoceros triqueter	low	high	intermediate
	Flustra foliacea	low	moderate	intermediate
	Balanus crenatus	low	very high	low
	Pholas dactylus	low	moderate	intermediate
	Cancer pagurus	intermediate	moderate	intermediate
	Echinus esculentus	low	moderate	intermediate
	Ciona intestinalis	low	moderate	intermediate
Habitat sensitivity				**very high**

Physical habitat – explanatory notes

The extraction of sediment implies the complete removal of substrate and attached organisms. Severe alterations of seabed topography occur, therefore resistance to selective extraction is rated as low. Regeneration of gravel and rock substrata by hydrodynamic or other processes is not possible (Herrmann & Krause 1998), thus the recovery of the reef habitat with the associated benthic fauna will not take place.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

The majority of characteristic species in reef habitats is sessile and attached to the substrate and would therefore be removed by aggregate dredging. Only some epifaunal and swimming species may be able to avoid the impact. Resident populations of the benthic endo- and sessile epifauna would be lost, so resistance for these species is assessed as low.

Specific information on the reproduction or dispersal abilities of the sponge *Leucosolenia botryoides* is not available. Most sponges however, tend to be slow growing and long lived, so precautionarily recoverability is assessed as low.

It is likely that the octocoral *Alcyonium digitatum* has a high recovery potential. Its reproductive strategy is to 'broadcast' gametes into the water for fertilization indicates that fecundity is high. The combination of spawning in winter and that the larvae may have a long pelagic life allows a considerable length of time for the planulae to disperse (recruits from other populations can replace impacted populations), settle and metamorphose ahead of the spring plankton bloom. Young *Alcyonium digitatum* will consequently be able to take advantage of an abundant food resource in spring and be well developed before the appearance of other forms which may compete for the same substrata. In addition, because the planulae do not feed whilst in the pelagic zone they do not suffer by being released at the time of minimum plankton density. They may also benefit by the scarcity of predatory zooplankton which would otherwise prey upon them.

However, the life span of *Alcyonium digitatum* certainly exceeds 20 years as colonies have been followed for 28 years in marked plots and sexual maturity is reached at 2-3 years. The species has a relatively slow growth rate and therefore recovery to adult biomass is likely to take many years. Recoverability is assessed as moderate.

The encrusting polychaete *Pomatoceros triqueter* is fairly widespread, reaches sexual maturity within 4 months and longevity has been recorded to be between 1.5 and 4 years. Larvae are pelagic for about 2-3 weeks in the summer and about 2 months in the winter, enabling them to disperse widely. Recovery is therefore likely to be high.

Recovery of the bryozoan *Flustra foliacea* will depend on recruitment from other populations and is assessed as high. The brooded, lecithotrophic larvae of bryozoans have a short pelagic life time of several hours to about 12 hours. Recruitment is dependent on the supply of suitable, stable, hard substrata. *Flustra foliacea* colonies are perennial, and potentially highly fecund when large. In the strong currents occupied by *Flustra foliacea* populations many larvae are probably swept away, either to colonize other substrata or lost. Recruitment may be enhanced in areas subject to sediment abrasion, where less tolerant species are removed, making more substratum available for colonization, especially if larvae release in spring coincides with the end of winter storms. Once settled, new colonies take at least 1 year to develop erect growth and 1-2 years to reach maturity, depending on environmental conditions. Where the population was removed, recruitment would depend on the proximity of other populations or individuals and the hydrographic regime, and is likely to be more protracted, taking up to 5 years. In areas isolated by either by distance or hydrographic regime, *Flustra foliacea* may take longer to recolonize. Recoverability is recorded to be moderate.

The barnacle *Balanus crenatus* is an important early colonizer of sublittoral rock surfaces and it heavily colonized a site that was dredged for gravel within 7 months. Therefore, recovery is predicted to be very high.

Provided a similar substratum remains and there is larval availability, recolonization of the boring bivalve *Pholas dactylus* is likely to occur and so recovery within five years should be possible, though maybe not to previous abundance. Recoverability is estimated to be moderate.

Substrate removal is likely to remove a proportion of *Cancer pagurus* although some will escape. Those that escape undamaged will quickly recolonize whatever seabed remains and migrate to new habitats if necessary. Female *Cancer pagurus* have high fecundity of 0.25-3 million eggs per spawning but mortality of larvae is high. Since juveniles spend the first 3 years post-settlement in the intertidal, recovery of an adult population from a mortality event is likely to take several years. If *Cancer pagurus* were to be completely eradicated from an area, repopulation would occur by larval input from surrounding areas and adult migration. Therefore, a resistance of intermediate and a recoverability of moderate have been recorded.

Sea urchins like *Echinus esculentus* are slow moving and unlikely to escape removal of their substratum. Sea urchin recruitment is sporadic and dependent on location but populations would probably recover within 5 years, except in locations isolated by geography or hydrography. *Echinus esculentus* has a high larval dispersal potential but is slow to mature and it would take up to 8 years for adult biomass to be restored.

Adult individuals of the sea squirt *Ciona intestinalis* are sessile and so cannot contribute to recovery through active immigration. Rafting by adults attached to floating objects or shipping may form an important mechanism for recolonization. Dispersal through attachment to ships is believed to be the main reason behind the widespread global distribution. Otherwise, dispersal is mediated by the larval stage. Larval recruitment from other populations may be restricted by the larvae being retained near the adults in mucus threads. Settling time of the larva is quite short - usually a few hours so dispersal may be limited. No information is available regarding the fecundity of this species. Reproductive frequency and longevity varies from semelparous and annual to iteroparous and living 2-3 years depending on depth and salinity (in Sweden at least). Reproduction (in Plymouth) is recorded as occurring all year round. Recoverability is assessed to be moderate.

3.9.6.4 Abrasion

Table 3-35: Sensitivity of reefs towards the pressure 'abrasion'.

Abrasion		Resistance	Recoverability	Sensitivity
Physical habitat		intermediate	very low	high
Characteristic species	*Leucosolenia botryoides*	intermediate	moderate	intermediate
	Alcyonium digitatum	intermediate	moderate	intermediate
	Pomatoceros triqueter	intermediate	very high	low
	Flustra foliacea	intermediate	moderate	intermediate
	Balanus crenatus	intermediate	very high	low
	Pholas dactylus	intermediate	moderate	intermediate
	Cancer pagurus	intermediate	moderate	intermediate
	Echinus esculentus	low	moderate	intermediate
	Ciona intestinalis	low	moderate	intermediate
Habitat sensitivity				**high**

Physical habitat – explanatory notes

Geogenic reef habitats in the German EEZ of the North Sea are fished with heavily-rigged beam trawls, which often damages or even destroys habitat structures (BfN 2012). Even though hard substrates are relatively resistant to physical damage from towed gears, fishing with mobile gears may result in modification of the substratum, including removal of shell debris, cobbles and rocks and the movement of boulders. Recovery of the benthic reef species will depend on the life-history characteristics of the species affected, including the ability of damaged adults to repair or regenerate lost or damaged parts and the ability of larvae to reach and recolonize the habitat. Re-establishment of long-lived, slow-growing species in which maturity occurs late will be slower than for smaller species with faster life cycles (MarLIN 2013). However, a pre-condition for the recovery of the benthic community is the presence of hard substrate for settlement, which may be partly removed. Resistance is assessed as intermediate and recovery of hard substrata is predicted to be very low.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Specific information on the biological traits of the sponge *Leucosolenia botryoides* is not available. *Leucosolenia botryoides* is a very delicate, soft, white, tubular sponge that grows to up 2 cm wide and 1 cm thick. Abrasion may physically damage or dislodge the sponge, therefore resistance is judged as intermediate. Regarding reproduction and dispersal abilities of this species no information is available. Sponges may also regrow from tissue remaining in crevices or other irregularities and that were not affected by the abrasion. Precautionarily recoverability is estimated as moderate.

The octocoral *Alcyonium digitatum* is prone to damage and abrasion by fishing gears e.g. rock hopper otter trawls and dredges that are designed to penetrate the sea bed. In addition, the anchoring of boats for purposes of recreational diving may cause cumulative damage in heavily visited sites. Veale et al., 2000 reported that the abundance, biomass and production of epifaunal assemblages, including *Alcyonium digitatum*, decreased with increasing fishing effort. A resistance rank of intermediate is recorded as it is likely that the proportion of the population on vertical slopes and under overhangs will be unaffected by mechanical abrasion. The populations inhabiting horizontal surfaces at greater depths are at risk from abrasion. However, the fact that *Alcyonium digitatum* is more abundant on high fishing effort grounds suggests that this seemingly fragile species is more resistant to abrasive disturbance than might be assumed, presumably owing to the ability for the replacement of senescent cells and regeneration of damaged tissue in addition to the early larval colonization of available substrata. Due to the relatively slow growth of the species, recoverability is estimated as moderate.

Pomatoceros triqueter has a hard-calcareous tube that is resistant to sand and gravel abrasion. Hiscock (1983) noted that a community, under conditions of scour and abrasion from stones and boulders moved by storms, developed into a community consisting of fast-growing species such as *Pomatoceros triqueter*. Off Chesil Bank, the epifaunal community dominated by *Pomatoceros triqueter*, *Balanus crenatus* and *Electra pilosa*, decreased in cover in October, was scoured away in winter storms, and was recolonized in May to June. Warner (1985) reported that the community did not contain any persistent individuals, being dominated by rapidly colonizing organisms. But, while larval recruitment was patchy and varied between the years studied, recruitment was sufficiently predictable to result in a dynamic stability and a similar community was present in 1979, 1980, and 1983. Scour due to winter storms is probably greater than the benchmark level. Scour and abrasion will probably remove a proportion of the population, suggesting a resistance of intermediate. However, it demonstrates rapid growth and recruitment so that it recoverability is assumed to be very high. The abundance of *Pomatoceros triqueter* may increase due to decreased competition from other species.

Flustra foliacea is tolerant of sediment abrasion but physical disturbance by fishing gear has been shown to adversely affect emergent epifaunal communities. Although *Flustra foliacea* is flexible, physical disturbance by a passing scallop dredge is likely to damage fronds and remove some colonies, suggesting a resistance of intermediate. Colonies on hard substrata are probably less vulnerable to fishing activity but would probably be damaged or partially removed. Colonies growing on rocks, cobbles and shells on coarse grounds, may be removed by a scallop dredge and therefore be highly intolerant. Overall, local recruitment is probably good and a damaged or reduced population may recover its numbers and percentage cover in less than 5 years. Recoverability is therefore assessed as moderate.

Balanus crenatus would probably be crushed by a heavy force, such as an anchor landing on it. However, it is small and individuals in fissures and crevices would probably survive. Resistance is assessed as intermediate. The species has a high dispersal and colonization potential as well as fast growth rates. Recovery is predicted to be very high.

The shell of *Pholas dactylus* is thin and brittle so a force, equivalent to a 5-10 kg anchor and its chain being dropped or a passing scallop dredge, is likely to result in death. However, because the common piddock lives within a burrow in soft rock, generally only those individuals close to the surface will be damaged by an abrasive force or physical disturbance. Therefore, a resistance of intermediate has been recorded to represent the possible loss of a proportion of the population. Recolonization of the affected area by pelagic larvae is likely to occur and with several months spawning every year recovery within five years is expected.

Berried *Cancer pagurus* are likely to be disturbed by dredging and trawls as they are relatively immotile and spend most of their time half buried in the sediment. Abrasion is also likely to make *Cancer pagurus* vulnerable to Burn Spot Disease which may cause some mortality. *Cancer pagurus* is often damaged or killed if struck by a dredge and annual mortality can be as much as 14% of the population. *Cancer pagurus* is a rather brittle animal, easily damaged or killed by heavy impacts, and a resistance of intermediate has been recorded because, although a high proportion of individuals die as a result of abrasion, the whole population is unlikely to be affected. Recoverability is assessed to be moderate.

Species with fragile tests such as *Echinus esculentus* were reported to suffer badly as a result of impact with passing scallop or queen scallop dredges. Adults can repair non-lethal damage to the test and spines can be re-grown but most dredge impact is likely to be lethal. Schroeder et al. (2008) reported on fishery-induced mortality of *Echinus esculentus* reaching up to 50 %. Resistance has therefore been assessed as low. Sea urchin recruitment is sporadic and dependent on location but populations would probably recover within 5 years, except in locations isolated by geography or hydrography. *Echinus esculentus* has a high larval dispersal potential but is slow to mature and it would take several years for adult biomass to be restored.

Ciona intestinalis is a large ascidian, with a soft, retractile body. Physical disturbance by a passing dredge is likely to cause physical damage and death. Therefore, a resistance of low has been recorded. Recoverability is assessed to be moderate.

3.9.6.5 Changes in siltation

Table 3-36: Sensitivity of reefs towards the pressure 'changes in siltation'.

Changes in siltation		Resistance	Recoverability	Sensitivity
Physical habitat		intermediate	high	low
Characteristic species	*Leucosolenia botryoides*	intermediate	moderate	intermediate
	Alcyonium digitatum	intermediate	high	low
	Pomatoceros triqueter	low	high	intermediate
	Flustra foliacea	tolerant	not relevant	not sensitive
	Balanus crenatus	low	very high	low
	Pholas dactylus	high	very high	very low
	Cancer pagurus	high	very high	very low
	Echinus esculentus	intermediate	high	low
	Ciona intestinalis	intermediate	high	low
Habitat sensitivity				**intermediate**

Physical habitat – explanatory notes

Smothering of sediment will significantly change the habitat structure. Animals may be affected by the prevention of feeding, reduction in growth and reproduction, interference with respiration and potentially localized anoxia and interference with larval settlement. Tall erect species may survive due to their size, while some hydroids may survive as dormant stages. But encrusting sponge species and ascidians are likely to be damaged or killed by smothering, while vertical surfaces and overhangs will provide refuges from the effects of the factor (MarLIN 2013). Resistance is estimated as intermediate. Recoverability strongly depends on the prevailing hydrodynamic regime. In high energy environments deposits will be rapidly removed, while in environments with low and moderate current energy, as prevails in large parts of the German North Sea, recovery may take more than one year (Hill et al. 2011). Recoverability is therefore predicted as high.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Leucosolenia botryoides is a very delicate, soft, white, tubular sponge that grows to up 2 cm wide and 1 cm thick. Accumulation of a few centimetres of sediment smothers the sponge. Increases in deposition of suspended sediment may interfere with feeding, clogging pores and channels etc. Many sponges have cleaning mechanisms for dealing with siltation such as sloughing of outer cells or mucus production. However, there may be significant inhibition of feeding and respiration and small colonies may suffer mortality if de-oxygenation below the silt occurs. Resistance is assessed as intermediate and recoverability due to lack of information on reproductive potential as moderate as a precautionary approach.

Alcyonium digitatum is permanently attached to the surface of rocky substrata. Thus, it would be unable to avoid the deposition of a smothering layer of material up to a depth of 5 cm. Some colonies can attain a height of up to 20 cm so would still be able to expand tentacles and columns of the polyps to filter feed, and materials may be sloughed off with a large amount of mucous. Smaller / younger colonies that initially form encrustations between 5 and 10 mm thick are likely to be killed by smothering as respiration is likely to be hindered and a resistance of intermediate is recorded. Recoverability is assessed to be high.

Smothering with a 5 cm layer of sediment would completely cover the tubes of *Pomatoceros triqueter* that usually lie flat against the surface of the rock. It is also likely that too much sediment on the surface of rocks or shells would prevent settlement of larvae and impair the long-term survival of populations. Resistance has been assessed to be low. Recoverability is likely to be high.

Flustra foliacea dominated communities were reported to form in, and hence tolerate, areas subject to sediment transport (mainly sand) and periodic, temporary, submergence by thin layers of sand (ca <5 cm). In some cases, *Flustra foliacea* was seen to be partially buried by sand. It is likely that *Flustra foliacea* would withstand smothering by 5 cm of sediment for a month. Large colonies are likely to be >6 cm in height and exposed autozooids will be able to feed, providing food for the rest of the colony. Therefore, not sensitive has been recorded.

Balanus crenatus can withstand covering by silt provided that the cirri can extend above the silt layer but smothering by 5 cm of sediment would prevent feeding and could cause death. Resistance is therefore judged to be low. The species has a high dispersal and colonization potential as well as fast growth rates. Recovery is predicted to be very high.

Resistance to smothering is expected to be high because feeding apparatus can be cleared of particles although this will be energetically costly. Experimental work with *Pholas dactylus* showed that large particles can either be rejected immediately in the pseudofaeces or passed very quickly through the gut. In Exmouth, Knight (1984) found *Pholas dactylus* covered in a layer of sand and in Eastbourne individuals live under a layer of sand with siphons protruding at the surface. Recoverability is estimated to be very high.

The crab *Cancer pagurus* is able to escape from under silt and migrate away from an area. Smothering is unlikely to cause mortality therefore a resistance of high has been recorded. Recovery is predicted to be very high.

The adults of the sea urchin *Echinus esculentus* are slow moving and unlikely to be able to avoid smothering. A 5 cm layer of sediment is likely to affect smaller specimens more than large specimens. Smothered individuals are unlikely to be able to move through sediment. However, individuals are unlikely to starve within a month. A layer of sediment may interfere with larval settlement. Resistance is assessed to be intermediate and recoverability as high.

The ascidian *Ciona intestinalis* is permanently attached to the substratum and is an active suspension feeder. Because the adults reach up to 15 cm in length and frequently inhabit vertical surfaces, smothering with 5 cm of sediment will probably only affect a proportion of the population. Resistance is judged as intermediate and recoverability as high.

3.9.7 Species-rich habitats on coarse sands, gravel or shell gravel

3.9.7.1 Definition

Coarse sediments in the south-eastern North Sea are settled by the *Goniadella-Spisula*-association. Rachor & Nehmer (2003) differentiate two variations of this association in the EEZ. Characteristic species for both are *Ophelia limacina*, *Aonides paucibranchiata* and *Thracia* spp. The species-rich association can be found on coarse sands and gravel, e.g. in the Borkum Reef Ground, the Amrum Outer Ground and the Sylter Outer Reef. Rachor & Nehmer (2003) identified only one characteristic species for this habitat, the lancelet *Branchiostoma lanceolatum*.

This habitat type comprises mixed or unmixed sediments of coarse sands, gravel and shell debris, which are settled by a specific, species-rich endofauna and benthic community. Characteristic species according to the mapping guidelines of the BfN (2011) are: *Aonides paucibranchiata*, *Branchiostoma lanceolatum*, *Polygordius* spp., *Protodorvillea kefersteini*, *Echinocyamus pusillus*, *Spisula elliptica* and *Pisione remota*. These species should also be used in the sensitivity assessment, however, little information on biological traits is currently available especially for the small polychaetes.

3.9.7.2 Selective extraction

Table 3-37: Sensitivity of species-rich habitats on coarse sands, gravel or shell gravel towards the pressure 'selective extraction'.

Selective extraction		Resistance	Recoverability	Sensitivity
Physical habitat		low	low	high
Characteristic species	*Aonides paucibranchiata*	low	moderate	intermediate
	Branchiostoma lanceolatum	low	moderate	intermediate
	Pisione remota	low	moderate	intermediate
	Echinocyamus pusillus	low	moderate	intermediate
	Spisula elliptica	low	high	intermediate
Habitat sensitivity				**high**

Physical habitat – explanatory notes

The extraction of sediment implies the complete removal of substrate by creating longitudinal tracks of generally 2-3 m width and up to 50 cm depth (trailer suction dredging) or rounded pits of around 10 m depth and with a diameter of 10-50 m (anchor dredging). Severe alterations of seabed topography and possibly also changes in sediment composition occur, therefore resistance to selective extraction is rated as low.

Physical seabed structures are supposed to have recovered when dredge tracks have disappeared and the original sediment composition is restored. Research on seabed recovery mostly focuses on observation of dredge furrows, while the recovery of sediment composition may take far longer but is less intense investigated. Recovery takes the longest period of time at dredge sites characterised by coarse sediments (Hill et al. 2011). Observations from studies conducted in sandy gravel sediments reveal that the morphological behaviour of dredged tracks and pits varies significantly. In an area exposed to long-period waves, dredge tracks 0.3–0.5 m deep, in a gravelly substrate at a depth of 38 m, were found to disappear completely within eight months. In contrast, at an experimental dredged gravel site off Norfolk, UK, in 25 m of water, dredge tracks appeared to have been completely eroded well within three years of the cessation of dredging. Erosion of dredge tracks in areas of moderate wave exposure and tidal currents have been observed to take from three to more than seven years in gravelly sediments. In the latter case, however, infill resulted mainly from sand in transport. Especially in coarse sediments, the refill material may be finer grained than the material on the surrounding seabed, which could lead to a permanent change in benthic communities (Herrmann & Krause 1998). In the southern North Sea where tidal currents are generally strong, sand with a grain size up to 2 mm is mobile across the area during spring tides (Hill et al. 2011). However, the regeneration of gravel may not be possible, as there are no hydrodynamic mechanisms known to restore gravel or stony habitats (Herrmann & Krause 1998). As there is the risk of at least part of the habitat being lost, recoverability is recorded as low.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

The majority of species in coarse sands, gravel or shell debris habitats is infaunal and would therefore be removed along with the substratum. Only some epifaunal and swimming species may be able to avoid the impact. Resident populations of the benthic endofauna would be lost, so resistance for all characteristic species is assessed as low.

Aonides paucibranchiata is a small-sized polychaete with limited mobility. The fecundity and dispersal potential of this genus is low (larval duration 2-10 days), so recolonisation from sources outside a disturbed area is likely to be slow. Recoverability is estimated to be moderate (MES 2008).

For lancelets in general, it is supposed that they are iteroparous (reproducing more than once in a lifetime), spawning repeatedly in their several-year lifetime, but only once per breeding season. Fuentes et al. (2007) studied the spawning behavior of the European Lancelet *Branchiostoma lanceolatum* along the Mediterranean coast of southern France. They found that spawning occurs from around mid-May to early July, but varies from year to year (EOL 2013). No information is available on potential of larval dispersal; therefore, recoverability has been assessed as moderate.

Pisione remota lives for 3-5 years and is likely to reach maturity after one year. Reproduction is from August-September and fertilisation is internal after which planktonic larvae are released into the water column. There is very little information on the length of the larval phase. It is probable that this genus has a moderate recoverability based on the presence of a pelagic dispersal phase, but more information is required on fecundity and larval biology to have confidence in this assessment (MES 2008).

Echinocyamus pusillus is small and only lives for 1-3 years, reaching sexual maturity after one year. There is little information available on its fecundity. Reproduction is external and the planktotrophic larvae occur in the plankton from March to September indicating a high dispersal potential. Once the sediment has become colonised, the abundance and biomass of *Echinocyamus pusillus* could be expected to recover within 3 years.

Little information is available on biological traits of *Spisula elliptica*, therefore sensitivity of the closely related species *Spisula solida* is used as reference. The bivalve *Spisula solida* can live up to ten years. Individuals are sexually mature at 1 year, regardless of their size. The sexes of *Spisula* are separate and both show a synchrony in gametogenic development and spawning. Gametogenesis starts in September when temperatures decrease and spawning begins in February. Larvae can remain in the water column for several weeks, allowing fairly wide dispersal. The potential recovery of this bivalve is high and is often recorded amongst the first colonizers of sediments disturbed by dredging.

3.9.7.3 Abrasion

Table 3-38: Sensitivity of species-rich habitats on coarse sands, gravel or shell gravel towards the pressure 'abrasion'.

Abrasion		Resistance	Recoverability	Sensitivity
Physical habitat		intermediate	moderate	intermediate
Characteristic species	*Aonides paucibranchiata*	intermediate	high	low
	Branchiostoma lanceolatum	intermediate	high	low
	Pisione remota	intermediate	high	low
	Echinocyamus pusillus	intermediate	high	low
	Spisula elliptica	intermediate	high	low
Habitat sensitivity				**intermediate**

Physical habitat – explanatory notes

Impacts of fishing gears on habitats with coarse sands include the smoothing of the seafloor by flattening of biogenic structures or sand ripples, the penetration of sediment, smothering by resuspended sediment and displaced or overturned gravel (Environment Agency 2010). Otter trawls generally disturb the upper 1-5 cm while beam trawls scour the sediment down to 8 cm (FAO 2004). Resistance towards abrasion is assessed as intermediate.

Recovery time in gravel habitats has been predicted to be in the order of ten years, while physical restoration of sandy habitats has been observed to be rapid (days to few months) (Environment Agency 2010). The visible dredge marks from towed gear have been shown to be relatively short lived, lasting no more than a year in coarse sediments. Monitoring of a 'closed area' of gravel habitat on Georges Bank, showed that five years after closure of the area to high levels of scallop fishing, the biomass and abundances of certain taxa (including crabs, molluscs, polychaetes and echinoderms) were still increasing. As such, the authors predicted that the recovery time for gravel habitats was in the order of ten years. Similar recovery rates were observed during 10 years of monitoring of a gravelly habitat off the Isle of Man following closure to scallop dredging. The authors speculate that the slow rate of recolonization of gravel habitat by structure-forming epifauna (sponges, bryozoans, anemones, hydroids, colonial tube worms) following fishing disturbance may be due to factors such as the low survival of recruits of these species, due to intermittent burial of the gravel by migrating sands, and the presence of high numbers of scavengers (crabs, echinoderms, nudibranchs, gastropods), the abundance of which increased rapidly on the gravel post disturbance. Hence, this suggests that the recovery of these habitats may be slower than individual life history traits predict. Recoverability is assessed as moderate.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Little is known about the life history of the polychaete worm *Aonides paucibranchiata* but its size and morphology suggest that it is likely to be vulnerable to physical disturbance. Infaunal polychaetes with little mobility are likely to be damaged by abrasion and suffer some degree of mortality. Resistance is judged as intermediate. As a short-lived animal with small body size, it is likely to recover adult biomass relatively quickly following colonisation by juveniles (MES 2008). Providing that part of the population survives, *Aonides paucibranchiata* is likely to have a high recoverability.

Although the lancelet *Branchiostoma lanceolatum* is able to swim, most of the time is spent partially buried in the sand filtering microscopic food particles from the water. Disturbance and penetration of the sediment is likely to damage or kill some individuals of the population. Resistance is therefore judged to be intermediate. Recoverability is assessed as high.

Pisione remota is a small free-living polychaete with a body length of 1.5 cm and lives burrowed in coarse sand where it is a carnivore feeding on small invertebrates. It has some mobility but may be vulnerable to abrasion and physical disturbance. Resistance is estimated to be intermediate and recoverability as high.

The sea urchin *Echinocyamus pusillus* has a fragile shell which may be damaged by abrasion. Resistance is assessed as intermediate and recoverability as high.

Little information is available on biological traits of *Spisula elliptica*, therefore sensitivity of the closely related species *Spisula solida* is used as reference. Fishing for demersal species will disturb the surface layer of sediment and any protruding or shallow burrowing species. Experimental trawls showed that 93% of the uncaught *Spisula solida* were undamaged, as they were well protected by their thick shells, and only 1% died. The impacts caused by a fishing dredge significantly increased the number of exposed *Spisula solida* clams and the abundance of potential predators. The impact of the dredge increased the time needed for *Spisula solida* to rebury, which rendered them vulnerable to predation for longer periods. Resistance has been assessed as intermediate as mortality may occur and recoverability has been assessed as high.

3.9.7.4 Changes in siltation

Table 3-39: Sensitivity of species-rich habitats on coarse sands, gravel or shell gravel towards the pressure 'changes in siltation'.

Changes in siltation		Resistance	Recoverability	Sensitivity
Physical habitat		intermediate	very high	low
Characteristic species	*Aonides paucibranchiata*	high	very high	very low
	Branchiostoma lanceolatum	high	very high	very low
	Pisione remota	high	very high	very low
	Echinocyamus pusillus	high	very high	very low
	Spisula elliptica	intermediate	high	low
Habitat sensitivity				**low**

Physical habitat – explanatory notes

Sediment plumes generated by construction works or aggregate extraction may cause changes in habitat structure such as infilling of small pits by fine sediments or siltation within crevices (Hill et al. 2011). Finer sediment particles remain in suspension longer than larger particulates and can disperse over a wider area. As suspended particles tend to be significantly finer than the prevailing coarse sands and gravels, changes in sediment composition are supposed to be more distinct than e.g. in mud habitats. Resistance of coarse sands, gravel and shell debris habitats is therefore regarded as intermediate. Recovery is dependent on seabed transport, wave and tidal energy. It is estimated to be very high in coarse sediments.

Characteristic species – explanatory notes

(Information on species characteristics is taken from the MarLIN web site unless otherwise stated)

Aonides paucibranchiata is a small deposit feeding polychaete with limited mobility. The species lives in a loosely constructed tube or is free-living (MES 2008). An additional 5 cm layer of sediment would result in a temporary cessation of feeding activity, and therefore growth and reproduction are likely to be compromised. However, *Aonides paucibranchiata* would be expected to quickly relocate to its favoured depth, with no mortality, and hence a high resistance is recorded. Recoverability will probably be very high.

Information on the impact of smothering to the lancelet Branchiostoma lanceolatum is not available. However, as a species burrowing in sediment, it is likely to be able to accommodate deposition of sediment. The population may still suffer from reduced viability, so tolerance is assessed as high. Recoverability is assumed to be rapid.

The burrowing polychaete *Pisione remota* may be able to accommodate deposition of small quantities of sediment, probably with some additional energetic costs (MES 2008). Resistance is estimated as high and recoverability as very high.

The sensitivity of *Echinocyamus pusillus* to sedimentation is difficult to assess due the paucity of information but as a burrowing species it is likely to be able to resurface through thin veneers of sediment (MES 2008). Resistance is assessed as high and recoverability as very high.

Little information is available on biological traits of *Spisula elliptica*, therefore sensitivity of the closely related species *Spisula solida* is used as reference. *Spisula solida* is a fast burrowing bivalve and suspension feeder. If *Spisula solida* were covered by sediments it would be able to reposition itself within the sediment. Fahy et al. (2003) noted that in a clam bed in Ireland, where part of the bed has silted up, numbers of *Spisula solida* and the size of the clam patch were reduced. Therefore, resistance has been assessed as intermediate to reflect the reduction in the size of the clam bed and *Spisula* numbers. Recoverability is assessed as high.

4 Work package 3: Hydrographical conditions (Descriptor 7)

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4.1 Objective

Descriptor 7 of the MSFD requires assessing the extent and impact of hydrographical conditions permanently altered by human activities. According to the Directive, the Good Environmental Status of Descriptor 7 is achieved when 'permanent alteration of hydrographical conditions does not adversely affect marine ecosystems' (EC 2008). Hydrographical conditions refer to the physical properties of seawater such as temperature, salinity, current and wave regime, upwelling patterns and bathymetry. These parameters play a crucial role in marine ecosystems as they structure the water masses and determine the characteristics of seabed and water column habitats. Hydrographical conditions influence for instance the dispersal of larvae, the production and growth of benthic and pelagic fauna and the exchange between different layers of water (EC 2012). GES is achieved when the hydrographical conditions of habitats (water column and sea floor) are not affected to the extent that their key functions (e.g. provision of spawning, breeding and feeding areas or migration routes) are degraded.

Within the framework of the research and development project 'Compilation and assessment of selected anthropogenic pressures in the context of the Marine Strategy Framework Directive' BioConsult Schuchardt & Scholle GbR was commissioned with the development of a concept to assess the criteria and indicators of Descriptor 7. This report identifies human activities and associated pressures on hydrographical conditions in German waters and describes impacts on ecosystem components as far as currently foreseeable. The understanding of Descriptor 7 by Regional Seas Conventions and neighbouring countries is discussed and a proposal for the national strategy is made. A first basic draft of an assessment concept is briefly outlined and considerations for setting baselines and targets for the Good Environmental Status are presented.

4.2 Rationale

The pressure exerted on hydrographical conditions is 'interference with hydrological processes' according to Annex III, table 2 of the MSFD. Hydrological processes refer to the movement, distribution and quality of water. Interference with hydrological processes can cause permanent alterations of hydrographical conditions by changes in the thermal or salinity regime, changes in the tidal regime, sediment and freshwater transport, current or wave action (WG GES 2011). Modifications of the physical and chemical characteristics may lead to negative impacts on the structure and function of marine habitats and biological communities. Hydrographical conditions are often highly variable and influenced by natural impacts as general oceanic circulation and climate as well as human activities (EC 2011). Descriptor 7 consists of three indicators to assess the spatial characterisation and the impact of permanent hydrographical changes (EC 2010):

7.1 Spatial characteristics of permanent alterations

7.1.1 Extent of area affected by permanent alterations

7.2 Impact of permanent hydrographical changes

7.2.1 Spatial extent of habitats affected by the permanent alteration

7.2.2 Changes in habitats, in particular the functions provided (e.g. spawning, breeding and feeding areas and migration routes of fish, birds and mammals) due to altered hydrographical conditions

Indicator 7.1.1 'Extent of area affected by permanent alterations' shall assess the spatial footprint of permanent changes in hydrographical conditions caused by human activities. This indicator provides an indication of the magnitude of the potential impacts and is therefore a necessary preliminary step for the assessment of the successive indicators. Indicator 7.2.1 'Spatial extent of habitats affected by the permanent alteration' refers to the area and proportion of a particular benthic or pelagic habitat and indicates the relative importance of the changes in habitat structure. This indicator is analogue to indicator 6.1.2 on the extent of the seabed significantly affected by human activities and may therefore be assessed in a similar way. Indicator 7.2.2 'Changes in habitats, in particular the functions provided (e.g. spawning, breeding and feeding areas and migration routes of fish, birds and mammals) due to altered hydrographical conditions' is closely linked to 7.2.1 as it refers to the function of impacted habitats and subsequently the effects on functional groups. The assessment of indicator 7.2.2 is also related to Descriptor 1 on biodiversity (e.g. indicators 1.1 species distribution, 1.2 population size), Descriptor 3 on commercially exploited fish and Descriptor 4 on marine food webs. Results of these descriptors may possibly be incorporated in the hydrographical indicator 7.2.2 and need not be assessed additionally. With regard to establishing an assessment concept, indicator 7.1.1 is regarded as the basis to assess the actual extent of hydrographical changes, indicator 7.2.1 on the spatial extent of habitats affected links the area with permanent alterations to benthic and pelagic habitats and indicator 7.2.2 on changes in functioning of the habitats provides information on impacts on functional groups.

The following section describes the understanding of Descriptor 7 by Regional Seas Conventions and neighbouring countries. A proposal for the national strategy to deal with this descriptor is also presented.

Approach by OSPAR

The OSPAR Commission does not consider Descriptor 7 to be of significant priority in the short term (OSPAR 2012a). OSPAR understands the descriptor to address new developments, to focus on large-scale developments and stresses that alterations have to be permanent, i.e. persisting for more than ten years. Small-scale activities like navigational dredging or extraction of aggregates may affect hydrographical conditions on a local scale which should be sufficiently covered by existing legislation such as the Environmental Impact Assessment (EIA). Even large-scale developments and small-scale developments with potentially cumulative impacts on GES should be addressed in the EIA or Strategic Environmental Assessment (SEA) processes. These assessments are to ensure that potential impacts of new developments in marine and coastal waters are properly managed and monitored as well as appropriate mitigation measures are executed if necessary. Existing or new proposals should also be considered with regard to their cumulative impacts on any ecosystem component. OSPAR further states that currently human activities with persistent effects on hydrographical conditions predominantly occur in coastal waters and are therefore covered under the obligations of the Water Framework Directive (WFD). Necessary measures and monitoring are regarded as being entirely the responsibility of the WFD. In conclusion, OSPAR recommends that if measures have been identified under the WFD to safeguard GES and permanent changes of hydrographical conditions are restricted to the coastal waters, the requirements of the MSFD are sufficiently fulfilled and Descriptor 7 will not need further work. However, OSPAR recognises that in the future situations may occur where the WFD does not apply (i.e. outside of coastal waters) or where the EIA may not be able to effectively assess cumulative effects. Such developments may be offshore wind farms, airports or tidal barrages (OSPAR 2012a).

Approach by HELCOM

HELCOM initiated the CORESET project which aims at building a set of core indicators to implement the Baltic Sea Action Plan and to facilitate the implementation of the MSFD in the EU member states. No indicator was proposed especially for the assessment of hydrographical conditions, but a range of supplementary environmental indicators describing hydrological parameters are identified which should support the core indicators and provide information for environmental assessments. These supplementary indicators deal with parameters such as surface water salinity, water exchange between the Baltic Sea and the North Sea, hydrography and oxygen in the deep basins, development of sea surface temperature or wave climate in the Baltic Sea (HELCOM 2012). Hydrographical alterations of benthic habitats caused by thermal discharge or bridges and dams are additionally assessed within the indicator 'Cumulative impact on benthic habitats' (Korpinen et al. 2013).

Other national approaches

Other EU member states which are currently working on the implementation of the MSFD mainly follow the recommendations by OSPAR. The UK indicates that at present there are no developments in national waters which could result in the broad scale alterations of hydrographical conditions this descriptor is intended to address. Impacts arising from marine and coastal development are considered to be appropriately managed through EIA, the WFD and the Habitats Directive. Marine Plans subject to SEA will provide the framework for the licensing and consents process (DEFRA 2012). Denmark agrees with this opinion and states that impacts of construction works are regulated through special permits or construction legislation. Therefore, Denmark sees no need to establish indicators or environmental targets for Descriptor 7 (Danish Nature Agency 2012). The Netherlands declare that until 2020 no interventions are planned in the Dutch North Sea which will negatively affect GES of hydrographical conditions. Permanent changes which occurred in the past, e.g. due to the construction of the artificial island Maasvlakte, are regarded as irreversible and negative effects are already compensated. GES is therefore considered to be achieved in the current situation and existing legislation is deemed to be sufficient to safeguard the conservation of the Good Environmental Status (Marine Dutch Strategy 2012).

Proposed German approach

In order to fulfil the requirements of the MSFD for Descriptor 7 and to establish a national assessment concept, it is proposed to mainly follow the advice provided by OSPAR and to be in line with neighbouring countries. The OSPAR Commission (2012a) assumes that anthropogenic activities impairing hydrographical conditions in coastal waters are sufficiently covered by the WFD. It is further understood that small-scale and temporary activities are not intended to be addressed by Descriptor 7, as they are regarded to be appropriately regulated by Environmental Impact Assessments.

For the national strategy, the importance of this descriptor is seen in the opportunity to assess hydrographical alterations of large-scale cumulative impacts outside coastal waters which are currently not satisfactorily considered within national policy. Especially regarding the numerous offshore wind farms planned in the German Exclusive Economic Zone (EEZ) of the North Sea, the necessity to monitor and manage effects on hydrographical conditions and the marine ecosystem is strongly felt. Descriptor 7 is regarded as having high potential to assess not only the extent of these alterations but also the impacts on habitats and the functions these habitats provide for marine species. It is proposed to primarily focus on the EEZ and such large-scale projects with potentially extensive and persistent effects on hydrographical conditions as not to weaken the significance of the descriptor and to avoid overlap with other policies. While the focus should thus be on offshore wind farms, other activities leading to hydrographical alterations should also be considered in order to account for cumulative effects.

4.3 Identification of pressures

A pressure resulting from anthropogenic activities can be described as a change in a physical, chemical or biological property of the environment compared with background levels or a reference condition. Depending on the intensity, pressures have the potential to cause direct or indirect impacts on the components of the ecosystem (WG GES 2011). Pressures on hydrographical conditions can affect water column and seabed habitats and thus indirectly affect the marine flora and fauna.

Annex III, table 2 of the Directive contains an indicative list of pressures and impacts on hydrographical conditions. According to this list two pressure types relating to interference of hydrological processes are stated: ‘Significant changes in thermal regime’ and ‘significant changes in salinity regime’. Other pressure types have been suggested e.g. by OSPAR (2012b) and Tyler-Walters et al. (2001).

Based on these descriptions, a proposal for definitions of hydrographical alterations to be used in the assessment of Descriptor 7 is made (Table 4-1). Pressure types and definitions mainly follow those stated by OSPAR (2012b) as it expands the indicative list of the MSFD and ensures a more comprehensive assessment of alterations.

Table 4-1: Proposed definitions of pressures on hydrographical conditions, adapted from EC (2008) and OSPAR (2012b).

Changes in temperature Local increase or decrease in sea water temperature. Associated activities: thermal discharge from e.g. power stations, wake effect of offshore wind farms.
Changes in salinity Local increase or decrease in salinity caused by discharge or physical modification. Associated activities: freshwater discharge from e.g. waste water treatment plants, brine discharge from salt caverns, constructions that affect water flow.
Changes in water flow Changes in water movement associated with tidal streams, prevailing winds and ocean currents. Associated activities: coastal defence structures, offshore wind farms, capital dredging, extraction of aggregates.
Changes in wave exposure Local changes in wave length, height and frequency. Associated activities: coastal defence structures.

4.4 Identification of activities

The identification of pressures is followed by a description of associated human activities occurring in German waters. Activities are described in terms of their geographical distribution and their impact, which is considered to result from the spatial and temporal footprint of the associated pressures. Table 4-2 represents the list of human activities potentially causing alteration of hydrographical conditions as indicated in the Commission Staff Working Paper (EC 2011) and those activities which occur in the area considered in this project.

Table 4-2: Indicative list of human activities affecting hydrographical conditions as in EC (2011) and activities occurring in the German North and Baltic Seas.

MSFD Activity theme	MSFD Activity	Activity in German waters
Man-made structures (incl. construction phase)	Land / sea physical interaction: land claim / coastal defence	Coastal defence
		Land claim
	Port operations	Dredging
	Placement and operation of offshore structures (other than for energy production)	-
	-	Submarine power cables
Extraction of non-living resources	Dredging	Aggregate extraction
	Desalination / Water abstraction	-
Energy production	Marine-based renewable energy generation (wind, wave and tidal power)	Offshore wind farms
Land-based activities / industries	Coastal, riverine and atmospheric inputs from land - industrial discharges and emissions	Thermal discharge Brine discharge

4.4.1 Offshore wind farms

Development of offshore wind farms in German waters

The production of offshore wind energy will be one of the most important activities in terms of area utilisation, especially in the EEZ of the North Sea. At present 28 wind farms are authorised in the North Sea and three in the Baltic Sea. With the test field ‘alpha ventus’ and ‘Bard Offshore 1’ in the North Sea and ‘Baltic 1’ in the territorial waters of the Baltic Sea three wind farms are already in operation. Several others are currently under construction in the German North Sea (BSH 2013a). Applications for many more wind farms are being assessed by the regulatory authorities. In the EEZ of the North Sea, the offshore wind farms approved and applied for so far will occupy an area of more than 15% of the total surface area (Ammermann 2011).

Environmental Impact Assessment for offshore wind farms

Environmental Impact Assessments necessary for the construction of an offshore wind farm currently not include sufficient consideration of impacts on hydrological processes, presumably due to lack of knowledge on potential effects. Especially cumulative effects on prevailing currents are not adequately assessed. Impacts on hydrographical conditions described in assessments are confined to the discharge of cooling water by the turbines and the converter platforms as well as sediment warming by power cables. Both mechanisms are not supposed to have significant large-scale effects on hydrographical conditions. Changes in current regime caused by the foundations are regarded as being locally restricted and not to affect hydrological alterations beyond the single piles (BSH 2013b).

Impacts on hydrographical conditions by offshore wind farm foundations

Permanent hydrographical alterations emanating from the operation of offshore wind farms may have several causes. The foundations of the wind turbines act as one source for changes in the hydrodynamic regime. Downstream of the piles an increase in turbulence and vertical mixing occurs with an extent of approximately the diameter of the piles, but with a much stronger longitudinal extent dependent on current velocity. In the vicinity of the piles a local increase in current velocity appears, whereas in the entire wind farm a slight reduction in current velocity is expected. The mean decrease in current velocity for more than 800 wind turbines placed in an area of 1000 km² at the Borkum Reef Ground is predicted to add up to around 3 % which is not assumed to produce significant alterations of the flow regime (Mittendorf & Zielke 2002). Relocation of sediments caused by changed hydrodynamics is likewise considered as being restricted to the area around the individual piles and not to result in large-scale effects (BSH 2009).

The impact of offshore wind farm foundations on currents and water mixing in the transition area between North Sea and Baltic Sea was investigated by the BMU-funded project QuantAS-Off. The study concentrated especially on near-bottom pathways of saline and oxygen-rich North Sea waters in the Western Baltic Sea, which represent the only oxygen source for the deep water in the Central Baltic Sea. Potential constrictions of these dense bottom currents by wind farms or other artificial offshore structures could arise from additional turbulent mixing between dense and saline bottom water with less dense and brackish surface water which may weaken the inflow in the Western Baltic Sea (Burchard & Rennau 2007). The major result of this project is that the extra mixing caused by wind farms is too small to significantly modify the bottom waters flowing towards the Central Baltic Sea. Even if all wind farms which have been planned or applied for permit in the Western Baltic Sea will be built, the impact on the salinity of the Baltic Sea deep water will be negligible (IOW 2009).

Impacts on hydrographical conditions by the wake effect

Another mechanism influencing hydrographical conditions is the ‘wake effect’. Operating turbines cause distortions in the ocean wind field which leads to a long tunnel of low wind speed downstream of the wind farm. Changes in the wind field may have effects on vertical exchange and horizontal circulation patterns. Temperature changes may result in alteration of the strength and duration of the thermocline which could affect phytoplankton growth and abundance (Nerge & Lenhart 2010). The marine fauna may experience positive or negative consequences through the alteration of the hydrodynamic environment or through changes in the food web. Variations in currents and salinity can also influence the spreading pattern of larvae and the breeding or spawning areas (OSPAR 2012a).

Broström (2008) gives a theoretical description of the oceanic response to changed wind patterns in the presence of an offshore wind farm. Based on analytical models and idealised numerical experiments he states that large wind farms exert a strong effect on the circulation pattern around the installation which increases with the size of the wind farm. It is shown that wind farms force an upper ocean divergence and thus generate upwelling or downwelling. With a wind speed of 5-10 m/s, upwelling or downwelling velocities exceeding 1 m/day may be induced. As a result of changed upwelling patterns, variations in the temperature structure and availability of nutrients may appear in the vicinity of the wind farm. The upwelling of nutrient rich deep water may thus enhance primary production and affect the local ecosystem.

Nerge & Lenhart (2010) investigated the impacts of the wake effect on the marine environment by means of a simulation model. In a theoretical approach they modelled the effects caused by an operating wind farm with 150 piles located in the German EEZ during three days in summer. They observed variations in the vertical exchange between the model run with and without the wake effect, which resulted in a temperature difference of 1°C and a difference in the depth of the thermocline of about 5 m within the wind farm. The study further revealed a strong impact on the horizontal exchange which is triggered by the change in temperature distribution. In contrast to the wind-induced vertical mixing the differences in horizontal exchange are not restricted to the upper layer above the thermocline but can be seen throughout the entire water column. As a result of the wake simulation, upwelling as well as downwelling could be observed. Possible consequences on the ecosystems discussed in this study were changes in the development of biologically important fronts and an increase in primary production induced by upwelling. Nerge & Lenhart (2010) conclude that the wake effect resulting from the operation of an offshore wind farm will generally lead to a more complex hydrodynamic system which affects an area much wider than the extension of the wind farm itself.

With the prospect of probably around one hundred wind farms placed in the German EEZ of the North Sea, questions arise considering the cumulative effects of the predicted changes in hydrodynamics. The working group on theoretical oceanography of the University of Hamburg tries to find answers by means of modelling scenarios for 2030. Additionally, field measurements are planned to validate modelling results and biological impacts are investigated by the inclusion of ecosystem data. The project is currently in progress, but it could already be shown that changes in the wind pattern exert significant disturbance on vertical exchange which in turn affects temperature and salinity. Reduction in the wind field results in cells with upwelling and downwelling downstream of the wind farm with a horizontal extension of approximately 30 x 30 km. The connected vertical velocities reach magnitudes of 10^{-5} m/s or around 3 to 4 m/d, respectively. These vertical currents induce changes in temperature and salinity stratification which result in a shifting of the thermocline of up to 10 m (Ludewig & Pohlmann 2013). The precise mechanisms and the scale of effects is still vague at present, however changes in hydrographical conditions have been observed in a distance as far as 50-60 km. Reactions of the marine ecosystem remain likewise indistinct, they could prove positive or negative. For example, the increased vertical mixing could promote reef building around the piles due to the transport of oxygen-rich water downwards or may lead to an increase in algal blooms. It is hoped that including ecosystem data in the simulation model improves understanding of biological responses. (E. Ludewig, personal communication). Ludewig & Pohlmann (2013) come to the conclusion that operational wind farms cause an intensified vertical mixing in the ocean which may result in fundamental changes in the North Sea ecosystem.

Further investigations, particularly on ecological impacts of offshore wind farms, are currently planned by the research group ‘Scientific Computing’ of the University of Hamburg. A project proposal has presently been submitted to the Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit (BMU) (H. Lenhart, personal communication).

It can be concluded that experts agree on the presence of significant impacts on hydrographical conditions caused by the wake effect. According to the definitions stated in chapter 4.3, pressures exerting from the operation of offshore wind farms are thus changes in water flow, changes in temperature and changes in salinity. Existing studies and modelling results are currently not sufficient to determine the magnitude of these alterations. Impacts on the marine biota can likewise not be predicted at present. So, there are still large gaps in knowledge that can hopefully be closed in the near future by further studies and field investigations.

Possible impacts on biological components

An increase in temperature which is predicted for major parts of the EEZ due to the wake effect may result in similar effects as expected for climate change and potentially intensify the impacts. A temperature increase is expected to result in a change of the species composition with more dinoflagellates in spring and cyanobacteria in summer. Results of mesocosm studies indicate that an increase in temperature due to climate change may differentially influence the seasonal succession of phyto- and zooplankton and may potentially increase the temporal mismatch between these groups in spring. Species shifts in spring affect the food web and thus biogeochemistry and functioning of the pelagic ecosystem during summer. Changes in the composition of the spring bloom community will also influence benthic organisms. Major changes in composition may also occur in zooplankton with a predicted decrease in copepods and consequently negative effects on planktivores. Especially in the Baltic Sea, changes in salinity may have major impacts on marine functional groups with geographical shifts in the distribution of species (HELCOM 2013). Hiscock et al. (2004) discuss effects of global warming on intertidal and subtidal species. They predict major changes in the distribution of species with possible alterations of habitat distribution, if the affected species are characterising or key species. The authors expect that changes will be most apparent in mobile species or benthic species with long-lived planktonic stages in their life histories. Changes in the distribution of zooplankton species due to climate change have already been observed in the North-East Atlantic and also in the North Sea. For instance, in the North Sea the cold-water copepod species *Calanus finmarchicus* has been replaced by the warm-temperate *C. helgolandicus*, accompanied by an overall decline in abundance of *Calanus* species and impacts on higher trophic levels (OSPAR 2010).

4.4.2 Thermal discharge

Several power stations are currently releasing cooling water in coastal areas. On the coast of the German North Sea there is one coal power plant which is situated at Wilhelmshaven. In the Baltic Sea there are four power plants at present, located at Flensburg, Kiel, Rostock and Lubmin. There is no evidence of relevant impacts on the coastal thermal regime by power stations or other industries which discharge cooling water in rivers and transitional waters (BLMP 2012a, b).

The warm-water outflow from coastal power plants may cause local changes in the thermal regime with consequent impacts on the marine environment. Direct effects of a temperature increase arise from the influence on metabolic processes of organisms. Each species features a specific range of temperature where physiological processes are at an optimum. If ambient temperatures exceed the upper limit of this range, dependent on duration and level of increase in temperature the reaction of organisms may range from stress to mortality. Concerning increased temperatures, fish are generally the most sensitive among the aquatic organisms. For stenothermic species such as Clupeoids a temperature of 22°C may already prove lethal. Demersal fish such as flatfish show an increased mortality at temperatures of 23-28°C for an extended time period. However, mobile species such as fish are able to avoid areas with unfavourable conditions, such as locally restricted plumes of warm water. In contrast, many benthic invertebrates, macrophytes and plankton do not have this possibility. An increase in water temperature reduces the oxygen solubility which may have further negative consequences for fish and benthic fauna (BLMP 2012a).

In general, the release of cooling water is locally restricted and a significant increase in temperature affects only a small area. Habitat loss on a small scale and stress-induced reactions of benthic and pelagic communities may occur as a result. However, local increases in temperature may affect together with other stressors and thus result in cumulative impacts on the ecosystem. Besides, temperature changes on a small scale may interact with global warming due to climate change (BLMP 2012a).

4.4.3 Brine discharge

At present, brine discharge occurs as a result of the construction of caverns for the future storage of gas and oil. Since 2008, several caverns were established in Niedersachsen where discharges in transitional waters may also affect coastal waters. Other discharges from potash mining further upstream (e.g. in the river Weser) are sufficiently diluted so that no changes in salinity are apparent in coastal waters. Another source of salt waters effluents is a desalinisation plant located at Helgoland but due to the low discharge rate the impact on natural salinity is negligible. Significant decreases in salinity caused by freshwater discharge from wastewater treatment plants or power plants are not known in German coastal waters (BLMP 2012a, b).

Impacts of brine discharge in coastal waters on the aquatic organisms range from disturbance of respiration and metabolic processes to paralysis and lethal toxicity. The area directly influenced by the effluents suffers values far exceeding the natural salinity of sea water which can result in localised loss of habitats and benthic communities. In the near vicinity of the plume, changes in species composition and abundance of zoobenthos, macrophytes and to some degree also of fish can occur. Frequent changes in salinity caused by brine discharge mean osmotic stress which may result in reduced growth rates and increased mortality of organisms. Beside the concentration, the composition of the discharged salt water is of relevance, as e.g. the relation of sodium to potassium plays an important role in the osmotic regulation of aquatic invertebrates and fish. Each species as well as different developmental stages have individual limits of tolerance of increased salinity. The size of the area influenced by changes in salinity depends on the deviation to natural salinity and hydrological processes such as currents and mixing by which the dilution of concentrations is determined (BLMP 2012a).

4.4.4 Submarine power cables

A series of submarine power cables is currently planned and a few already exist. With the NorNed cable between the Netherlands and Norway currently only one transit power cable in the German North Sea is in operation. In the Baltic Sea two transit power cables exist which connect Germany with Denmark and Sweden. Additionally, in the North Sea the first high-voltage power cable to link offshore wind farms with the coast is already in operation, many more will be established in the near future (BSH 2013a).

Thermal radiation emanating from power cables may result in alterations of seabed habitats. The amount of heat lost and subsequent warming of the sediment depends on the type of cable, transmission rates and thermal conductivity of the surrounding substrate. Some marine organisms react sensitively to an increase in sediment temperature, therefore changes in the local benthic community around power cables cannot be excluded. Other effects of temperature rise include changes of physico-chemical conditions, e.g. alteration of redox state, oxygen and sulphide concentrations and an increase in bacterial activity. Impacts may be most severe in areas with stratified or small water bodies and the Wadden Sea. According to a guideline established by the German Bundesamt für Naturschutz (BfN) a temperature rise above the buried cable of 2 K in a sediment depth of 0.2 m is not considered to be harmful to benthic organisms (Meissner & Sordyl 2006). At the Danish wind farm 'Nystedt' sediment temperatures in the vicinity of power cables were measured. The maximum increase in a distance of 25 cm from the cable was 2.5 K while the mean increase in comparison with a reference was below 1 K. However, these preliminary results may not be transferred to other sites (OSPAR 2008a). Field studies on the effects of heat dissipation by cables are currently scarce, so that the extent of thermal radiation and actual effects remain unclear (OSPAR 2009a).

4.4.5 Spatial characteristics of permanent hydrographical alterations

The first step of the suggested concept assesses indicator 7.1.1: ‘Extent of area affected by permanent alterations‘. This should be done by modelling the extent and intensity of the pressures on hydrographical conditions caused by permanent human activities, i.e. changes in temperature, salinity, currents and waves. The intensity of alterations could be categorized with a scale ranging from very low to very high. Intensity scales should be specific for each pressure, e.g. a scale for changes in temperature and another one for changes in salinity. The modelling results should be visualised on a map by means of a Geographic Information System (GIS).

The assessment of indicator 7.1.1 should provide the basis to decide if detected alterations in hydrological conditions are permanent and significant. Criteria for significant impacts have to be determined, e.g. an impact which affects an area of more than x kilometres or an increase in temperature of more than y °C.

4.4.6 Dredging and extraction of aggregates

Dredging can be distinguished into capital dredging due to the deepening of channels or port areas as well as various construction works and maintenance dredging. Maintenance dredging involves the regular removal of sediments deposited in coastal waterways or harbours in order to preserve the designed dimensions. Permanent alteration of hydrographical conditions arises predominantly from capital dredging. Major capital dredging projects in Germany have been carried out in the North Sea ports and estuaries, e.g. Weser, Elbe and Ems estuary. Due to an expected increase in cargo shipping and of ships with deeper draughts the need for dredging will remain high or even further increase (OSPAR 2009a). At present the massive deepening of both the Weser and Elbe estuary are planned.

Comparable to capital dredging, extraction of sand and gravel is also mostly carried out by means of a trailer suction hopper dredger. Impacts of intensive aggregate dredging are therefore similar to those of capital dredging.

The impacts of dredging on hydrodynamics are to a large part site- and project-specific. Although all dredging activities can cause some change to the hydrodynamic flow, the magnitude and type of effect will depend on the size of the area dredged in relation to the overall size of the water body. In estuaries alteration is often more pronounced and includes changes in current velocity, tidal dynamics, increased wave action and a saltwater intrusion further upstream than previously (BAW 2006).

Frequent and high intensity dredging due to sand and gravel extraction may result in a strongly disturbed topography with deep tracks and furrows remaining for several years (ICES 2006). A lowering of the seabed by up to 2-3 m may be a consequence of repeated dredging in the same area. Such changes in seabed topography may in turn lead to a locally altered hydrodynamic and sedimentation regime. Current velocities may be reduced due to disruption of local current strength (Hill et al. 2011).

4.4.7 Coastal defence and land claim

Extensive lengths of coastline in the German parts of the North and Baltic Sea are protected against erosion and floods by various artificial defence structures. The prevalently applied method is the construction of dykes, especially along the North and East Frisian coast. Other hard-engineering techniques found on German coasts include seawalls, bulkheads and groynes. Soft-engineering coastal structures, such as dunes and beach nourishment, are increasingly being employed as they are regarded to have less severe impacts on hydrography and sediment transport (OSPAR 2009c). In Mecklenburg-Vorpommern coastal defence dunes are the prevalent structure raised for the protection of the coast (MLUV 2009).

Today land reclamation from the sea or coastal wetlands mainly takes place for port expansions and associated industrial developments. Historical land claim sites were the polders in the Netherlands or the Köge in Schleswig-Holstein. At present the construction of Maasvlakte 2, the extension of the Rotterdam harbour is one of the largest land gain projects. In Germany several small-scale land claim projects were carried out, e.g. for the construction of the Jade-Weser-Port in Wilhelmshaven and for port expansions at Bremerhaven. Further projects in the near future are not foreseen (OSPAR 2008b).

The impacts on hydrographical conditions caused by coastal defence structures or land claim vary strongly according to the size and type of the structure and the surrounding conditions. Defence structures may cause changes in water and sediment circulation, transport patterns and tidal prism (OSPAR 2008b). Especially hard defence structures influence the patterns of wave movement and induce wave diffraction (EC 2004).

Table 4-3 summarises the information from this chapter and links human activities in the German EEZ with the definitions of pressures on hydrographical conditions from Table 4-1. In Figure 4-1 human activities and associated pressures on hydrographical conditions which have been identified in this chapter are compiled and physical, chemical and biological features which may be impacted are listed.

Table 4-3: Summary of human activities and associated pressures on hydrographical conditions in German waters.

Activity	Geographic distribution	Pressure	Description of pressure
Offshore wind farms	3 wind farms in operation in Baltic and North Sea, 28 authorised, around 90 planned, most of them in the EEZ of the North Sea	changes in temperature	wake effect induces increased vertical mixing with effects on temperature, salinity and stratification
		changes in salinity	
		changes in water flow	wake effect induces upwelling and downwelling cells and changes in horizontal currents presence of piles locally alters prevailing currents
Thermal discharge	coastal waters	changes in temperature	release of cooling water from power plants
Brine discharge	coastal waters (Niedersachsen)	changes in salinity	release of brine from the construction of caverns
Power cables	several power cables in the North and Baltic Sea, more power cables planned for offshore wind farms	changes in temperature	local increase in sediment temperatures due to transfer losses
Capital dredging	predominantly estuaries and harbours, North and Baltic Sea	changes in water flow	changes in current velocity and / or tidal dynamics subsequent to capital dredging
		changes in wave exposure	increase in wave energy
Aggregate extraction	several areas licensed for extraction in both the North and the Baltic Sea	changes in water flow	reduction of current velocity
Coastal defence	extensive parts of the coastline in both North and Baltic Sea	changes in water flow	changes in near-shore currents and tidal dynamics
		changes in wave exposure	alteration of wave movement
Land claim	near harbours in the North Sea	changes in water flow	changes in near-shore currents and tidal dynamics
		changes in wave exposure	alteration of wave movement

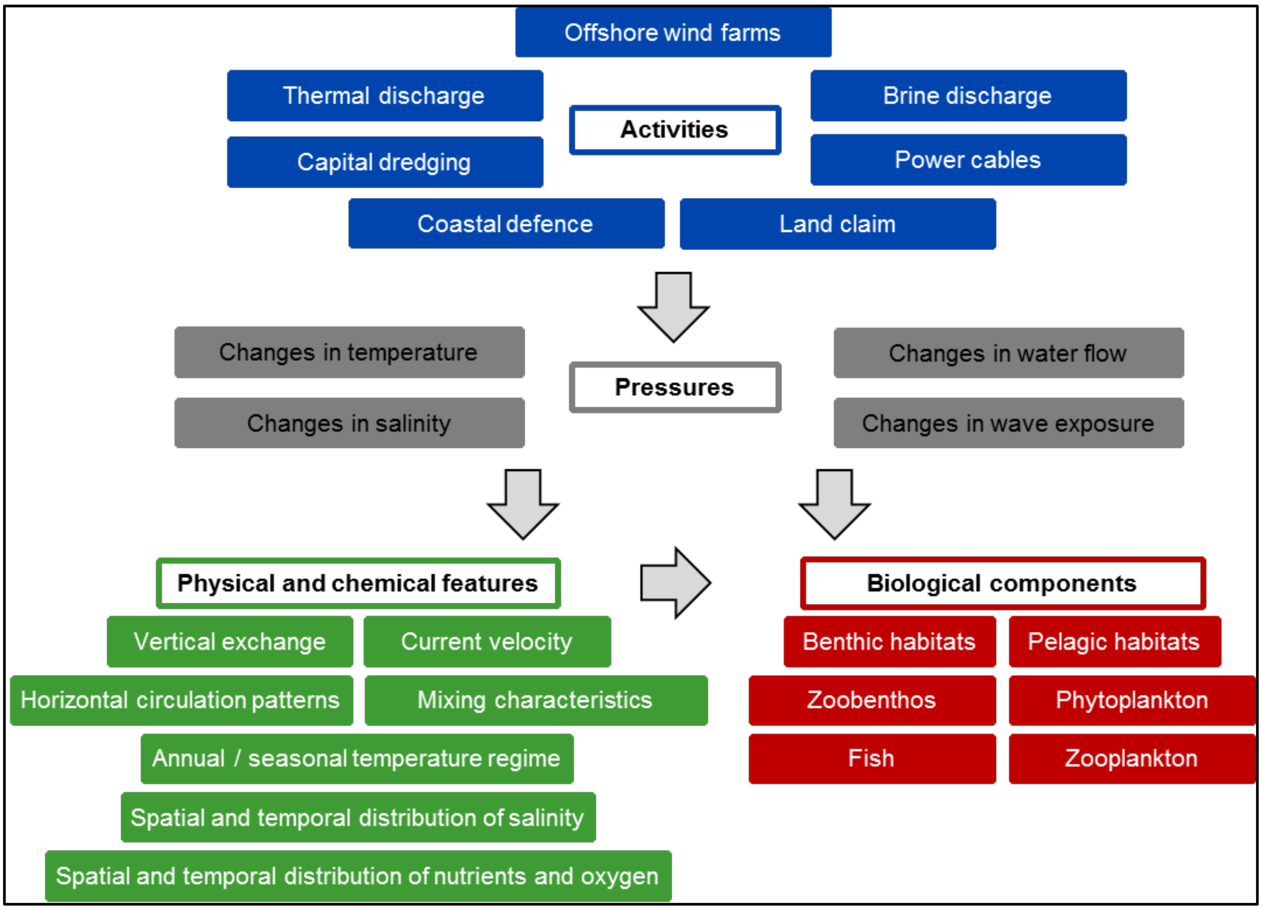

Figure 4-1: Relationship between human activities and hydrographical pressures relevant in German waters and physical, chemical and biological components affected.

4.5 Considerations for an assessment concept

Based on considerations stated in chapter 4.2, the focus of a national assessment concept is supposed to be on future large-scale projects in the EEZ with permanent effects on hydrographical conditions, which are currently not appropriately managed by existing national policy. With regard to the activities listed above, these preconditions are fulfilled only by operational offshore wind farms. While the focus of an assessment approach should thus be on offshore wind farms, other activities leading to hydrographical alterations should also be considered in order to account for cumulative effects.

The following section briefly outlines the draft of a possible assessment concept for Descriptor 7. Due to large gaps in knowledge about permanent hydrographical alterations and their impacts on biological components it is currently not possible to be more detailed about the approach. Figure 4-2 shows the steps of a possible assessment concept which is based on the successive assessment of the individual indicators.

It is suggested to base the assessment on impact modelling as a pragmatic and cost-effective approach. For some aspects monitoring data may be available (e.g. hydrographical data, phytoplankton monitoring) which could be used to support the modelling results.

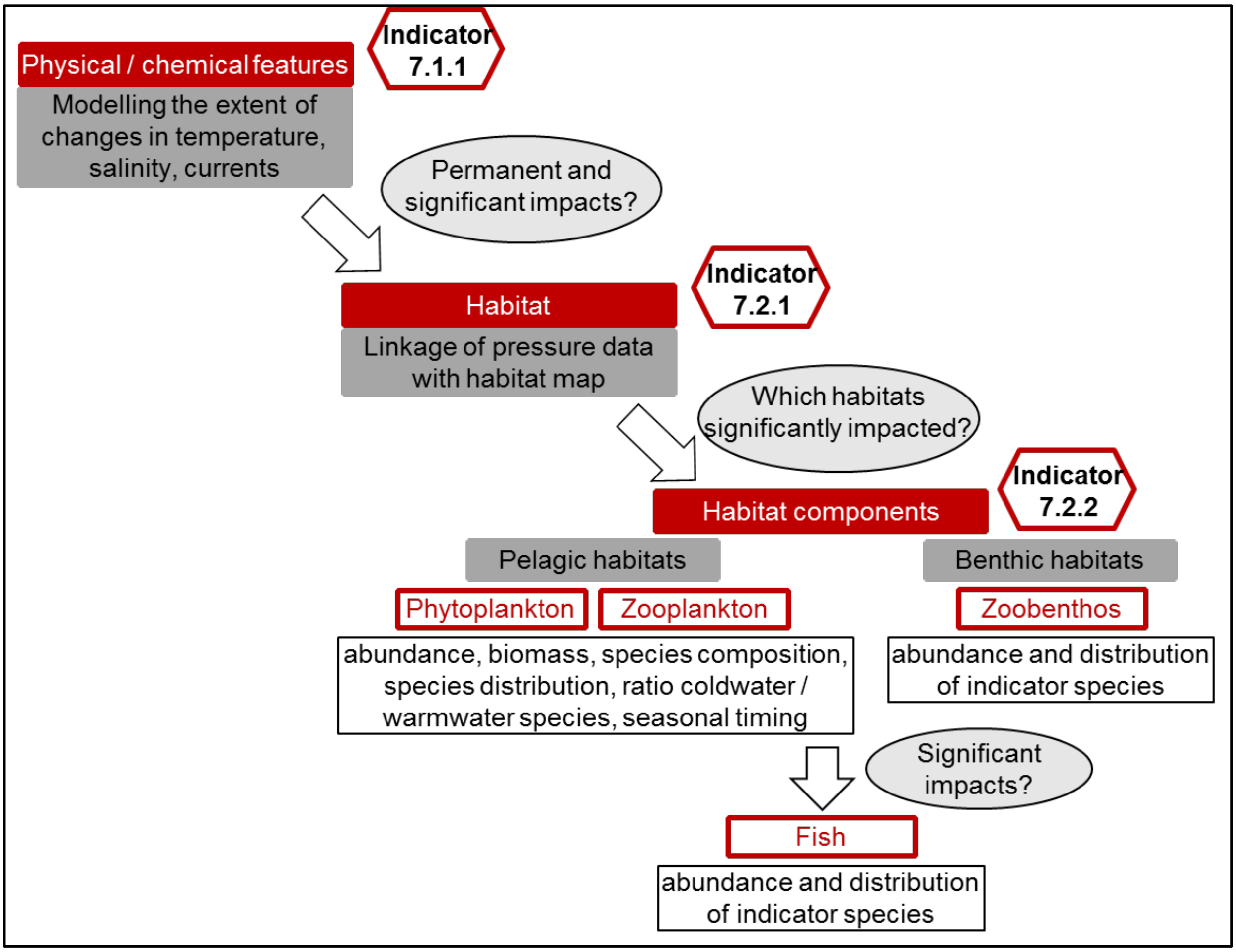

Figure 4-2: Draft of a possible assessment concept for Descriptor 7.

4.5.1 Spatial extent of habitats affected by permanent hydrographical changes

If extent and intensity of hydrographical alterations result in significant and permanent alterations the next step is to examine effects on ecosystem components. Indicator 7.2.1 focuses on the extent of habitats affected by permanent changes. The assessment of this indicator would thus imply the linkage of the modelled pressure map established by the assessment of indicator 7.1.1 with a habitat map of the North and Baltic Seas.

The assessment of indicator 7.2.1 should allow for the identification of the extent and intensity to which benthic and pelagic habitats are impacted by significant and permanent hydrographical alterations.

Habitat types

According to the indicative list of characteristics in Annex III, table 1 of the MSFD three habitat types are specified (EC 2008):

- Predominant habitat types - The predominant seabed and water column habitat types with a description of the characteristic physical and chemical features, such as depth, water temperature regime, currents and other water movements, salinity, structure and substrata composition of the seabed,

- Special habitat types - Identification and mapping of special habitat types, especially those recognised or identified under Community legislation (the Habitats Directive and the Birds Directive) or international conventions as being of special scientific or biodiversity interest,
- Habitat types which merit special reference - Habitats in areas which by virtue of their characteristics, location or strategic importance merit a particular reference. This may include areas subject to intense or specific pressures or areas which merit a specific protection regime.

The Commission Staff Working Paper (EC 2011) gives further instructions on definitions of habitat types. **Predominant habitat types** are closely linked to level 3 of the EUNIS habitat classification scheme. Seabed habitats are classified according to their depth (littoral, shallow, shelf, bathyal and abyssal) and their substrate. Substrates are differentiated in rock and biogenic reef and sediment habitats (coarse, sand, mud, mixed). Water column habitats are differentiated due to the level of salinity and the region. Based on the list presented in EC (2011), the following habitat types occur in German waters:

Seabed habitats:	Water column habitats:
• littoral sediment	• reduced salinity water
• sublittoral coarse sediment	• variable salinity (estuarine) water
• sublittoral sand	• marine water: coastal
• sublittoral mud	• marine water: shelf
• sublittoral mixed sediment	

Special or listed habitat types refer to those identified under several regulatory frameworks such as the EU legislation or international conventions (EC 2011). Habitat types in German waters belonging to this category are therefore priority habitats of the Habitats Directive, protected biotopes according to § 30 BNatSchG (Federal Nature Conservation Act), the OSPAR list of threatened and/or declining species and habitats and the HELCOM red list of marine and coastal biotopes and biotope complexes. The following set of habitat types occurs in German coastal and marine waters:

- seagrass beds
- macrophyte meadows and beds
- *Mytilus edulis* beds
- sea-pen and burrowing megafauna communities
- *Sabellaria spinulosa* reefs
- shell gravel bottoms
- gravel bottoms with *Ophelia* species
- species-rich habitats on coarse sands, gravel or shell debris
- reefs
- sandbanks

Habitats in particular areas can include areas subject to specific or multiple pressures and are therefore likely to entail risks to marine biodiversity, marine ecosystems, human health or legitimate uses of the sea, or areas already designated or which should be designated due to various forms of spatial and management protection. Currently, particular habitats have neither been identified by the European Commission nor by the Regional Seas Conventions.

4.5.2 Impacts of permanent hydrographical alterations on habitat components and functions

Indicator 7.2.2 covers 'changes in habitats, in particular the functions provided'. With regard to the possible impacts of offshore wind farms especially changes in food webs should be considered. As indicated in the Commission Decision on GES criteria (EC 2010), the term habitat addresses both the abiotic characteristics and the associated biological community. The associated biological features of water column habitats would thus be phytoplankton and zooplankton communities, while seabed habitats are combined with angiosperms, macro-algae, invertebrate bottom fauna and associated vertebrate fauna.

The focus of the assessment of habitat components should initially be on the pelagic system and the phyto- and zooplankton communities, as first effects should be detected within these groups. Possible impacts on phyto- and zooplankton due to hydrographical alterations have been described in chapter 0. Parameters which could be modelled or measured in order to assess these impacts are e.g. abundance, biovolume, species composition and distribution, the ratio of coldwater to warmwater species or seasonal timing of phytoplankton spring bloom and succession by zooplankton. Some parameters should be selected which best reflect changes caused by pressures on hydrographical conditions. Regarding benthic habitats, it is supposed that impacts may be sufficiently detected by assessing the abundance and distribution of selected indicator species of zoobenthos. Hiscock et al. (2004) give a key for determining likely effects of temperature increase on marine species, which could be used to identify benthic species indicating changes in temperature.

If significant changes can be modelled or observed in phyto-, zooplankton and zoobenthos communities, as the next trophic level demersal and pelagic fish species should also be considered. Like for zoobenthos, it is recommended to select a set of indicator species and to focus on abundance and distribution of these species.

Assessment of impacts on biological components may be a combination of modelling and monitored data. Especially for phyto- and zooplankton some long-term time series are available and in coastal waters phytoplankton and nutrients are monitored regularly. These may be used to calibrate and validate modelling results.

4.6 Considerations for baselines

Within the MSFD the baseline is defined as a state or condition against which the Good Environmental Status can be assessed. Therefore, determining the baseline state is an essential precondition to the development of reasonable GES targets and the subsequent assessment of the present state of ecosystem components in relation to these targets. Several methods for setting baselines are described in literature and shall at first briefly be presented here:

<u>Reference state:</u> Baselines can be set as a state or condition in which impacts from anthropogenic pressures are absent or negligible. This could be a presently existing area without impacts, the historical state or a modelled reference state.

<u>Past state:</u> Baselines can be set as a state in the past, based on a time-series dataset for a specific habitat.

<u>Current state:</u> Baselines can be set as the date when a particular environmental directive or policy comes into force or the first assessment of state (OSPAR 2012b).

The operation of offshore wind farms is the activity which is supposed to be the main source of pressures on hydrographical alterations. The first offshore wind farm in German waters was put into operation in 2009. Therefore, it is proposed to use the 'current state' as a baseline, which could be the date of the implementation of the MSFD in 2008. This would be in line with recommendations by OSPAR (2012a) and underline the demand to focus on future projects. The advantage would also be to have a baseline under current physiographic, geographic and especially climatic conditions.

4.7 Considerations for GES targets

Targets for the Good Environmental Status represent boundaries or thresholds between the acceptable and unacceptable status of the marine environment (GES or below GES). At that point it is not yet possible to define GES targets for Descriptor 7, there shall be rather some suggestions presented here.

Generally, targets are established with reference to a baseline. A target associated with 'current state' as a baseline is typically directed towards the prevention of any further deterioration (OSPAR 2012b). The GES target recommended by OSPAR (2012a) for Descriptor 7 is to minimize impacts resulting from alterations of hydrographical conditions. This target may be achieved by preventing further deterioration and by ensuring that the area of different habitat functions (e.g. feeding zones, spawning areas etc.) stay in comparable quantity or quality (OSPAR 2012a). The mitigation of impacts should be regarded within Environmental Impact Assessments and may include measures such as the shutdown of wind turbines at a certain wind speed and/or direction. A possible way to achieve these targets could as well be to set thresholds for hydrographical changes, e.g. 'temperature changes must not exceed x °C'. Stricter targets should be applied to habitats with specific importance like spawning or feeding areas.

4.8 Conclusion

In this report human activities which may impact hydrographical conditions and their associated pressures have been presented. Impacts on ecosystem components have been described as far as currently foreseeable. It has been underlined that the focus of Descriptor 7 should be on future large-scale projects in the EEZ with permanent effects on hydrographical conditions, which are currently not appropriately managed by existing national policy. In German waters this would apply to the operation of offshore wind farms and their cumulative effects. With the first larger wind farm in the North Sea with 80 turbines just being in operation a few months and due to large gaps in knowledge on associated pressures and impacts on hydrographical conditions and ecosystem components, it is currently not possible to present a detailed assessment concept for Descriptor 7. Instead, a first draft of a possible concept is briefly outlined. The purpose is to have a basic framework which would be open to changes and adaptable for future developments in research.

For the further development of the assessment approach the cause-effect relationships between hydrographical alterations caused by offshore wind farms and biological reactions should be especially regarded. Another factor which has to be considered is the impact of climate change which could interfere with effects from offshore wind farms and possibly intensify impacts. Natural variations in hydrography should also be taken into account and be distinguished from human impacts. Scientific research on human activities and their effects on the marine ecosystem should be further closely observed and integrated in the assessment concept.

The importance of this descriptor as the opportunity to assess hydrographical alterations of large-scale cumulative impacts has been highlighted in this report. Although with three wind farms currently in operation in German waters significant effects on hydrographical condition are not expected, the further expansion of offshore wind farms especially in the North Sea may cause fundamental changes in the marine ecosystem

Therefore it is strongly recommended to develop the assessment concept further with improved scientific knowledge on changes and impacts.

5 Work package 4: Pollutants in the marine environment (Descriptor 8)

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[a)] Chapter 5.1 to 5.3

[b)] Chapter 5.4

5.1 Criterion 8.1 Concentration of contaminants

5.1.1 Introduction

The EU's "Marine Strategy Framework Directive" (MSFD)[13] aims at achieving or maintaining a Good Environmental Status (GES) in the marine environment by 2020 at the latest. According to Article 3.5 of the MSFD, a GES is defined as: "The environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable, thus safeguarding the potential for uses and activities by current and future generations…"

The MSFD qualitative descriptors for GES include two descriptors (8 and 9) for the status of hazardous substances and their effects. Descriptor 8 states that "Concentrations of contaminants are at levels not giving rise to pollution effects". The descriptor includes two criteria: the concentrations of contaminants and the effects of contaminants.

With regard to the MSFD and the WFD the European Commission has outlined criteria and methodological standards on the good environmental status of marine waters in a European Commission decision (EC, 2010)[14].

This EC-Decision on the GES criteria (EC, 2010) only includes one broad indicator for the criterion 'concentration of contaminants':

"8.1 Concentration of contaminants

— Concentration of the contaminants mentioned above, measured in the relevant matrix (such as biota, sediment and water) in a way that ensures comparability with the assessments under Directive 2000/60/EC (8.1.1)"

[13] Directive 2008/56/EC

[14] European Commission Decision 2010/477/EC

The EC (2010) furthermore defines the relevant substances or substance groups as the ones that exceed the quality standard set in different matrices (water, sediment, biota), should be comparable with the list of Priority Substances under the “Water Framework Directive” and the subsequent “Priority Substances Directive”[15], may additionally be other substances which are considered significant and should also be taken into account. As the Water Framework Directive (WFD)[16] also includes coastal waters, the coastal waters form a common study area of the MSFD and WFD.

With the introduction of the MFSD, the question arises which substances should be monitored in the marine environment to ensure GES and how these substances should be monitored (matrix). More than 500 substances are potentially harmful to the marine environment[17], but in the light of limited resources, not all can be monitored. This report gives recommendations on important criteria for the selection of substances and aims to identify the most relevant pollutants for the marine environment. Factsheets for candidate substances are compiled with the reasons for inclusion and exclusion of these substances in the monitoring procedure under D 8.1 (chapters 5.1.7.2.1 and following). Based on these factsheets, different sets of substances are proposed.

5.1.2 Background

One of the main pressures affecting the marine environment today results from chemical pollution: the release and effects of chemicals in marine environments, whether transported in solution, bound to particles, or incorporated in biota and originating from industrial, agricultural and residential waste in liquid, gaseous or solid form, or from diffuse emissions. Worldwide, the production of chemicals is increasing with the total production volume expected to double in comparison with 2000 levels by 2024. About 100,000 chemicals are available on the EU market (ESF, 2011[18], Wilson and Schwarzman, 2009). The marine environment as “final sink” for many pollutants reflects this development (SRU 2004[19]). It receives a cocktail of chemicals, and the effect long term exposure, continuing bioaccumulation and combined effects of chemicals on the marine ecosystem are very difficult to predict.

Almost all anthropogenic pollutants eventually find their way into the sea. Some of these pose an environmental risk due either to high input levels, persistence and accumulation, or even direct toxic impact. Risks of this type are posed in particular by heavy metals, some persistent organic compounds and oil inputs. Endocrine disruptors and polar pollutants also give cause for increasing concern. Both the North and the Baltic sea remain under considerable – in some areas increasing – pressure of use (SRU 2004).

The occurrence of marine pollutants is the result of direct releases (e.g. from shipping), land-based river runoff or atmospheric deposition, all of which contribute significantly to marine pollution. Contamination of the marine environment by chemical substances gives rise to considerable concern as it may result in serious adverse effects on the structure and functioning of ecosystems, the goods and services they provide, and on

[15] Directive 2008/105/EC

[16] Directive 2000/60/EC, resp. Directive 2013/39/EU, amending 2000/60/EC

[17] More than 500 substances were identified in this project to be potentially harmful for the marine environment (see chapter 5.1.6.2)

[18] Monitoring Chemical Pollution in Europe’s Seas: Programmes, Practices and Priorities for Research

[19] SRU 2004: Marine Environment Protection for the North and Baltic Seas. Special Report

human health. Unwanted chemical substances may, for example, reduce biodiversity and productivity in marine ecosystems, resulting in a reduction and depletion of human marine food resources (ESF 2011).

Chemicals can cause not only direct intoxication and obvious effects such as death of marine biota, but they can also cause more subtle adverse effects such as impairment of the reproductive, hormone and immune systems. As stated by Paracelsus (16th century) it is “the dose (concentration) that determines if a substance is a poison”. From this it follows that for chemicals which are toxic at very low concentrations; release into the (marine) environment should be prevented, taking into account the precautionary principle. Other substances - which may not cause a direct effect – may cause indirect impacts through food-chain transfer (ESF 2011).

Many of the most notorious pollutants are persistent including for example halogenated organics. This means they are only slowly degraded in natural environments and accumulate over time. The risks posed to ecosystems and human health by long term exposure, continuing bioaccumulation and the combined effects of “cocktails” of different chemicals are very difficult to predict. But once persistent substances are released into the environment, it is almost impossible to remove them again. Therefore, whatever the potential effects of these substances on ecosystems and wildlife are, they are irreversible.

Input of substances into the marine environment occurs via water (riverine input, dissolved or particle bound), land (erosion and leaching from coasts, transport and discharge of dredged material) or air (gases, aerosols, airborne particles) or directly to the sea (from offshore installations and shipping).

For the purpose of this project, the general term “hazardous substances” is used for contaminants and pollutants including organic and inorganic substances. The term contamination is simply the presence of a substance where it should not be or at concentrations above background (Chapman, 2007). Pollution is contamination that results in or can result in adverse biological effects. All pollutants are contaminants, but not all contaminants are pollutants (Chapman, 2007).

Complex patterns of emission

A general overview of sources and pathways of hazardous substances is given in Figure 5-1. “Classical” emission sources are industries, where hazardous substances are produced or used (production or industrial use, see left hand side of graph). Usually, industries have “technical barriers” in place to tackle their waste streams e.g. wastewater treatment plants or flue gas scrubbers. But sometimes these technical barriers are not effective against hazardous substances. Also, emission can bypass abatement processes e.g. via dust or even mismanagement.

Another pathway for hazardous substances out of the industrial domain is via products for private use. Although every product usually contains only small amounts of hazardous substances, a large stock can pile up in urban areas. Consequently, urban areas or the urban stock of hazardous substances are also emission sources.

The urban stock also includes imported products and “historical” products with a long technical lifespan. The emission from urban stock is mostly channelled through urban infrastructure systems, but standard treatment is not very effective for many hazardous substances. Emission from urban areas can also bypass urban infrastructure systems via informal pathways, e.g. via activities like (illegal) burning of household or electricity waste, illegal disposal of waste, illegal discharge of wastewater or losses from sewer systems.

For some substances, unintended production (e.g. dioxins from combustion processes) and geogenic sources (e.g. background concentrations of heavy metals) also play a role.

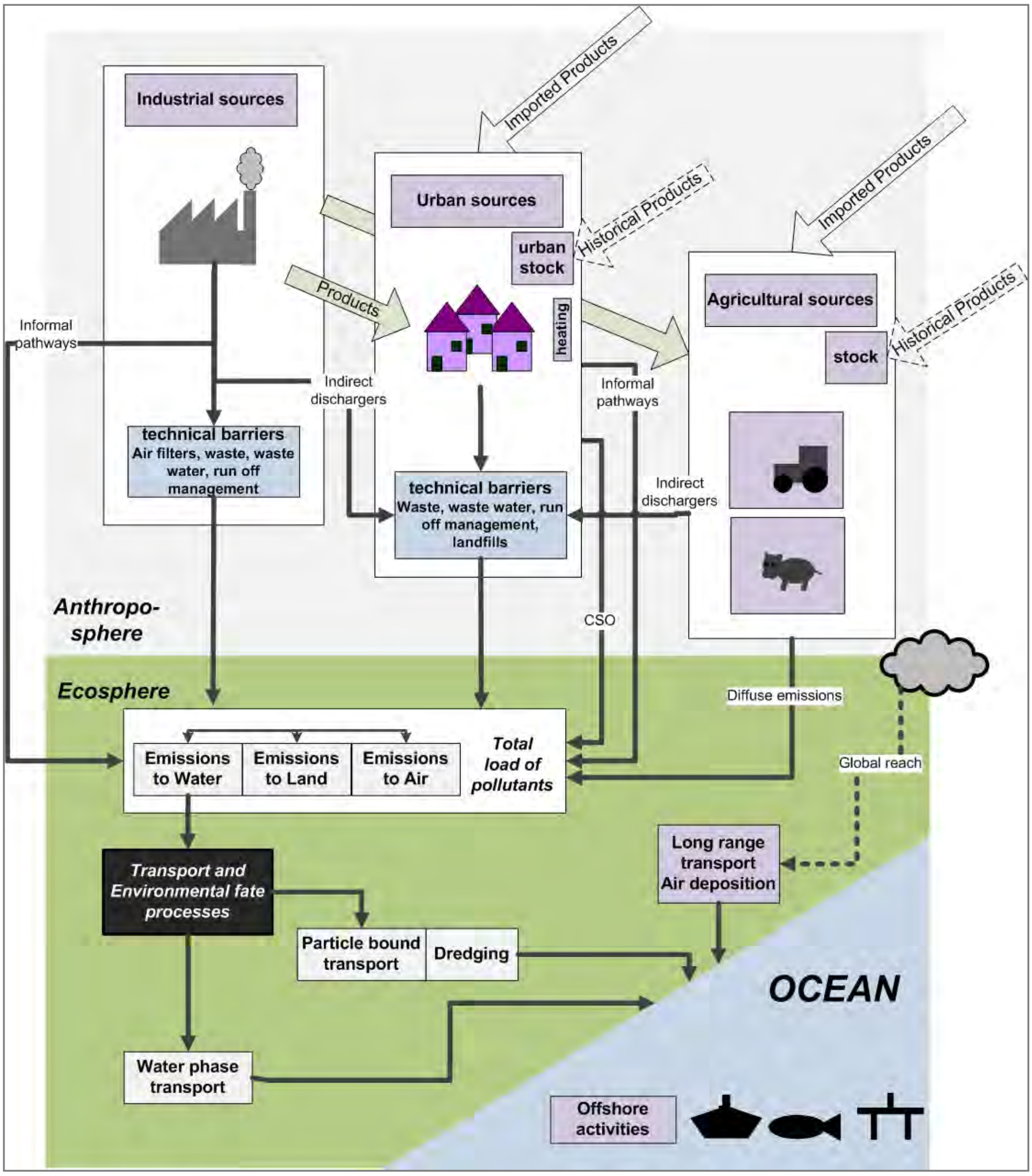

Figure 5-1: Overview of sources and pathways (adapted and extended from Mathan et al. 2012); www.cohiba-project.net.

As Figure 5-1 shows, there is a multitude of sources and pathways for hazardous substances to the marine environment. Generally speaking, sources are Industry (Large point sources: Production of chemicals, manufacturing / formulation of products), non industrial sources: urban and residential sources (stock of products; wastewater and landfills, as well as "informal pathways" which bypass the technical barriers: wrongly connected pipes, combined sewer overflow, littering etc.) and diffuse emissions e.g. from agricultural sources including application of plant protection products.

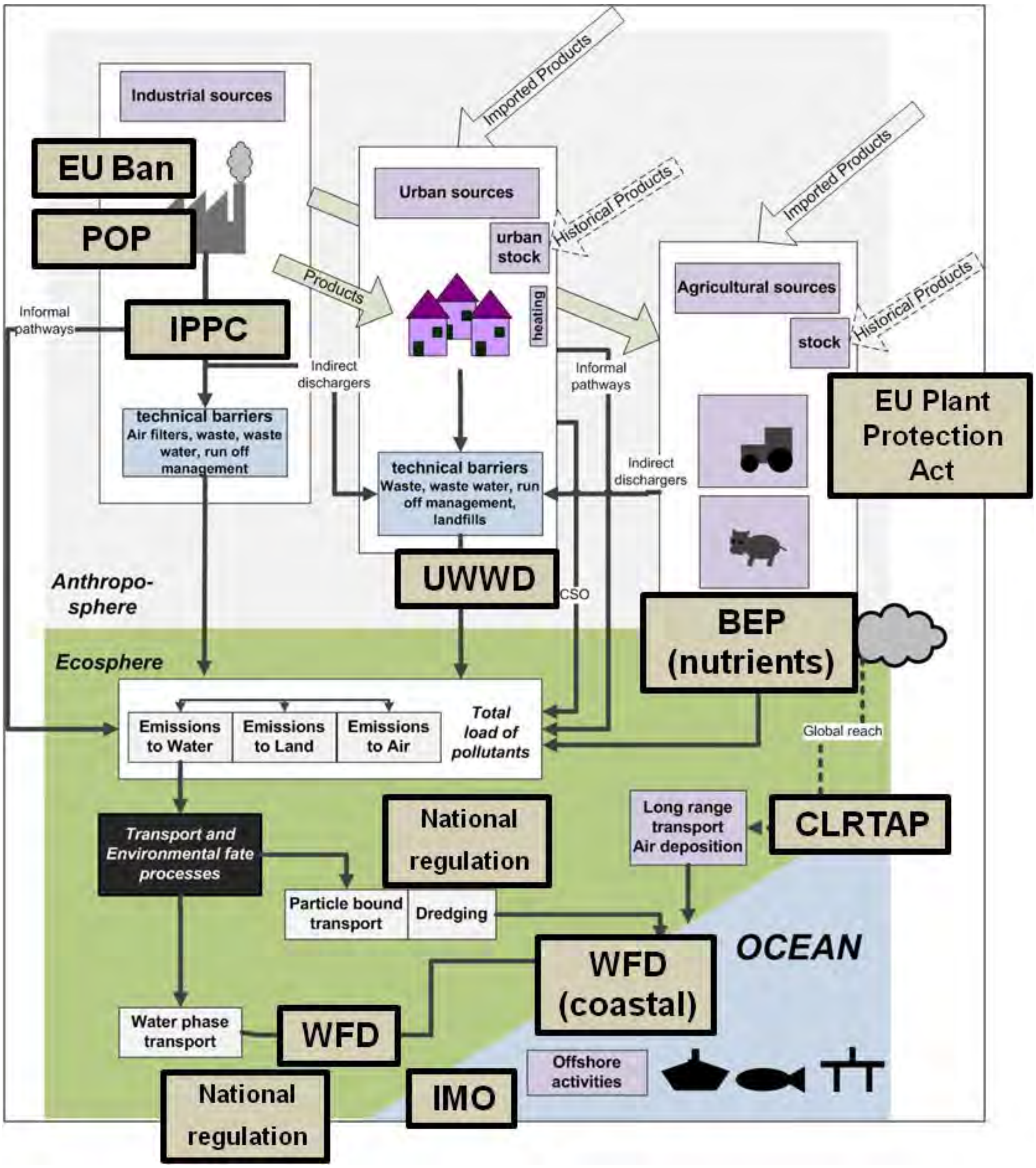

Figure 5-2: Overview of sources and pathways with relevant regulations (adapted and extended from Mathan et al. 2012).

For these sources, different regulations exist on the national, european and international level. Regulations act as a “valve“ for emissions., as shown in Figure 5-2.

Some regulations target substances at the primary source regulating the production (EU ban, EU biocides directive, EU regulation on plant protection products[20] Stockholm Convention on Persistent Organic Pollutants (Stockholm POPs), Convention on Long-range Transboundary Air Pollution (CLRTAP).

Others set emission limits for waste streams leaving the anthroposphere (IPPC, UWWD, BEP for nutrient reduction). In addition, there are also national regulations, for example for dredging or specific industries.

Others regulate the status of surface waters (WFD, MSFD). In addition to the EU regulations, there are also national (and multi national e.g. for multi national river catchments such as Rhine or Odra) regulations, for example targeting additional substances in surface waters (river specific pollutants). Other regulations target specific substances in specific applications, such as IMO banning anti-fouling substances in marine uses.

Dynamic patterns of emission

The source pattern of pollutants is a complex network per se, and it is also dynamic. Due to regulations and other changes (e.g. in product design or waste management), the pattern of emission is dynamic for many hazardous substances (see Figure 5-3).

The story usually goes like this: A substance causes a harmful effect in the environment or is detected in environmental samples due to improved measurement techniques. Then the hazardous substance is regulated and industrial emissions decrease. Often, the production volume and emission load of industrial facilities is only revealed during the regulatory process, and is therefore outdated shortly after publication. This is a question of confidential business information (CBI) of the chemical industry.

Generally speaking, as industrial sources become more regulated, the overall burden for the environment decreases. But the importance of non- industrial sources increases in proportion, making the source pattern more complex. For many hazardous substances, changes of emission patterns could be observed in the last decades. Due to regulations and consequent emission reduction measures at large industrial point sources, emissions from industrial sources were considerably reduced leading to a reduction of emission for a great number of hazardous substances to the environment. Non-industrial emission sources are therefore becoming more important. These sources have a more diffuse character i.e. emissions from the use of products in urban areas or contaminated sediments.

Following the ban from the market, a time lag occurs until the concentrations in the (marine) enviroment of those hazardous substances show decreasing trends. This is mainly due to substance characteristics: persistence and cycling within the marine ecosystem (e.g. resolving of Mercury (Hg) from contaminated sediments). Many so called “legacy21” hazardous substances can be detected in environmental samples long after their phasing out, e.g. DDT in the arctic (Muir and de Wit 2010[22]). Different lag times are also due to e.g. different lifetimes of products in urban stock.

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[20] EC 1107/2009, approved substances are specified in EU No 540/2011

[21] legacy hazardous substances describes a substance that is (highly) regulated but still occurs in environmental samples, for example obsolete pesticides such as DDT

[22] Trends of legacy and new persistent organic pollutants in the circumpolar arctic: Overview, conclusions, and recommendations

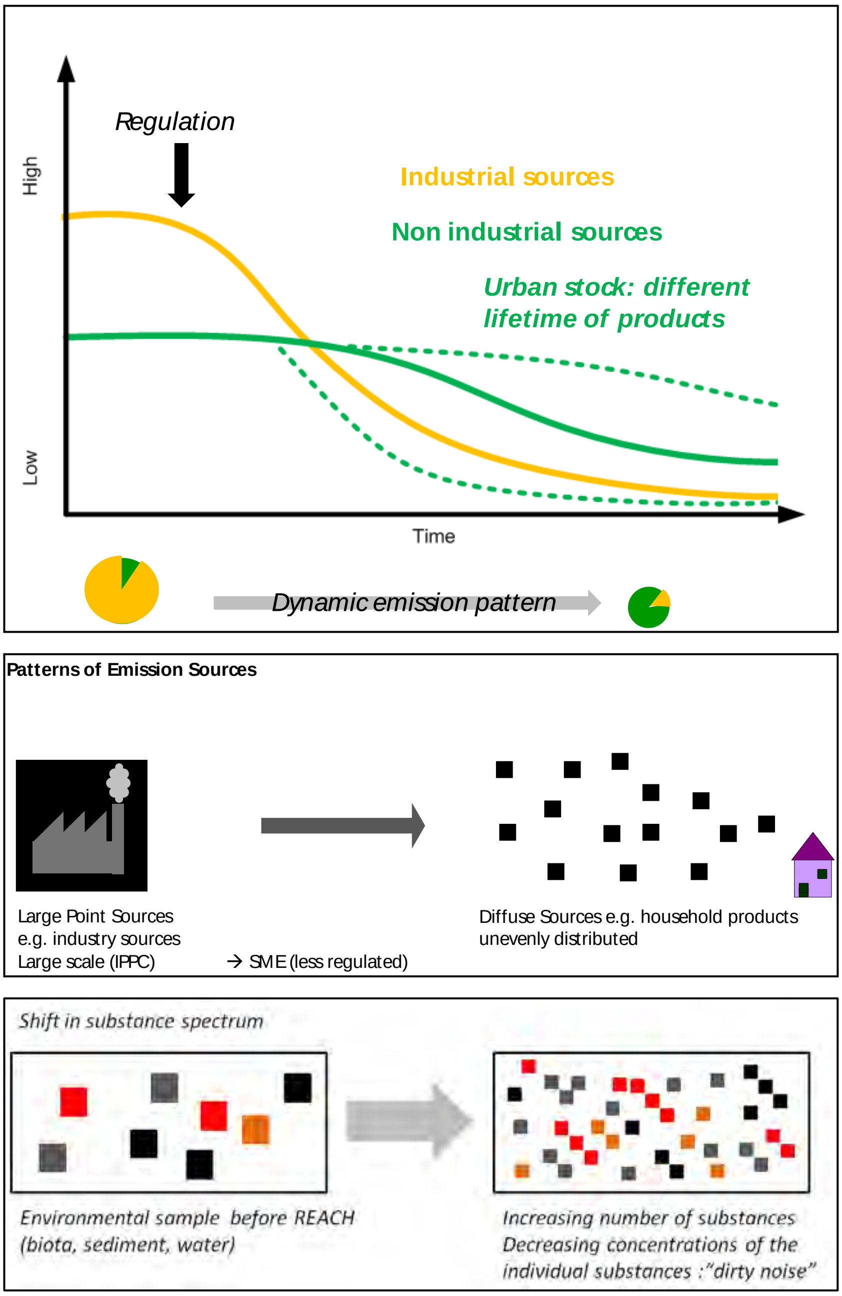

Figure 5-3: Dynamic emission pattern (A changes in time, B changes in spatial distribution and C changes in substance spectrum).

Also, a shift in the substance spectrum can be observed over the last two decades, with an increasing number of hazardous substances found in environmental samples at decreasing concentrations ("dirty noise"). This trend is partly due to the intrest in "chemical cocktails" leading to more multi-substance analysis, partly due to advances in analytics which progresses to detect an ever-increasing number of substances in ever decreasing concentrations, but mainly due to progresses in the chemical industry putting more and more substances on the market and the fact that most highly regulated substances are substituted by multiple other (less hazardous) substances.

In synopsis, the complexity of the chemical pollution problem has various dimensions (see Figure 5-4).

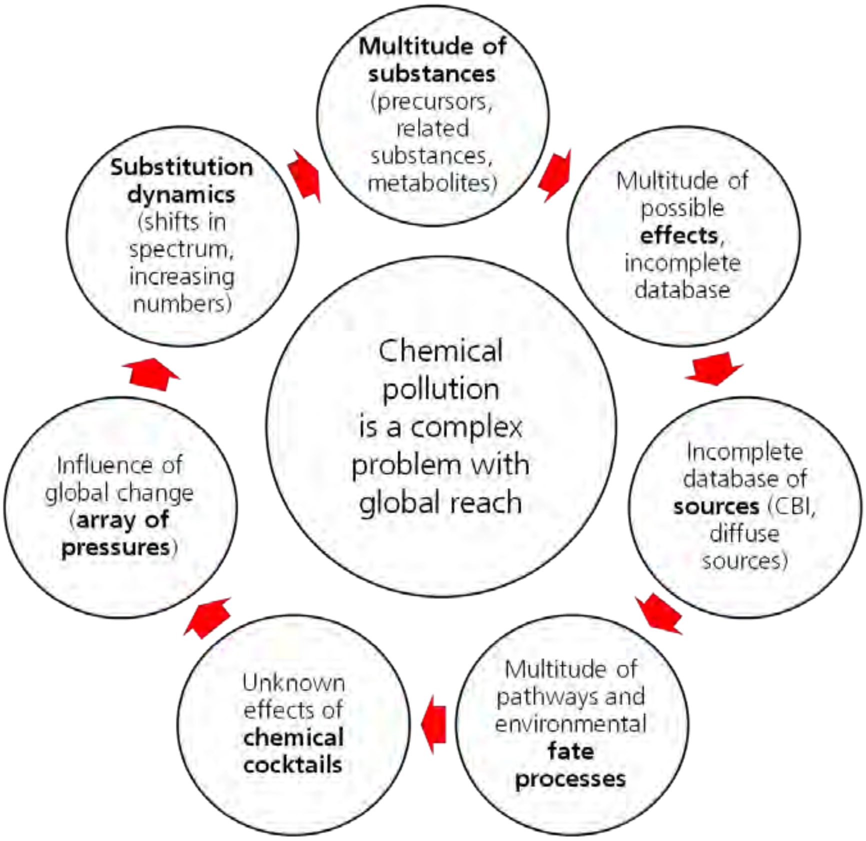

Figure 5-4: Various dimensions of the complexity of the chemical pollution problem.

5.1.3 Relevance for the marine environment

Given the dynamics and complexity of emission patterns and the incomplete database, it becomes obvious that all recommendations are subject to high uncertainities. But there is a need to tackle the chemical pollution problem now, making use of the best knowledge available.

In theory, the question is simple. To find the most harmful substances, the environmental concentration of the substance X has to be put into relation to the concentration that causes harm to the weakest link in the ecosystem. This can be expressed as the ratio of PEC to PNEC. The substances that have the highest ratio of PEC to PNEC are the most relevant.

In practice, the question is very complex. On the ecotoxicological side (PNEC) the abundance of possible target species and ecotoxicological effects contributes to the complexity, let alone the effect of chemical cocktails (concerted effects of chemicals) and the cross effect with other stressors, such as climate change and eutrophication.

Hahn et al. (2013) published a study that describes the extent of variability of PNEC. The authors chose 5 chemicals to represent well-known substances for which sufficient high-quality aquatic effects data were available: ethylene glycol, trichloroethylene, nonylphenol, hexachlorobenzene, and copper (Cu).

The observed variation in the PNECs for all chemicals was up to 3 orders of magnitude, and this was not simply due to obvious factors such as the size of the data set or the methodology used. Rather, this was due to individual decisions of the assessors within the scope of the methodology used, especially key study selection, acute versus chronic definitions, and size of assessment factors.

On the load side (PEC), the situation is similarily complex. The concentration in the environment may be subject to considerable changes in time and space (hot spots and hot times of emission). For substances showing high fluctuations in their concentrations over time, there is a high chance of overlooking short term peak concentrations, especially for compounds with an expected intermittent release, such as pesticides, when applying the widely used method of monthly or even quarterly water grab samples, as for WFD-compliant sampling. This also relates to the question whether to use maximum or average concentrations.

Another issue is how to treat data below the limit of quantification (LOQ). In chemical databases, the LOQ and LOD (limit of detection) are often not reported, leading to uncertainties whether a concentration is just below LOQ or even below LOD. In Environmental Risk Assessments, half of the reported LOQ value is often used to consider the worst case, i.e. concentrations just below the LOQ. The risk of these substances may be overestimated by assuming they are present at half of the LOQ, when they are actually not, especially for chemicals with expected no-effect levels even below the LOQ (James et al., 2009, von der Ohe et al. 2011).

Reliable databases covering the spatial extent of the marine ecosystems and time trends are available only for few substances. These are usually substances that are highly regulated, as non-regulated substances (emerging hazardous substances) are generally measured only punctually in time and space. The emerging substances are caught in a “vicious cycle”. Because emerging substances are not usually considered in conventional priorisation methodologies, they are monitored less often and as a result, little data are available to show evidence of risk. Many priorisation excersises e.g. for the WFD, exclude substances with insufficient data base (Daginnus et al., 2010; Götz et al., 2010; James et al., 2009; Klein et al., 1999; Muir and Howard, 2006; von der Ohe et al. 2011, Wilkinson et al., 2007). Thus, especially for emerging hazardous substances, in most cases the PEC has to be derived by approximations, with only few actually measured concentrations for validation.

While there are substances with a sufficient and reliable data set for the freshwater environment, data is even scarcer for the marine environment. For the marine environment, which represents a final sink, the amount of a substance X that reaches the marine environment depends on two aspects. Firstly, on the persistence and the mobility of the substance in the ecosphere (transport and environmental fate processes). Secondly, on the source pattern, technical barriers and existing regulation in the anthroposphere, as laid out in the introduction.

In the light of the uncertainities, this project takes a pragmatic approach, compiling the arguments for inclusion or exclusion of a substance for monitoring under the MSFD from the available data, even if PEC/PNEC ratios are not available. The main criteria that were selected are described below.

Generally, in order to reach the marine ecosystem as final sink, a substance has to be persistent and mobile or emitted in close distance/ offshore. The prime condition is persistence: The more persistent a substance is, the more likely it reaches the ocean as final sink (or other remote areas like the arctic).

Mobility of a substance is also important. From the perspective of the marine, there are three general input pathways for a substance:

- Transport in water (hydrophilic substances) → riverine input as “classical” pathway
- Transport bound to particles (non-hydrophilic / lipophilic substances) → riverine input of hazardous substances bound to particles is less in the focus of current regulation (WFD)
- Transport by air → potential for long range transport LRT.

Non-hydrophilic / lipophilic substances bound to particles, generally speaking, have a lower tendency to be transported over large distances than highly mobile hydrophilic substances in the water phase. But once they reach the marine environment, contaminated sediments act as a secondary source of these hazardous substances. Besides riverine input of particle-bound hazardous substances and erosion from coast (close distance), dredging is another important input pathway for particle bound hazardous substances.

Long-range transport of substances by air also gives reasons for concern, as substances produced in other parts of the world (e.g. China and USA) can be transported to European seas.

Persistence and mobility are substance characteristics determining how much of the emitted load of a hazardous substance reaches the marine ecosystem. But the emission pattern as a function of sources, pathways and existing regulation as shown in Figure 5-2, is also relevant as it determines the load of the substance from the anthroposphere to the ecosphere. To recap, existing regulation acts as a valve in the emission pattern. The status of regulation is a very important driver of the emission pattern.

Regarding their regulatory status, substances can be grouped in highly regulated substances, medium regulated and low regulated substances. For the purpose of this project, "highly regulated" substances describe substances that are included in the Stockholm Convention on POP or subject to an EU ban on the marketing and use. Medium regulated are substances in focus of WFD, OSPAR and HELCOM[23], and low regulated substances are in focus of none of the above, but only of river specific regulations[24].

Looking at the highly regulated substances, the argument can go both ways. One expert can argue that highly regulated substances e.g. listed in the Stockholm Convention on POPor under an EU ban are obviously hazardous and the implementation of regulations has to be confirmed by medium to long term monitoring of the decrease (5->10 years). Thus, "legacy" substances should be included in the monitoring process.

Another expert can argue that highly regulated substances can be expected to decrease in the environment and can be excluded from routine monitoring in the marine environment. Instead, some sort of "fade out monitoring" should be executed. Finally, if a substance is already highly regulated, monitoring results cannot lead to any regulatory action. Violation of the regulation or "loopholes"[25] can be better detected on watershed or facility level or in wastewater effluents. Instead of legacy hazardous substances, less regulated substances or emerging hazardous substances should be monitored, as regulatory action can be taken for these substances.

The MSFD requires monitoring of the substances that threatens GES – this can well be legacy hazardous substances in line with expert 1– but also focusses on measures that have to be taken when the GES is endangered – in line with expert 2. As both perspectives are valid, this report includes different sets of substances (chapter 5.1.8).

Another perspective on the regulatory status is the freshwater focus vs. the marine focus (North and Baltic Sea). Looking at the substances in focus of OSPAR, HELCOM, WFD, and river-specific regulations, there is some overlap. Some substances are in focus of all three - OSPAR, HELCOM, WFD – denoted with category "**A"** for the purpose of this project. Some substances are observed in the maritime sector (OSPAR, HELCOM) but not in the WFD "**B"**. Some substances are observed in the WFD but not in the maritime sector "**C"**. A fourth group of substances appears exclusively in river-specific inventories "**D**".

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23 For the medium regulated substances, the regulation by WFD means that measures have to be taken if the EQS is violated in EU Water

24 For the low regulated substances in focus of river specific regulations (regulated in national context, not on EU-level)

25 Violation e.g. import with carpets treated with PFOS, „loophole“ exemptions from EU ban

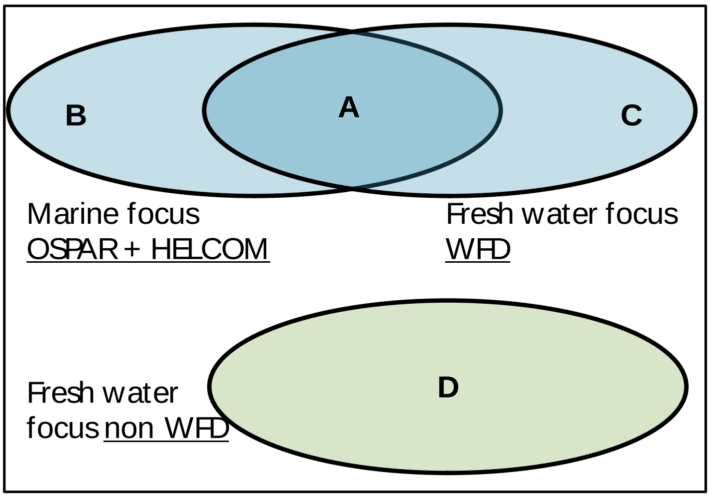

Figure 5-5: Overlaps of relevant substance inventories

Other important criteria that are compiled in the fact sheets and summarized in Figure 5-9 are

- Criterion 3a: High Bioaccumulation / Biomagnification Potential e.g. Hg, PFOS
- Criterion 3b: Toxic substances or endocrine disruptors
- Criterion 3c: Potential for long-range transport and atmospheric deposition, substances listed in Convention on Long-Range Transboundary Air Pollution (CLRTAP) hazardous substances: e.g. Hg, PFOS
- Criterion 3d: Substances with contaminated sediments as secondary source e.g. TBT, Hg
- Criterion 3e: Substances with very large production volumes e.g. TBBP-A
- Criterion 3f: Substances with very diffuse (partly non-regulated) sources e.g. flame retardants in products and building material (TBBP-A, octaBDE)

Criteria 3a-f are no stand-alone criteria leading to an inclusion or exclusion of a substance. Instead all criteria have to be viewed together (synopsis) to find the ”pros“ and ”cons“ for each substance. In addition, the regulatory status of substances is documented (Criteria 4a-c).

- Criterion 4a: Regulatory status of the substances: Legacy hazardous substances or medium to low regulated substances, freshwater or marine focus
- Criteria 4b: Recommended for monitoring by BLMP[26]
- Criteria 4c: Recommended for monitoring under MSFD D9 (seafood)

The methodological approach used in this project is described in detail in chapter 5.1.6. The results are reported in Chapter 5.1.7 and the conclusions are presented in Chapter 5.1.8.

The following subchapters give an overview of existing concepts for the identification and priorization of relevant substances and existing monitoring concepts (Chapter 5.1.4 and 5.1.5).

5.1.4 Existing methods for the identification and priorization of relevant substances

Von der Ohe et al. 2011 describes the different priorisation excercises with focus on freshwater (von der Ohe et al. 2011). The Combined Monitoring based and Modeling-based Priority setting (COMMPS) procedure was

[26] Preliminary results of the working group

the first European wide prioritization exercise and resulted in the current list of PS (Klein et al., 1999). In that study, emphasis was given to the availability of complete exposure and hazard information, reducing the list of evaluated substances to only 279, disregarding potentially problematic substances with limited data sets. The analytical techniques and the limits of quantification available at that time further limited the number of possible detections (Klein et al., 1999).

A similar approach was applied in the prioritization study carried out for the revision of the first list of PS. According to Article 16 of the WFD, the list of PS needs to be reviewed every four years. The revision of the first list of PS involved two distinct prioritization procedures: a monitoring-based prioritization study conducted by L'Institut National de l'EnviRonnement Industriel et des RiSques (INERIS) (James et al. 2009, von der Ohe et al. 2011), and a modeling-based prioritization study conducted by JRC (Daginnus et al., 2010).

The modeling-based prioritization exercise again evaluated a total of 2034 compounds according to pre-defined hazard and exposure criteria that yielded 78 substances of potential high concern, for which a more intensive assessment was performed (Daginnus et al., 2010). The modeling-based approach used a risk scoring that ranged from 1 to 5, which therefore did not allow for a quantitative assessment based on PEC/PNEC ratios. It was adapted from another study, performed in the UK (Wilkinson et al., 2007), which also depended on the integration of hazard and exposure predictions and aimed to develop a robust and transparent methodology for identifying and prioritizing Annex VIII chemicals in the UK, referring to Specific Pollutants.

In Switzerland, an exposure-based methodology was developed to rank microcontaminants for monitoring according to their potential occurrence in surface waters (Götz et al., 2010). The methodology was based on the chemicals' distribution behavior between different environmental media, degradation data and input dynamics, while the hazard aspect was ignored. Although this allows for prioritizing chemicals for monitoring based on their potential presence in the environment, chemicals with high toxicity but rather low exposure levels might be overlooked. Similarly, a study by Muir and Howard (2006) focused their prioritization on the persistence, bioaccumulation and the long-range transport potential of substances, disregarding their potential toxicity.

A study by von der Ohe et al. (2011) classifies chemicals into six categories, based on the quality and quantity of the available information. Depending on the outcome of the classification, major actions to be taken by water managers were suggested (e.g. derivation of EQS, improvement of analytical methods, etc.). The study assesses the risk of 500 organic chemicals based on analytical observations. The dataset comprised the four river basins of Danube, Elbe, Scheldt and Llobregat.

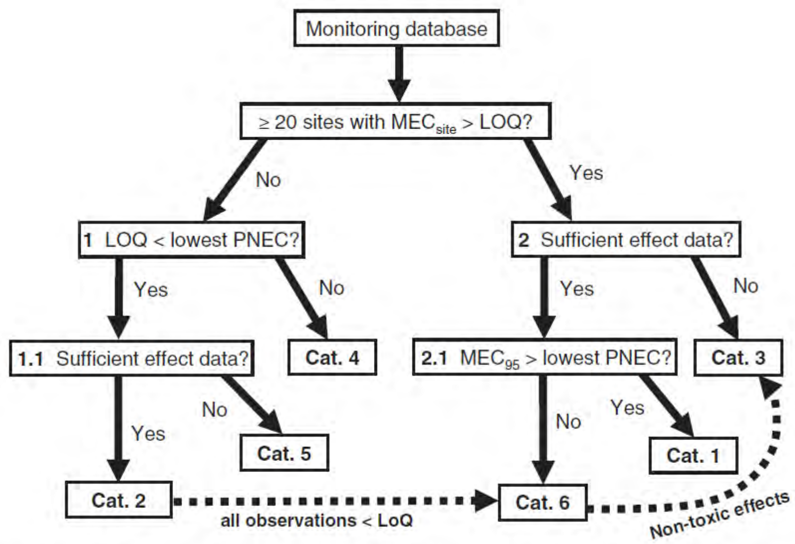

Fig. 4. Decision tree to classify emerging and classic organic pollutants into six categories of defined actions. For explanation of the categories, please see text. (MEC_{site} = maximum environmental concentration at each site in recent years, MEC_{95} = 95th percentile of all MEC_{site}, Cat. = Category, LOQ = Limit of quantification).

Figure 5-6: Decision tree that classifies chemicals into six categories according to available data (from von der Ohe et al. 2011).

In a first step, compounds are separated according to the availability of sufficient exposure data (i.e. more than 20 sites with analytical measurements above the LOQ).

Two groups are created, which differ with regard to evidence of exposure: Group 1 lacks clear evidence, while Group 2 consists of chemicals with enough evidence. Group 2 is then further separated by the availability of sufficient effect data for EQS derivation.

Those compounds that do not comply with this requirement form Category 3: i.e. the available monitoring data (MEC95) for these compounds show evidence of exposure. However, hazard assessment is based on predicted toxicity (P-PNEC) only and hence, a rigorous effect assessment is recommended in order to derive a legally binding EQS for compounds of high priority.

The remainder of Group 2.1 was further split based on the evidence of risk, i.e. by means of the hazard quotient of the exposure level (MEC95) and the effect level (lowest PNEC).

Compounds with MEC95/PNEC ratios above one would trigger the substance's classification in Category 1: these compounds should be included in the list of river basin specific pollutants according to Annex VIII of the WFD.

The remaining chemicals show evidence that the exposure does not pose harm to ecosystem and human health at the observed concentrations and form Category 6. For these chemicals, monitoring efforts could be reduced, if no other non-toxic effects (e.g. endocrine disruption) are expected, in which case the compound should be reclassified to Category 3.

Group 1 again was further split based on the availability of appropriate analytical methods: those compounds for which the maximum LOQ in the dataset exceeds the lowest PNEC, comprise Category 4. For these chemicals, analytical methods should be improved to assess the real risk of the substance.

The group of compounds for which analytical performance can be considered satisfactory (1.1), was further split into two groups based on the availability of sufficient effect data for the derivation of EQS.

Compounds for which there is sufficient effect data (i.e. experimental data available for the three trophic levels) but only a few observations above the LOQ, comprise Category 2: for these compounds, a screening study should be performed to inform about the current exposure situation. However, it is also possible to find compounds in this category which are already well investigated but rarely quantified: if a compound is already well investigated (i.e. more than 10,000 observations) and the data indicated no risk for the environment (i.e. all entries below the LOQ and LOQ below the lowest PNEC), this should be reclassified into Category 6.

The remaining substances in Group 1.1 have no or few observations in the environment and there is no hard evidence on potential effects (since hazard evaluation can only be based on predicted data due to lack of experimental data). They comprise Category 5: for these compounds, both a screening study and a rigorous effect assessment would be required before final conclusions could be drawn (von der Ohe et al. 2011).

Category 1, for which inclusion in the monitoring program is recommaded, includes a total of 73 compounds. These substances are ranked according to the frequency of exceedance and the extent of exceedance. The first indicator (frequency of exceedance) considers the spatial distribution of potential effects of a certain compound, i.e. the frequency of sites with observations above a certain effect threshold. For the calculation of this indicator, a compound's maximum observed concentration at each site (MECsite) is compared to the lowest PNEC.

The second indicator (extent of exceedance) ranks compounds with regard to the extent of the expected effects. While the previous indicator considers that some compounds might be widely distributed, it may overlook that some of these chemicals occur only in rather low concentrations close to their effect threshold. These compounds might be still of concern, but with regard to local impacts (i.e. effects on the ecological status), other compounds might be much more relevant. In this way, compounds that have a somewhat narrower spatial distribution might reveal their “local importance”. Therefore, also the ratio of the 95th percentile of all MECsite values per compound (MEC95) is calculated.

The compound with the highest frequency (88%) is the pesticide diazinon, followed by terbutylazine (64%), as well as the priority substances DEHP (56%) and endosulfan I (51%). Interestingly, 18 of the 33 PS (i.e. 24 individual substances) were classified into category 1, as well as four of the eight PHS.

In Schluep et al. (2006), describes different methods applied to the prioritization of substances:

- COMMPS – Process used by the EU to select priority substances.
- DYNAMEC: OSPAR[27]-process for the selection and prioritization of hazardous substances (described in Poremski and Wiandt (2002))
- CCL – Contaminated Candidate List (US-EPA)

The steps of the EU’s COMMPS process (EC, 2001) are summarized by Schluep et al. (2006) as follows:

[27] OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic

• Step A	Selection of substances as candidates for the evaluation process. The substances are selected from different official substance lists and monitoring programs ("lists-based approach").
• Step B	Ranking based on exposure data
• Step C	Ranking based on effect data
• Step D	Calculation of risk The risk of a substance is calculated by multiplying the listing status of exposure data by the listing status of effect data.
• Step E	Recommendation of priority substances The priority substances are chosen based on the risk list. A two-step procedure is applied: In the first step, the lists are examined and a candidate group of priority substances is selected. In the second step, recommendations are made about to include or exclude these candidates from the list of priority substances.

Figure 5-7 shows the OSPAR-process for the identification and prioritization of hazardous substances.

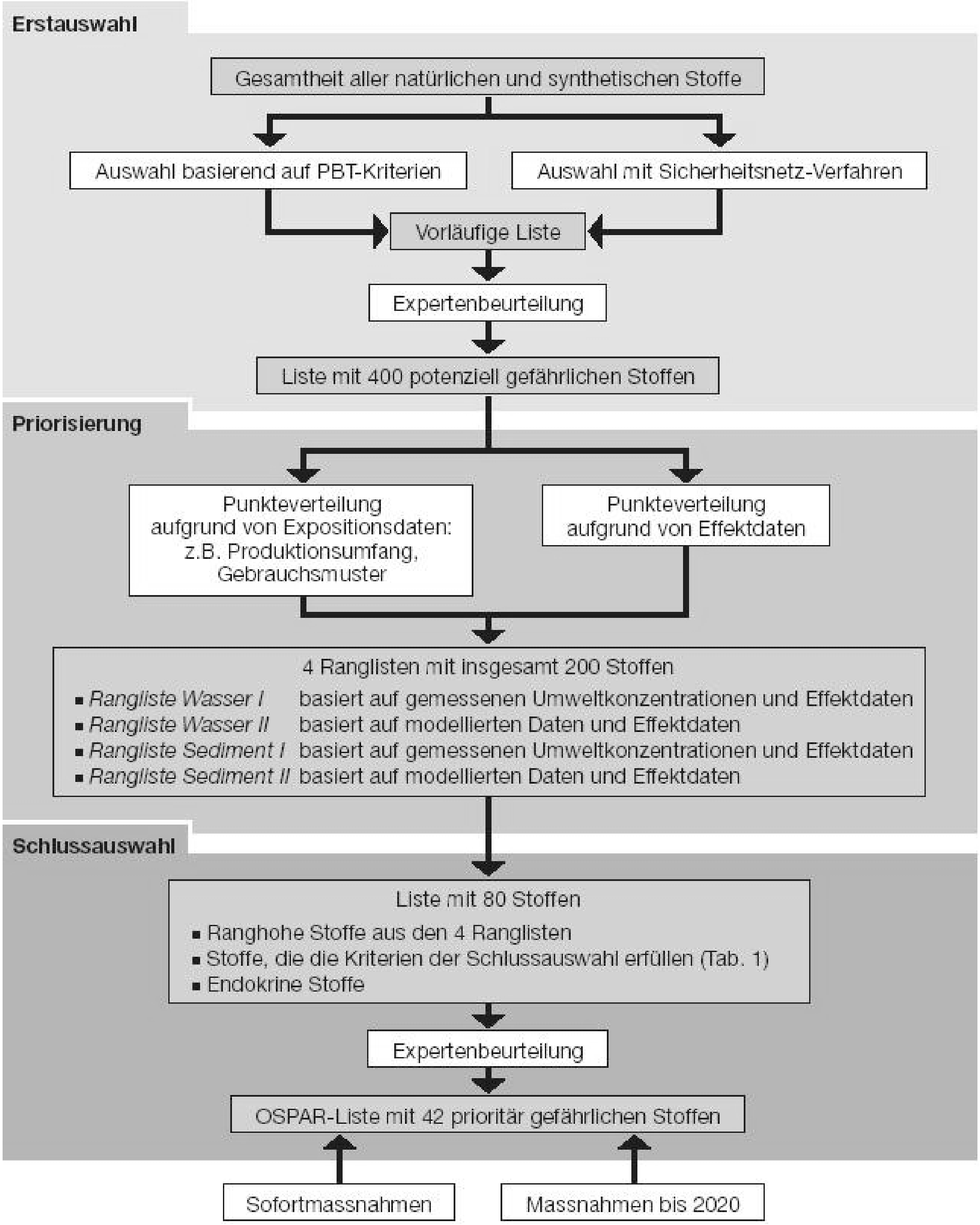

Figure 5-7: OSPAR-process for the identification and prioritization of hazardous substances (Poremski and Wiandt, 2002 in Schluep et al. 2006).

Schleup et al. (2006) describe the common ground of these methods as follows: The total number of substances is restricted to those in existing lists. To further limit them, this is followed by a first evaluation based on substance data and a second evaluation based on risk assessment including expert knowledge.

The COMMPS-method focuses on existing priority lists, whereas the DYNAMEC-method also contains a safety net procedure. DYNAMEC is based on existing European chemical databases and a selection using PBT-criteria (persistent, bioaccumulative, toxic). The safety net procedure further ensures that relevant substances are also included which neither fulfill the PCB-criteria, nor are listed on existing chemical databases. Such substances are reported by third parties to the OSPAR expert committee in the form of a demand for inspection and are then also run through the prioritization method (Schluep et al., 2006).

5.1.5 Existing Monitoring and Assessment concepts (OSPAR and HELCOM)

This chapter reviews Existing Monitoring and Assessment concepts with focus on regional Convention important for the North and Baltic Sea.

OSPAR

As part of the OSPAR Convention the "Joint Assessment and Monitoring Programme 2010-2014 – JAMP" was outlined (OSPAR Agreement 2010-4, amended 2011)[28]. The agreement sets out how OSPAR implements the suite of existing and agreed monitoring strategies. Contracting parties will continue to monitor as part of:

- CAMP - Comprehensive Atmospheric Monitoring Programme

  CAMP includes the JAMP guidelines for sampling and analyzing of **mercury** in air and precipitation, the JAMP guidelines on methods and criteria for harmonized sampling and analyzing of **PAHs** in air and precipitation and the guidance note on sampling and analyzing of **PCBs** in air and precipitation

- CEMP - Coordinated Environmental Monitoring Programme (Agreement 2009-1)

  According to OSPAR[29] the CEMP is currently focussed on monitoring

  - metals (cadmium, mercury and lead) in sediment and biota
  - PAHs in biota and sediment
  - PCBs in biota and sediment
  - brominated flame retardants in biota and sediment
  - the effects of tributyltin in gastropods and concentrations in sediment and/or biota
  - nutrients in sea water
  - eutrophication effects.

- RID - Riverine Inputs and Direct Discharges

  The monitoring commitments are set out in the RID principles (Recerence number 1998-5). Mandatory are:

  - Total Mercury (Hg)
  - Total Cadmium (Cd)
  - Total Copper (Cu)

[28] http://www.ospar.org/v_measures/browse.asp?menu=01290301790125_000002_000000

[29] http://www.ospar.org/content/content.asp?menu=00900301400000_000000_000000 (accessed on 01.11.2013)

- Total Zinc (Zn)
- Total Lead (Pb)
- Gamma-HCH (lindane)
- Ammonia expressed as N
- Nitrates expressed as N
- Orthophosphates expressed as P
- Total N
- Total P
- Suspended particulate matter (SPM)
- Salinity (in saline waters).

Contracting Parties will continue, at their discretion, to monitor the following substances as voluntary determinands under the RID:

- polychlorinated biphenyls (the congeners: IUPAC Nos 28, 52, 101, 118, 153, 138, 180);
- polycyclic aromatic hydrocarbons.
- Other hazardous substances (particularly organohalogen compounds - in order to determine which organohalogen compounds should be included in future input studies).

Data on contaminants collected on the CEMP are assessed using methodologies set out in the CEMP Assessment Manual[30] (OSPAR 2008). In order to create a comparibility of data the CEMP Assessment Manual applies a common set of base data which expresses the concentrations. OSPAR (2008) proposes that concentrations in the sediment are expressed on a dry weight basis. Where available the concentrations should be normalized with respect to the content of aluminium (for metals) or total organic carbon (TOC) (for organic contaminants), using the following conversions: Sediment/metals (Pb, Cd, Hg): dw to 5% Al, and Sediment/POPs (PCB/PAH): dw to 2.5% TOC.

For biota the following bases are preferred:

- dry weights for metals, organometals, organochlorines and PAHs in bivalve soft body tissues;
- wet weights for metals and organochlorines in fish muscle, including tail muscle (crustaceans), and
- metals in fish liver; and
- lipid weights for organochlorines in fish liver.

<u>HELCOM[31]</u>

The HELCOM Monitoring and Assessment activities are set out in the Monitoring and Assessment strategy that includes

[30] http://www.ospar.org/content/content.asp?menu=00900301400000_000000_000000 link: “CEMP Assessment Manual”

[31] HELCOM MONITORING AND ASSESSMENT STRATEGY (www.helcom.fi)

- the Joint Coordinated Monitoring System with the HELCOM Data and Information Strategy and
- the HELCOM Assessment Strategy

Within the Joint Coordinated Monitoring System, the national Monitoring data are collected in a data pool and can be provided to all member countries.

With respect to quality and comparability it is recommended that sampling and analyses should be carried out using certified methods and laboratories.

The data pool can be used as basis for the assessment strategy. Different Assessment products (Core indicator reports, Baltic Sea Environment Fact Sheets, different thematic assessment products, holistic assessments) are reported to the member countries and the EU.

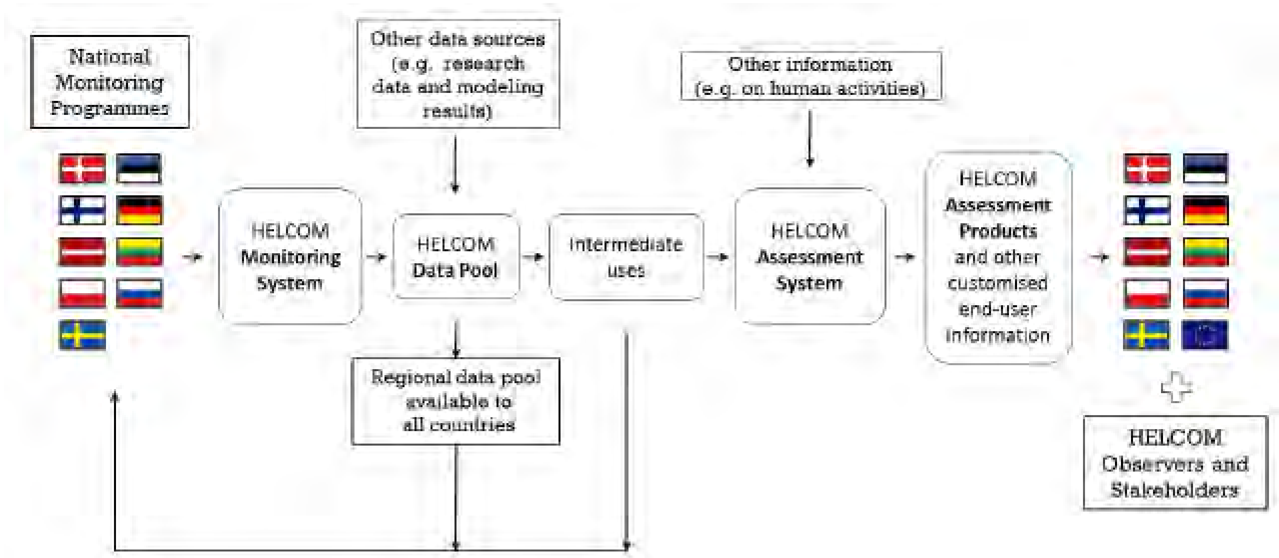

Figure 5-8: Flow-chart of the HELCOM Monitoring and Assessment System described in the HELCOM Monitoring and Assessment Strategy (http://helcom.fi/action-areas/monitoring-and-assessment)

<u>HELCOM CHASE 1.0</u>

The HELCOM strategic goals and ecological objectives for hazardous substances are described as follows:

Strategic Goal:

- Baltic Sea life undisturbed by hazardous substances

Ecological objectives:

- Concentrations of hazardous substances close to natural levels
- All fish safe to eat
- Healthy wildlife
- Radioactivity (radionuclides) at pre-Chernobyl-level

Against this background, the HELCOM instrument for the holistic Assessment of Ecosystem Health status was developed (HOLAS). It contains the following three tools

HEAT – HELCOM Eutrophication Assessment Tool

BEAT – HELCOM Biodiversity Assessment Tool

CHASE – Hazardous Substances Status Assessment Tool

Within the CHASE-Tool the Contamination Ratio CR of one substance is defined as the ratio of the observed value in relation to the threshold conditions.

$$CR = \frac{observed\ value}{Threshold\ value}$$

This represents the PEC/PNEC approach. Beyond this, the CRs of all substances or indicators are calculated for four groups: sediment, blue mussel, fish and VDSI[32], and are integrated by taking the sum and dividing by the squareroot of the count for each matrix. This integrated value will yield a status classification (<0.5 "high", 0.5-1 "good", 1-5 "moderate", 5-10 "poor" or >10 "bad") (see Table 5-1).

Table 5-1: Envrionmental status according to CHASE[33]

Contamination Ratio	Status	Disturbance	Classification according to (MSFD)[34]
0,00			
	High	None or very minor deviation from undisturbed conditions	Acceptable Status
	Good	Slight deviation	
1,00			
	Moderate	Moderate deviation	Impaired Status
	Poor	Major deviation	
	Bad	Severe deviation	
∞			

The target values are defined by the ecological objectives "Concentrations close to natural levels", "all fish safe to eat", "healthy wildlife", "radioactivity at a pre-chernobyl-level".

In the following this mathematical approach is interpreted in terms of its effect:

The effect of this calculation mode is that through the devision by the squareroot of the amount of values the quotient becomes larger. The greater the number of values, the stronger the effect of the square root.

The arithmetic mean would relate the measured concentrations to the absolute numbers of substances and thus neglect that there is a large number of individual substances that could have a cumulative effect in their sum.

[32] VDSI: Vas Deferens Sequence Index is used to monitor levels of TBT in marine environment (measure of imposex)

[33] Source: Minna Pyhälä, EMODNET chemical Data Products Experst workshop, Sep. 2010, Venice, Italy, Andersen 2015 http://waters.gu.se/digitalAssets/1531/1531488_andersen_holas_waters-symposium_v1.pdf

[34] Source: Minna Pyhälä, EMODNET chemical Data Products Experst workshop, Sep. 2010, Venice, Italy, Andersen 2015 http://waters.gu.se/digitalAssets/1531/1531488_andersen_holas_waters-symposium_v1.pdf

The weighting of the sum by the squareroot of the number of summands in contrast, takes into account that with an increasing number of individual concentrations the denominator is reduced (square root of the number of summands) and the quotient = CR becomes larger. This way, the cumulative effects are taken into account.

If a large number of hazardous substances is found, which may even be present in low concentrations, the described calculation mode in the CHASE tool takes the cumulative effects into account.

Example: The substances A, B, C, D are monitored under the target "Concentrations close to natural levels". The following CRs are calculated:

$CR_A = 0.2$

$CR_B = 1.5$

$CR_C = 0.1$

$CR_D = 1.8$

In this case the arithmethic mean would lead to (0.2+1.5+0.1+1.8)/4=0.9. According to the definition of CR, the environmental status would be acceptable. The cumulative effect is not regarded.

The CHASE calculation mode leads to (0.2+1.5+0.1+1.8)/√4=1.8. Here, the cumulative effect is regarded in the sense that the result exceeds the target value <1, if the nominator exceeds the squareroot of the number of substances (denominator).

Hence, the more hazardous substances are detected the lower their single CRs must be in order to reach an acceptable environmental status. This mathematical approach therefore supports the consideration of the cumulative effects and, in this sense, seems to be a suitable approach for the application also in national context.

The Harmony Project – CHASE 2.0

Within the Harmony Project[35] the CHASE-Tool was applied to the greater North Sea (Andersen et al. 2010 *draft*). The described CHASE 1.0 was improved to CHASE 2.0. The key difference is the grouping of substances. While in CHASE 1.0 the substances are grouped according to the four targets "Concentrations close to natural levels", "all fish safe to eat", "healthy wildlife", "radioactivity at a pre-chernobyl-level", in CHASE 2.0 the following headings are used: Water, Sediments, Biota, Biological effects.

In CHASE 2.0 the data was derived from quality checked monitoring data submitted to international databases like EIONET and ICES. Beyond this the data was aggregated into so-called assessment units (Andersen et al. 2010 *draft*).

Concerning assessment, the Harmony Project refers to the CEMP Assessment Manual (OSPAR 2008). Concerning the target values, it refers to OSPAR EAC and the EPA threshold "ERL" (Effects Range – Low).

5.1.6 Methodological Approach

Before the background of the methodologies described above, this chapter describes the steps taken in this project towards the recommendation of substance sets. These are the following:

[35] http://harmony.dmu.dk.

- Compile criteria
- Compile candidate list (long list) from different substance inventories
- Clustering of substances to reduce the complexity
- Compile fact sheets using the criteria, report arguments for inclusion and exclusion of a substance
- Derive recommendations: different sets of substances (short list).

5.1.6.1 Compilation of criteria

The following criteria are reported in the fact sheets for the substances and substance groups (see

Figure 5-9).

Highest priority criteria

- Criterion 1: High PEC to PNEC ratio

The PEC/PNEC-ration is the prime criterion for assessment of the relevance of a substance. The EQS, which are defined in the WFD can be interpreted as the PNEC for coastal waters but not for the open sea. Reliable databases covering the spatial extent of the marine ecosystems and time trends are only for few substances available and those usually are already highly regulated. At long last the database for the PEC/PNEC ratio is very low and therefore this criterion is not really applicable.

The reasons for the here chosen pragmatic approach beyond the PEC/PNEC ratio are described extensively in chapter 5.1.3.

If reliable values are not available, the second priority criteria and the additional criteria need to be applied.

Second priority

- Criterion 2a: Persistence
- Criterion 2b: Mobility
- Persistence and mobility are both necessary conditions for a substance to reach the marine environment. (Criteria 2a and b) need to be fulfilled to denote a substance “relevant for the marine environment”. If not fulfilled, then (Criterion 2c) or (Criterion 2d) need to be fulfilled to denote a substance “relevant for the marine environment”.
- Criterion 2c: Substances with marine applications directly discharged to sea e.g. TBT (historic), PAH (oil)
- Criterion 2d: Substances that are regularly detected in samples from the marine environment (biota, water, sediments), even if they not fulfill (Criteria 2a and b)

Additional important criteria that are compiled in the factsheets are:

- Criterion 3a: Potential for long range transport and atmospheric deposition, substances listed in Convention on Long-Range Transboundary Air Pollution CLRTAP: e.g. Hg, PFOS
- Criterion 3b: High Bioaccumulation / Biomagnification Potential e.g. Hg, PFOS
- Criterion 3c: Substances with potential for endocrine disruption (Indications from scientific discourse and literature)
- Criterion 3d: Substances with contaminated sediments as secondary sourc e.g. TBT, Hg
- Criterion 3e: Substances with very diffuse (non-regulated) sources e.g. flame retardants in products and building material (TBBP-A, octaBDE).
- Criterion 3f: Substances with very large production volumes e.g. TBBP-A

For (Criteria 3a-f), the assessment cannot rely on one single criteria, but synopsis of all these criteria is required. These criteria are compiled as they are regarded as additional reasons for inclusion or exclusion of a substance in the framework of this project.

In addition, the regulatory status of a substance is reported, and whether it has already been recommended for monitoring in marine waters.

- Criterion 4a: Regulatory status of the substances: Legacy hazardous substances or medium to low regulated substances, freshwater or marine focus)
- Criteria 4b: Candidate for national monitoring under MSFD D8 (BLMP[36])
- Criteria 4c: Recommended for monitoring under MSFD D9 (seafood)

The following substances are suggested to be monitored under Descriptor 9 (JRC 2010)[37]

- Heavy metals (Pb, Cd, Hg)
- PAH
- Dioxins Dx (including dioxin-like PCBs)
- Radionuclides

Second priority:

- Arsenic
- Non-dioxin like PCBs
- Phthalates
- Organochlorine pesticides
- Organotin compounds
- Brominated flame retardants
- Polyfluorinated compounds.

[36] Preliminary results of the working group

[37] JRC 2010: Task Group 9 – Contaminants in fish and other seafood

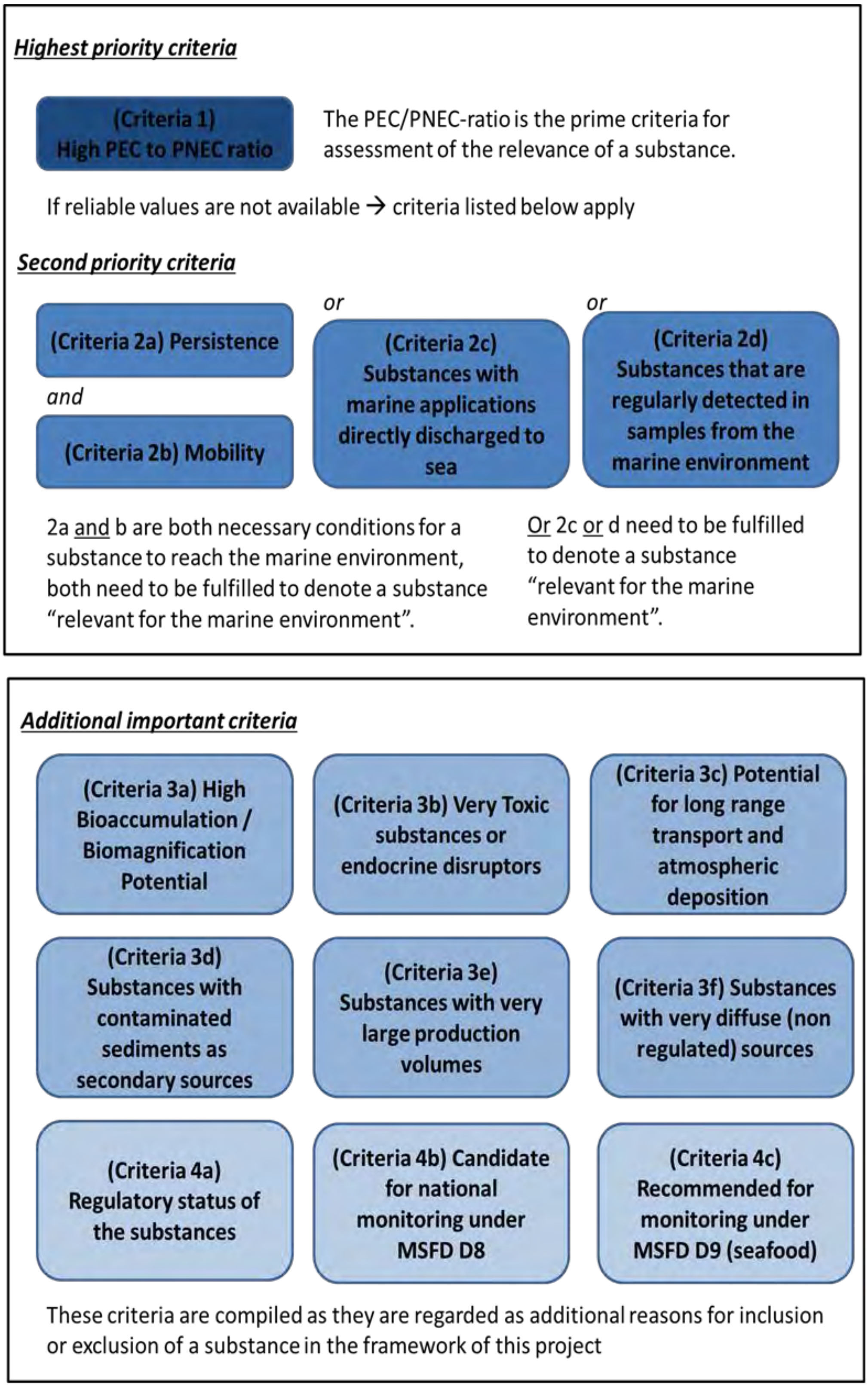

Figure 5-9: Applied criteria.

5.1.6.2 Compilation of substance inventories

All the relevant substance inventories were summarized in one table and consolidated based on names and CAS-numbers (Chemical Abstracts Service). The following inventories have been identified as relevant for the MSFD:

- Water Framework Directive (WFD) Priority Substances

  The WFD is a European Union Directive, which creates a regulating framework for all water bodies in all member states of the EU. It identifies 13 dangerous priority substances and 20 priority substances. (Directive 2000/60/EC)[38]

  Since 12th of August 2013 the new Directive 2013/39/EU contains the relevant list of priority and other substances (45 substances), amending Directives 2000/60/EC (WFD) and 2008/105/EC (environmental quality standards)

- Oberflächengewässerverordnung (OGewV) - Surface Waters Ordinance

  The OGewV is the implementation of the Directive 2000/60/EC (WFD) under German law. Its objective is the protection of surface waters[39].

- OgewV Candidate Substances (where available from public debate)

  Candidate substances, whose integration in the OGewV as river specific substances is being considered. The listed substances are only suggestions[40].

- HELCOM Baltic Sea Action Plan (BSAP)

  The BSAP is a HELCOM program to restore the good ecological status of the Baltic marine environment by 2021 (HELCOM, 2012a and b)[41].

- HELCOM CORESET (incl. candidate and supplementary substances)

  The HELCOM CORESET program aims to develop a set of HELCOM core indicators for biodiversity and hazardous substances with quantitative targets to allow an assessment of the status of the Baltic Sea in relation to the ecological objectives for hazardous substances. The list of the hazardous substances can be found in HELCOM, 2012a and b[42]. The final set of core indicators is listed in HELCOM (2013).

- OSPAR List of Chemicals for Priority Action (Revised 2011)

  The OSPAR Convention is the current legal instrument guiding international cooperation on the protection of the marine environment of the North-East Atlantic. The priority chemicals listed were identified at different OSPAR Commission Meetings between the years 1998-2003 (OSPAR, 2004)[43].

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[38] http://www.bmu.de/binnengewaesser/gewaesserschutzrecht/europa/doc/38010.php

[39] http://www.gesetze-im-internet.de/bundesrecht/ogewv/gesamt.pdf.

[40] e.g. http://www.umweltbundesamt.de/chemikalien/veranstaltungen/ws-monitoring-arzneimittel/3_praesentation_vietoris.pdf

[41] http://www.helcom.fi/BShazardous substances

[42] http://www.helcom.fi/projects/on_going/en_GB/coreset

[43] http://www.ospar.org

- OSPAR MIME

  The working group on Monitoring and on Trends and Effects of Substances in the Marine Envrionment (MIME) layed down a set of Potential OSPAR Common Indicators and Candidate Indicators (OSPAR 2012)

- ICPR – International Commission for the Protection of the Rhine

  The ICPR is an international Commission targeting sustainable development and good environmental status of the Rhine and its catchment basin (IKSR, 2011)[44].

- Stockholm Convention on POP (persistent organic pollutants)

  The Stockholm Convention on Persistent Organic Pollutants is a global treaty to protect human health and the environment from chemicals that remain intact in the environment for long periods, become widely distributed geographically, accumulate in the fatty tissue of humans and wildlife, and have adverse effects on human health or the environment[45][46].

In order to complete the picture concerning management status and regulation, the list was compared with following lists (”others“):

- REACH svhc ”svhc“

  1. Under EU REACH regulation, substances that are one of the following can be regarded as substance of very high concern (SVHC):

     - carcinogenic, mutagenic or toxic to reproduction (CMRs);
     - persistent, bio-accumulative and toxic (PBTs);
     - very persistent and bio-accumulative (vPvBs);
     - seriously and / or irreversibly damaging the environment or human health, as substances damaging the hormone system;

  2. The competent authority or agency of a Member State can suggest the inclusion of a substance with above properties on SVHC candidate list by preparing a dossier. Interested parties are then invited to comment on the substance for which a dossier has been prepared. The resulting outcome of this identification process is the creation of a list of identified substances, which are then deemed candidates for authorization. (“SVHC candidate list”).

  3. Some substances from the candidate list will be prioritized for authorization and be included in Annex XIV ("**SVHC authorization list**"). Those substances on authorization list will not be allowed to be used, placed on the market or imported into the EU after a date to be set unless the company is granted an Authorization.

- SIN list (”substitute it now“) http://www.chemsec.org/what-we-do/sin-list

[44] http://www.iksr.org

[45] http://chm.pops.int/Convention/tabid/54/Default.aspx

[46] http://chm.pops.int/default.aspx

1. The SIN List consists of chemicals that have been identified by ChemSec as being Substances of Very High Concern, based on the criteria for these defined within REACH. The SIN List aims to speed up the REACH process

- UBA-Projekt: “Revision der Umweltqualitätsnormen der Bundes-Oberflächengewässerverordnung nach Ende der Übergangsfrist für Richtlinie 2006/11/EG und Fortschreibung der europäischen Umweltqualitätsziele für prioritäre Stoffe“[47] (Fraunhofer IME) “IME“
- UBA-Projekt: “Maßnahmen zur Verminderung des Eintrages von Mikroschadstoffen in die Gewässer“ (Fraunhofer ISI)[48] “ISI“
- Convention on long-range Transport and Air Pollution CLRTAP[49] ”LRT“
- Internationale Kommissionen zum Schutz von Rhein und Elbe (IKSR / IKSE)

The resulting long list of hazardous substances forms the basis for further considerations and results. Where available, EQS etc also were compiled in a long list of substances. Thus, this tool is also valuable for further analysis beyond this project.

5.1.6.3 Clustering of substances

In order to reduce complexity, the substance list is clustered in functional groups. Ideally groups reflect similar use patterns and emission pathways. The groups are based on function rather than chemical structure (see Figure 5-10).

[47] https://www.umweltbundesamt.de/publikationen/revision-der-umweltqualitaetsnormen-der-bundes; UBA Texte 47/2015

[48] https://www.umweltbundesamt.de/en/publikationen/massnahmen-zur-verminderung-des-eintrages-von; UBA Texte 85/2014

[49] http://www.unece.org/env/lrtap

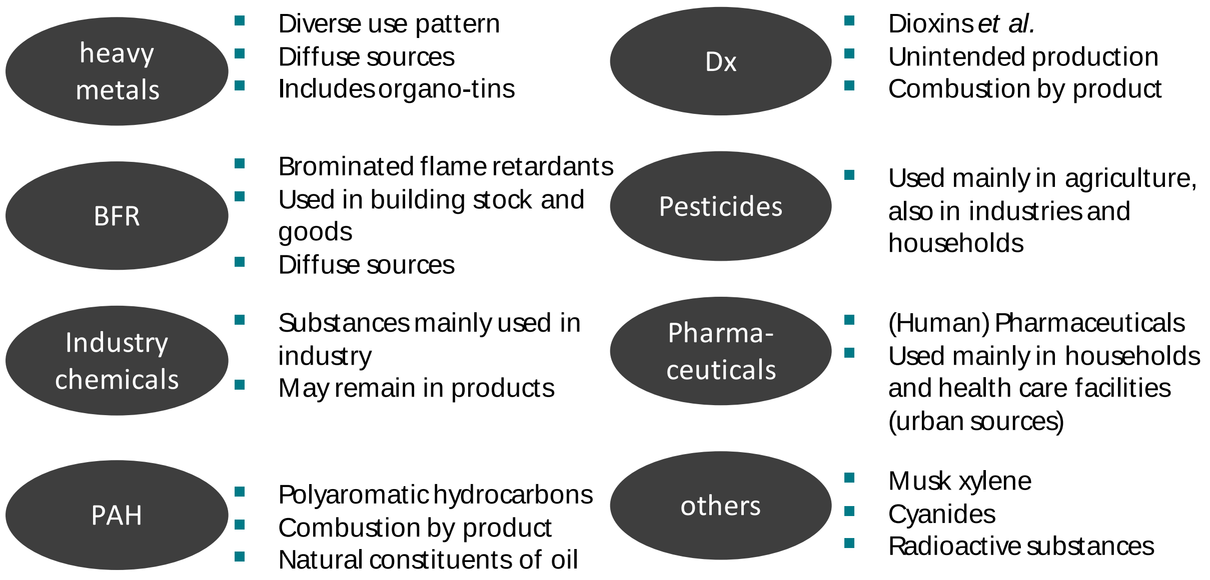


Figure 5-10: Functional groups[50].

In Figure 5-11 the relevant industry chemicals are summarized.

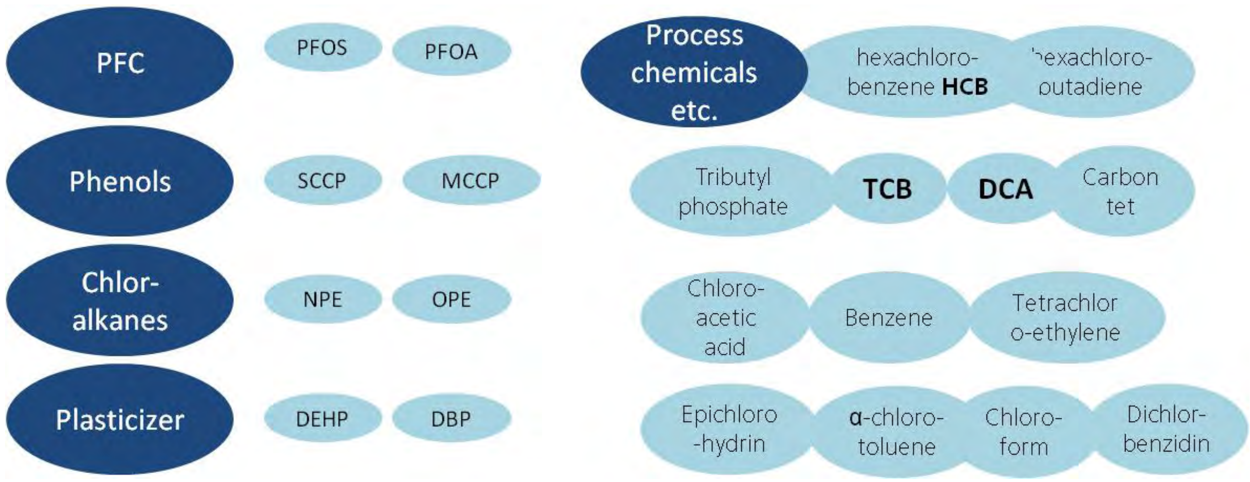


Figure 5-11: Industry chemicals.

5.1.7 Priorization of substances (results)

In the following the results of the two methodological approaches of grouping and priorizing the hazardous substances are documented.

[50] pesticides include both biocides and plant protection products

5.1.7.1 Clustering of substances: regulatory status

The grouping of substances according to their regulation status refers to the idea that the substances listed in the WFD have to be monitored at least in coastal waters. Coastal water is the overlap between WFD and MSFD. It is obvious that the substances which are not yet in focus of HELCOM and OSPAR, but listed in WFD must be part of the WFD monitoring at least in coastal waters.

The following tables summarize the results of the review of the overlaps of the different lists according to Figure 5-5. Beyond this, the tables contain the environmental assessment criteria of the lists as well as background concentration as far as known.

Table 5-2: Hazardous substances, Categorie "A" – listed in OSPAR and/or HELCOM as well as in the WFD

Stoff-Inventar	CAS-Nr.	Stoffname							BC water	BC/LC sed.
Helcom CORESET				UQN marin [µg/l]	UQN benth [µg/kg]				(Quelle: OSAPR CEMP)	
OgewV	UQN oberird. Gew einschl. Übergangsgew. sowie Küstengew nach § 7 Abs 5 Satz 2 WHG - *Quelle:* OGewV 2011 Anlage 5 (zu § 2 Nummer 6, § 5 Absatz 4 Satz 2 und 3, § 9 Absatz 2			water [µg/l]	susp o sed [mg/kg]					
OSPAR ICG EAC	OSPAR EAC Rep. Belgium, based on CEMP; Mus=Mussels; Oy=Oysters; BAC=Background assessment concentration; ERL=Effects Range low; EAC= Environmental assessment criteria			BAC Sed. [µg/kg]	ERL/EAC Sed. [µg/kg]	BAC Mussels / oysters [µg/kg]	ERL/EAC Mussels / oysters [µg/kg]		BC = backgr.conc. LC=low con.	
WFD Prio. Stoffe	AA-annual average; MAC-maximum allowed concentration			AA EQS Inland surface waters [µg/l]	AA EQS other surface waters [µg/l]	MAC EQS Inland surface waters [µg/l]	MAC EQS other surface waters [µg/l]	EQS biota [µg/kg] wet weight	[µg/l]	[µg/kg dry weight]
HELCOM BSAP 10	104-40-5 und 2515	NP Nonylphenols	A							
Helcom CORESET cand	104-40-5 und 2515	Nonylphenole (NP), Nonylphenols (NP)	A							
WFD Prio. Stoffe **24** (2013/39/EU, Ann 1)	104-40-5 und 8485	NP Nonylphenol, 4-Nonylphenol)	A	0,3	0,3	2	2			
HELCOM BSAP 12	140-66-9	OP Octylphenols (OP)	A							
Helcom CORESET cand	140-66-9	OP Octylphenole (OP), Octylphenols (OP)	A							
OSPAR List for priority action 28	140-66-9	OP octylphenol	A							
WFD Prio. Stoffe 25 (2013/39/EU, Ann 1)	140-66-9	OP Octylphenol (4-(1,1',3,3'-Tetramethylbutyl)-phenol)	A	0,1	0,01	not appl.	not appl.			
Helcom CORESET 09 final	15307-86-5	Diclofenac	A							
OgewV Kand. 04 (inoffizielle Mitteilung)	15307-86-5	Diclofenac	A	0,1						
WFD watch list Prio Stoffe 48	15307-86-5	Diclofenac	A	0,1						
Helcom CORESET 09 final	57-63-6	17-α-ethynelestradiol	A							
WFD watch list Prio Stoffe 46	57-63-6	17-α-Ethinylestradiol	A	0,0035						
Helcom CORESET 05-03 final	120-12-7	Anthracene	A	0,1	85					3
OSPAR ICG EAC 03	120-12-7	Anthracene	A	5	85		290			3
WFD Prio. Stoffe **02** (2013/39/EU, Ann 1)	120-12-7	Anthracen	A	0,1	0,1	0,1	0,4			3
Helcom CORESET 05-05 final	191-24-2	Benzo[*ghi*]perlyene	A	0,002	85					45
OSPAR ICG EAC 10	191-24-2	Benzo[*ghi*]perylene	A	80	85	2,5	110			45
WFD Prio. Stoffe 28d (2013/39/EU, Ann 1)	191-24-2	Benzo[ghi]perylen	A	footnote11	footnote11	8,2*10^-3	8,2*10^-4	footnote11		45
Helcom CORESET 05-11 final	193-39-5	indeno[1,2,3-cd]pyrene	A	0,002	240					50
OSPAR ICG EAC 11	193-39-5	Indeno[1,2,3-cd]pyrene	A	103	240	2,4				50
WFD Prio. Stoffe 28e (2013/39/EU, Ann 1)	193-39-5	Indeno[1,2,3-cd]pyren	A	footnote11	footnote11	not appl.	not appl.	footnote11		50
Helcom CORESET 05-08 final	205-99-2	Benzo[b]fluoranthene	A	0,03						
WFD Prio. Stoffe 28b (2013/39/EU, Ann 1)	205-99-2	Benzo[b]fluoranthen	A	footnote11	footnote11	0,017	0,017	footnote11		
Helcom CORESET 05-02 final	206-44-0	Fluoranthene	A	0,1	600					20
OSPAR ICG EAC 05	206-44-0	Fluoranthene	A	39	600	12,2	110			20
WFD Prio. Stoffe 15 (2013/39/EU, Ann 1)	206-44-0	Fluoranthen	A	0,0063	0,0063	0,12	0,12	30		
Helcom CORESET 05-07 final	207-08-9	Benzo[k]fluoranthene	A	0,03						
WFD Prio. Stoffe 28c (2013/39/EU, Ann 1)	207-08-9	Benzo[k]fluoranthen	A	footnote11	footnote11	0,017	0,017	footnote11		
Helcom CORESET 05-06 final	50-32-8	Benzo[a]pyrene	A	0,05	430					15
OSPAR ICG EAC 09	50-32-8	Benzo[a]pyrene	A	30	430	1,4	600			15
WFD Prio. Stoffe 28a (2013/39/EU, Ann 1)	50-32-8	Benzo[a]pyren	A	1,7*10^-4	1,7*10^-4	0,27	0,027			15
Helcom CORESET 05-04 final	91-20-3	Naphtalene	A	1200	160					5
OSPAR ICG EAC 01	91-20-3	Naphtalene	A	8	160		340			5
WFD Prio. Stoffe 22 (2013/39/EU, Ann 1)	91-20-3	Naphthalene	A	2	2	130	130			5
CLRTAP Recognized	PAH	polychlorierte Naphtaline PCNs	A							
OSPAR List for priority action 39	PAH	Polychlorinated naphthalenes	A							
Stockh. Conv. POPs cand	PAH	Chlorinated naphthalenes	A							
OSPAR List for priority action 01	PAH	PAH hier: Aromatic hydrocarbon	A							
OSPAR List for priority action 30	PAH	PAH polyaromatic hydrocarbons (PAHs)	A							
OSPAR MIME Indicator 3	PAH	PAH	A							
CLRTAP Originally included	PAH	PAHs	A							
WFD Prio. Stoffe **28** (2013/39/EU, Ann 1)	PAH	PAH Polycyclische aromatische Kohlenwasserstoffe (PAK)	A	not appl.	not appl.	not appl.	not appl.			
HELCOM BSAP 09	25637-99-4	Hexabromocyclododecane (HBCDD)	A							
Helcom CORESET 02	25637-99-4	Hexabromocyclododecane (auch als bromierte BFR bei OSPAR)	A	0,031	170					
Stockh. Conv. POPs cand	25637-99-4	Hexabromocyclododecane	A							
WFD Prio. Stoffe **43** (2013/39/EU, Ann 1)	25637-99-4	Hexabromocyclododecan)	A	0,0016	0,0008	0,5	0,05	167		
HELCOM BSAP 04	32534-81-9	PBDE pentaBDE Pentabromodiphenyl ether (pentaBDE)	A							
Helcom CORESET 01 final	32534-81-9	PBDE pentaBDE Pentabromodiphenyl ether (pentaBDE)	A							
Helcom CORESET 01 final	32534-81-9	PBDE's	A	0,0049	4,5					
Stockh. Conv. POPs 20 (2009)	32534-81-9	PBDE hier: TetraBDE and pentaBDE (Annex A)	A							
WFD Prio. Stoffe **05** (2013/39/EU, Ann 1)	32534-81-9	PBDE Bromierte Diphenylether (nur Tetra- ,Penta-,Hexa-, Hepta)	A			0,14	0,014	0,0085		
Stockh. Conv. POPs 16 (2009)	36355-01-8	PBDE hier: Hexabromobiphenyl (Annex A)	A							
OSPAR List for priority action 12	BFR	PBDE ua hier: BFR (Anm. PBDE, TBBPA, HBCD)	A							
OSPAR MIME Indicator 5	BFR	PBDE	A							
Stockh. Conv. POPs 17 (2009)	BFR	Hexabromodiphenyl ether and heptabromodiphenyl ether (Annex A)	A							
OSPAR ICG EAC 15	32598-14-4	PCB-105, 2,3,3',4,4'-Pentachlorobiphenyl	A			0,75				
OgewV 107 (UBA: Verbleib in OgewV)	35065-27-1	PCB-153 (2,2',4,4',5,5'-Hexachlorobiphenyl)	A	0,0005	0,02					0.0/0.05
OSPAR ICG EAC 18	35065-27-1	PCB-153, C12H4Cl6, 2,2',4,4',5,5'-Hexachlorobiphenyl	A	0,19	40	0,6	80			0.0/0.05
OgewV 106 (UBA: Verbleib in OgewV)	35065-28-2	PCB-138 (2,2',3,4,4',5'-Hexachlorobiphenyl)	A	0,0005	0,02					0.0/0.05
OSPAR ICG EAC 17	35065-28-2	PCB-138, C12H4Cl6, 2,2',3,4,4',5'-Hexachlorobiphenyl	A	0,15	7,9	0,6	15,8			0.0/0.05
OSPAR ICG EAC 20	35065-29-3	PCB-180, C12H3Cl7, 2,2',3,4,4',5,5'-Heptachlorobiphenyl	A	0,1	12	0,6	24			0.0/0.05
OgewV 103 (UBA: Verbleib in OgewV)	35693-99-3	PCB-52, (2,2',5,5'-Tetrachlorobiphenyl)	A	0,0005	0,02					0.0/0.05
OSPAR ICG EAC 13	35693-99-3	PCB-52, C12H6Cl4, 2,2',5,5'-Tetrachlorobiphenyl	A	0,12	2,7	0,75	5,4			0.0/0.05
OgewV 104 (UBA: Verbleib in OgewV)	37680-73-2	PCB-101 (2,2',4,5,5'-Pentachlorobiphenyl)	A	0,0005	0,02					0.0/0.05
OSPAR ICG EAC 14	37680-73-2	PCB-101, C12H5Cl5, 2,2',4,5,5'-Pentachlorobiphenyl	A	0,14	3	0,7	6			0.0/0.05
OSPAR ICG EAC 19	38380-08-4	PCB-156, 2,3,3',4,4',5-Hexachlorobiphenyl	A			0,6				
OgewV 102 (UBA: Verbleib in OgewV)	7012-37-5	PCB-28 (2,4,4'-Trichlorobiphenyl)	A	0,0005	0,02					0.0/0.05
OSPAR ICG EAC 12	7012-37-5	PCB-28, C12H7Cl3, 2,4,4'-Trichlorobiphenyl	A	0,22	1,7	0,75	3,2			0.0/0.05

Stoff-Inventar	CAS-Nr.	Stoffname							BC water	BC/LC sed.
Helcom CORESET				UQN marin [µg/l]	UQN benth [µg/kg]				(Quelle: OSAPR CEMP)	
OgewV	UQN oberird. Gew einschl. Übergangsgew. sowie Küstengew nach § 7 Abs 5 Satz 2 WHG *Quelle:* OGewV 2011 Anlage 5 (zu § 2 Nummer 6, § 5 Absatz 4 Satz 2 und 3, § 9 Absatz 2			water [µg/l]	susp o sed [mg/kg]					
OSPAR ICG EAC	OSPAR EAC Rep. Belgium, based on CEMP; Mus=Mussels; Oy=Oysters; BAC=Background assessment concentration; ERL=Effects Range low; EAC= Environmental assessment criteria			BAC Sed. [µg/kg]	ERL/EAC Sed. [µg/kg]	BAC Mussels / oysters [µg/kg]	ERL/EAC Mussels / oysters [µg/kg]		BC = backgr.conc. LC=low con.	
WFD Prio. Stoffe	AA-annual average; MAC-maximum allowed concentration			AA EQS Inland surface waters [µg/l]	AA EQS other surface waters [µg/l]	MAC EQS Inland surface waters [µg/l]	MAC EQS other surface waters [µg/l]	EQS biota [µg/kg] wet weight	[µg/l]	[µg/kg dry weight]
CLRTAP Originally included	Dx	PCBs	A							
CLRTAP Originally included	Dx	PCDDs/PCDFs	A							
HELCOM BSAP 01	Dx	Dioxins (PCDD), furans (PCDF) & dioxin-like polychlorinated biphenyls	A							
Helcom CORESET 04 final	Dx	Dioxins (PCDD), furans (PCDF), dioxin-like PCBs	A							
OSPAR List for priority action 13	Dx	PCB polychlorinated biphenyls (PCBs)	A							
OSPAR List for priority action 14	Dx	PCDD polychlorinated dibenzodioxins (PCDDs)	A							
OSPAR List for priority action 15	Dx	PCDF polychlorinated dibenzofurans (PCDFs)	A							
OSPAR MIME Indicator 2	Dx	PCB	A							
Stockh. Conv. POPs 09 (2004)	Dx	PCB Polychlorinated Biphenyls (PCB) (Annex A and C)	A							
Stockh. Conv. POPs 11 (2004)	Dx	PCDD u.a. Polychl. dibenzo-p-dioxins (dx) and polychl. dibenzofurans (	A							
Stockh. Conv. POPs 12 (2004)	Dx	PCDF Polychlorinated dibenzofurans (Annex C)	A							
WFD Prio. Stoffe **37** (2013/39/EU, Ann 1)	Dx	Dioxine + dl-PCB	A	nurBiota	Sum PCDD+PCDF+PCB-DL 0,0065µg/kg TEQ					
HELCOM BSAP 02	36643-28-4	Tributyltin compounds (TBT)	A							
Helcom CORESET 08 final	36643-28-4	Tributyltin compounds (TBT)	A							
OSPAR ICG EAC 24	36643-28-4	Tributyltin compounds TBT	A			5	21			
OSPAR List for priority action 05	36643-28-4	Tributyltin TBT u.a. hier: organic tin compounds	A							
WFD Prio. Stoffe **30** (2013/39/EU, Ann 1)	36643-28-4	Tributylzinnverbindungen 2)(Tributylzinn-Kation) (TBT)	A	0,0002	0,0002	0,0015	0,0015			
OSPAR MIME Indicator 4	organotin	organotin	A							
CLRTAP Recognized	45298-90-5	PFOS	A							
HELCOM BSAP 07	45298-90-6	PFOS, Perfluorooctane sulfonate	A							
Helcom CORESET 03 final	45298-90-6	PFOS, Perfluorooctane sulfonate (PFOS)	A							
OgewV Kand. 14 (inoffizielle Mitteilung)	45298-90-6	PFOS, Perfluoroctan-sulfonsäure (PFOS)	A							
OSPAR List for priority action 07	45298-90-6	PFOS/PFOA perfluorooctanyl sulphonic acid and its salts (PFOS)	A							
Stockh. Conv. POPs 21 (2009)	45298-90-6	PFOS, its salts and PFOsulfonyl fluoride (PFOSF) (Annex B)	A							
WFD Prio. Stoffe **35** (2013/39/EU, Ann 1)	45298-90-6	PFOS	A	6,5*10^-4	1,3*10^-4	36	7,2	9,1		
OSPAR List for priority action 50	465-73-6	Isodrin	A							
Prio andere St. 09a-5 (OgewV Anl. 7 Tab 2)	465-73-6	Isodrin (als Cyclodien Pestizid)	A							
WFD Prio. Stoffe 09a (2013/39/EU, Ann 2)	465-73-6 u variou	Cyclodiene pesticides (Aldrien, Dieldrin, Endrin, Isodrin	AC	Σ0,01	Σ0,005	not appl.	not appl.			
CLRTAP Originally included	118-74-1	Hexachlorobenzene	A							
OSPAR MIME Indicator 6	118-74-1	HCB	A							
Stockh. Conv. POPs 06 (2004)	118-74-1	Hexachlorobenzol, Hexachlorobenzene (Annex A and C)	A							
WFD Prio. Stoffe **16** (2013/39/EU, Ann 1)	118-74-1	Hexachlorobenzol, (Hexachlorbenzen)	A			0,05	0,05	10		
OSPAR List for priority action 21	608-73-1	Hexachlorocyclohexane isomers (HCH)	A							
WFD Prio. Stoffe **18** (2013/39/EU, Ann 1)	608-73-1	Hexachlorocyclohexan	A	0,02	0,002	0,04	0,02			
CLRTAP Originally included	608-73-2	Hexachlorocyclohexanes	A							
HELCOM BSAP 16	115-29-7	Endosulfan	A							
OSPAR List for priority action 20	115-29-7	endosulfan	A							
Stockh. Conv. POPs 22 (2011)	115-29-7	Endosulfan	A							
WFD Prio. Stoffe **14** (2013/39/EU, Ann 1)	115-29-7	Endosulfan	A	0,005	0,0005	0,01	0,004			
OSPAR List for priority action 19	115-32-2	Dicofol	A							
WFD Prio. Stoffe **34** (2013/39/EU, Ann 1)	115-32-2	Dicofol	A	1,3*10^-3	3,2*10^-5	not appl.	not appl.			
OSPAR List for priority action 23	87-86-5	Pentachlorophenol (PCP)	A							
Stockh. Conv. POPs cand	87-86-5	Pentachlorophenol	A							
WFD Prio. Stoffe 27 (2013/39/EU, Ann 1)	87-86-5	Pentachlorphenol	A	0,4	0,4	1	1			
OSPAR List for priority action 24	1582-09-8	trifluralin	A							
WFD Prio. Stoffe **33** (2013/39/EU, Ann 1)	1582-09-8	Trifluralin	A	0,03	0,03	not appl.	not appl.			
Helcom CORESET 06 final	7439-92-1	Pb Lead (Pb)	A							25000
OSPAR ICG EAC 23	7439-92-1	Pb	A	38000	47000	1300	7500			25000
OSPAR List for priority action 03	7439-92-1	pb lead and organic lead compounds	A							25000
OSPAR MIME Indicator 1c	7439-92-1	Pb concentrations in biota und sediment	A							25000
WFD Prio. Stoffe 20 (2013/39/EU, Ann 1)	7439-92-1	Pb Blei und Bleiverbindungen	A	1,2	1,3	14	14			25000
HELCOM BSAP 17	7439-97-6	Hg Mercury	A							50
Helcom CORESET 06 final	7439-97-6	Hg Quecksilber, Mercury	A							50
OSPAR ICG EAC 21	7439-97-6	Hg Mercury	A	70	150	90	2500			50
OSPAR List for priority action 04	7439-97-6	hg mercury and organic mercury compounds	A							50
OSPAR MIME Indicator 1a	7439-97-6	Hg	A							50
WFD Prio. Stoffe **21** (2013/39/EU, Ann 1)	7439-97-6	Hg Quecksilber und Quecksilberverbindungen	A			0,07	0,07	20		50
WFD Prio. Stoffe 23 (2013/39/EU, Ann 1)	7440-02-0	Ni Nickel und Nickelverbindungen	A	4	8,6	34	34			30000
HELCOM BSAP 18	7440-43-9	Cd Cadmium	A							200
Helcom CORESET 06 final	7440-43-9	Cd Cadmium, Cadmium	A							200
OSPAR ICG EAC 22	7440-43-9	Cd	A	310	1200	960	5000			200
OSPAR List for priority action 02	7440-43-9	cd cadmium	A							200
OSPAR MIME Indicator 1b	7440-43-9	Cd concentrations in biota und sediment	A							200
WFD Prio. Stoffe **06** (2013/39/EU, Ann 1)	7440-43-9	Cd Cadmium und Cadmiumverbindungen	A	0,08(Kl.1) 0,0	0,2	0,45(Kl.1) 0,	0,45(Kl.1) 0			200
OSPAR List for priority action 29	84-74-2	dibutylphthalate (DBP), diethylhexylphthalate (DEHP) certain phthalat	A							
WFD Prio. Stoffe **12** (2013/39/EU, Ann 1)	117-81-7	DEHP Bis(2-ethyl-hexyl) phthalat (DEHP)	A	1,3	1,3	not appl.	not appl.			
HELCOM BSAP 14	85535-84-8	SCCP Short-chain chlorinated paraffins (chloroalkanes, C_{10-13})	A							
OSPAR List for priority action 16	85535-84-8	SCCP short chained chlorinated paraffins (SCCP)	A							
Stockh. Conv. POPs cand	85535-84-8	SCCP Short-chained chlorinated paraffins	A							
WFD Prio. Stoffe **07** (2013/39/EU, Ann 1)	85535-84-8	SCCP (Chloralkanes C10-13)	A	0,4	0,4	1,4	1,4			
CLRTAP Recognized	85535-84-9	SCCPs	A							
CLRTAP Recognized	87-68-3	Hexachlorobutadiene	A							
OSPAR MIME Indicator 7	87-68-3	Hexachlorobutadiene HCBD	A							
Stockh. Conv. POPs cand	87-68-3	Hexachlorobutadiene	A							
WFD Prio. Stoffe **17** (2013/39/EU, Ann 1)	87-68-3	Hexachlorobutadien	A			0,6	0,6	55		

Table 5-3: Hazardous substances, Categorie “B” – listed in OSPAR and/or HELCOM

substance inventory	CAS-Nr.	name						BC water	BC/LC sed.
Helcom CORESET				UQN marin [µg/l]	UQN benthisch [µg/kg]			(Quelle: OSAPR CEMP) BC = backgr. conc. LC=low con.	
OSPAR ICG EAC	OSPAR EAC Rep. Belgium, based on CEMP; Mus=Mussels; Oy=Oysters; BAC=Background assessment concentration; ERL=Effects Range low; EAC= Environmental assessment criteria			BAC Sed. [µg/kg]	ERL/EAC Sed. [µg/kg]	BAC Mussels / oysters [µg/kg]	ERL/EAC Mussels / oysters [µg/kg]	[µg/l]	[µg/kg dry weight]
OSPAR List for priority action 11	108-70-3	Trichlorobenzene hier:1,3,5-trichlorobenzene	B						
Helcom CORESET 01 final	1163-19-5	PBDE, Decabromodiphenyl ether (decaBDE)	B						
OSPAR List for priority action 10	120-82-1	Trichlorobenzene hier: 1,2,4-trichlorobenzene	B						
Helcom CORESET 05-09 final	129-00-0	Pyrene	B		665				13
OSPAR ICG EAC 06		Pyrene	B	24	665	9	100		13
OSPAR List for priority action 42	1321-64-8	Pentachloronaphthalene*	B						
OSPAR List for priority action 40	1321-65-9	Trichloronaphthalene*	B						
OSPAR ICG EAC 04	132-65-0	Dibenzothiophene	B		190				0,6
OSPAR List for priority action 43	1335-87-1	Hexachloronaphthalene*	B						
OSPAR List for priority action 41	1335-88-2	Tetrachloronaphthalene	B						
Helcom CORESET 03	1763-23-1	PFOA Perfluorooctanoic acid (PFOA)	B						
OSPAR List for priority action 38	1825-21-4	Pentachloroanisole	B						
Helcom CORESET 05-14 final	208-96-8	Acenaphtylene	B		44				
OSPAR List for priority action 48	2104-64-5	Ethyl O-(p-nitrophenyl) phenyl phosphonothionate (EPN)*	B						
Helcom CORESET 05-12 final	218-01-9	Chrysene (Benzo[a]phenanthren)	B		384				11
OSPAR ICG EAC 08		Chrysene / Triphenylene	B	20	384	8,1			11
OSPAR List for priority action 51	2227-13-6	Tetrasul	B						
OSPAR List for priority action 45	2234-13-1	Octachloronaphthalene*	B						
OSPAR List for priority action 25	23593-75-1	Clotrimazole	B						
Helcom CORESET 05-17 final	2433-56-9	Hydroxy phenanthren hier: 1-hydroxy phenanthrene	B						
OSPAR List for priority action 37	28680-45-7	Heptachloronorbornene (CAS 28680-45-7 / 2440-02-0)	B						
OSPAR List for priority action 33	294-62-2	Cyclododecane	B						
OSPAR ICG EAC 16	31508-00-6	PCB-118, 2,3',4,4',5-Pentachlorobiphenyl	B	0,17	0,6	0,6	1,2		0.0/0.05
OSPAR List for priority action 44	32241-08-0	Heptachloronaphthalene*	B						
Helcom CORESET 01 final	32536-52-0	PBDE OctaBDE , Octabromodiphenyl ether (octaBDE)	B						
OSPAR List for priority action 07	335-67-1	PFOA Perfluorooctanoic acid	B						
OSPAR List for priority action 35	36065-30-2	Bromphenyl hier: 2,4,6-bromophenyl 1-2(2,3-dibromo-2-methylpropyl)	B						
OSPAR List for priority action 32	4904-61-4	Cycododecatriene hier: 1,5,9 cyclododecatriene	B						
OSPAR List for priority action 06	51000-52-3	Neodecanoic acid, ethenyl ester	B						
OSPAR List for priority action 52	512-04-9	Diosgenin	B						
Helcom CORESET 05-16 final	5315-79-7	Hydroxypyren hier: 1-hydroxypyrene	B						
Helcom CORESET 05-01 final	53-70-3	Dibenzo[a,h]anthracen	B		63,4				
OSPAR List for priority action 47	55525-54-7	Trimethylcyclohexyl: 3,3'-(ureylenedimethylene)bis(3,5,5-trimethylcyclohexyl	B						
Helcom CORESET 05-10 final	56-55-3	Benzo[a]anthracene	B		261				9
OSPAR List for priority action 34	59447-55-1	Pentabromomethylester hier: 2-propenoic acid, (pentabromo)methyl ester	B						
Helcom CORESET 08	668-34-8	Triphenyltin compounds (TPhT)	B						
OSPAR List for priority action 49	70124-77-5	Flucythrinate	B						
OSPAR List for priority action 46	70776-03-3	Naphthalene, chloro derivs.	B						5
OSPAR List for priority action 22	72-43-5	Methoxychlor	B						
OSPAR List for priority action 26	732-26-3	Tri-tert-butylphenol hier: 2,4,6-tri-tert-butylphenol	B						
Helcom CORESET 07 final	7440-46-2	Cs Cesium-137	B						
Helcom CORESET suppl	7440-50-8	Cu Copper	B						20000
Helcom CORESET suppl	7440-66-6	Zn Zinc	B						90000
OSPAR List for priority action 17	793-24-8	Dimethylbutylamin o diphenylamin hier: 4-(dimethylbutylamino)diphenylami	B						
OSPAR List for priority action 08	79-94-7	Tetrabromobisphenol A (TBBP-A)	B						
OSPAR List for priority action 31	81-15-2	Musk Xylene	B						
Helcom CORESET 05-15 final	83-32-9	Acenapthene	B		16				
Helcom CORESET 05-13 final	85-01-8	Phenanthrene	B		240				17
OSPAR ICG EAC 02		Phenanthrene	B	32	240	11	1700		17
OSPAR List for priority action 36	85-22-3	Pentabromoethylbenzene	B						
OSPAR List for priority action 09	87-61-6	Trichlorobenzene hier: 1,2,3-trichlorobenzene	B						
Helcom CORESET suppl	biocides	Organochlorine compounds/pesticides	B						
OSPAR List for priority action 18	biocides	Organophosphate	B						
COHIBA 06b	9016-45-9	NPE, Nonylphenol ethoxylates (NPE)	B						
HELCOM BSAP 11	9016-45-9	NPE Nonylphenol ethoxylates (NPE)	b						
OSPAR List for priority action 27	9016-45-9	NPE nonylphenol/ethoxylates (NP/NPEs) and related substances	B						
COHIBA 07b	NP/OP-E	OPE Octylphenol ethoxylate (OPEO), Octylphenol ethoxylates (OPE)	B						
HELCOM BSAP 13	NP/OP-E	OPE Octylphenol ethoxylates (OPE)	B						

Table 5-4: Hazardous substances, Categorie “C” – listed in the WFD, not in OSPAR or HELCOM

Stoff-Inventar	CAS-Nr.	Stoffname						
OgewV	UQN oberird. Gew einschl. Übergangsgew. sowie Küstengew nach § 7 Abs 5 Satz 2 WHG - *Quelle:* OGewV 2011 Anlage 5 (zu § 2 Nummer 6, § 5 Absatz 4 Satz 2 und 3, § 9 Absatz 2 Satz 1)			Wasserphase [µg/l] 2)	Schwebstoff Sed [mg/kg] 3)			
WFD Prio. Stoffe		AA-annual average; MAC-maximum allowed concentration		AA EQS Inland surface waters [µg/l]	AA EQS other surface waters [µg/l]	MAC EQS Inland surface waters [µg/l]	MAC EQS other surface waters [µg/l]	EQS biota [µg/kg] wet weight
OgewV 088	1024-57-3	Heptachlorepoxid	C	0,1				
WFD Prio. Stoffe **44** (2013/39/EU, Ann 1)	1024-57-3	Heptachlor/-epoxid	C	2*10^-7	1*10^-8	3*10^-4	3*10^-5	6,7*10^-3
REACH svhc	107-06-2	Dichloroethan hier 1,2-Dichlorethan	C					
sin list 331	107-06-2	Dichloroethan hier 1,2-dichloroethane	C					
WFD Prio. Stoffe 10 (2013/39/EU, Ann 1)	107-06-2	Dichloroethan hier: 1,2-Dichlorethan = Ethylendichlorid	C	10	10	not appl.	not appl.	
WFD Prio. Stoffe 31 (2013/39/EU, Ann 1)	12002-48-1	Trichlorbenzole 12) (Trichlorbenzene)	C	0,4	0,4	not appl.	not appl.	
WFD Prio. Stoffe 29 (2013/39/EU, Ann 1)	122-34-9	Simazin	C	1	1	4	4	
WFD Prio. Stoffe **36** (2013/39/EU, Ann 1)	124495-18-7	Quinoxyfen	C	0,15	0,015	2,7	0,54	
Prio andere St. 29a (OgewV Anl. 7 Tab 2)	127-18-4	Tetrachlorethylen (tetrachlorethen, Per, Tetrachlorethylen	C					
sin list 544	127-18-4	Perchloroethylene; tetrachloroethylene	C					
WFD Prio. Stoffe 29a (2013/39/EU, Ann 2)	127-18-4	Tetrachlorethylene	C	10	10	not appl.	not appl.	
OgewV Kand. 09 (inoffizielle Mitteilung)	15687-27-1	Ibuprofen	C					
WFD zKandidat	15687-27-1	Ibuprofen	C	0,1				
WFD Prio. Stoffe 01 (2013/39/EU, Ann 1)	15972-60-8	Alachlor	C	0,3	0,3	0,7	0,7	
WFD Prio. Stoffe 03 (2013/39/EU, Ann 1)	1912-24-9	Atrazine	C	0,6	0,6	2	2	
OgewV Kand. 11 (inoffizielle Mitteilung)	28159-98-0	Cybutryn	C					
WFD Prio. Stoffe 40 (2013/39/EU, Ann 1)	28159-98-0	Cybutryn (Substitut für TBT)	C	0,0025	0,0025	0,016	0,016	
WFD Prio. Stoffe 09 (2013/39/EU, Ann 1)	2921-88-2	Chlorpyrifos (Chlorpyrifos-ethyl)	C	0,03	0,03	0,1	0,1	
CLRTAP Originally included	309-00-1	Aldrin	C					
Prio andere St. 09a-2 (OgewV Anl. 7 Tab 2)	309-00-2	Aldrin (als Cyclodien Pestizide)	C					
Stockh. Conv. POPs 01 (2004)	309-00-2	Aldrin (Annex A)	C					
sin list 41	3194-55-6	Hexabromocyclododecane	C					
WFD Prio. Stoffe 13 (2013/39/EU, Ann 1)	330-54-1	Diuron	C	0,2	0,2	1,8	1,8	
WFD Prio. Stoffe 19 (2013/39/EU, Ann 1)	34123-59-6	Isoproturon	C	0,3	0,3	1	1	
WFD Prio. Stoffe 39 (2013/39/EU, Ann 1)	42576-02-3	Bifenox	C	0,012	0,0012	0,04	0,004	
WFD Prio. Stoffe 09a (2013/39/EU, Ann 2)	465-73-6 u varic	Cyclodiene pesticides (Aldrien, Dieldrin, Endrin, Isodrin	AC	Σ0,01	Σ0,005	not appl.	not appl.	
WFD Prio. Stoffe 08 (2013/39/EU, Ann 1)	470-90-6	Chlorfenvinphos	C	0,1	0,1	0,3	0,3	
WFD watch list Prio Stoffe 47	50-28-2	17-β-Estradiol	C	0,0004				
CLRTAP Originally included	50-29-3	DDT	C					
Prio andere St. 09b-2 (OgewV Anl. 7 Tab 2)	50-29-3	DDT hier: Para-para-DDT	C					
Stockh. Conv. POPs 10 (2004)	50-29-3	DDT (Annex A)	C					
WFD Prio. Stoffe 09b2 (2013/39/EU, Ann 2)	50-29-3	para DDT	C	0,01	0,01	not appl.	not appl.	
Prio andere St. 09b-1 (OgewV Anl. 7 Tab 2)	50-29-3 ua	DDT insgesamt 13)	C					
WFD Prio. Stoffe 09b1 (2013/39/EU, Ann 2)	50-29-3 ua	DDT	C	0,025	0,025	not appl.	not appl.	
WFD Prio. Stoffe 41 (2013/39/EU, Ann 1)	52315-07-8	Cypermethrin	C	8*10^-5	8*10^-6	6*10^-4	6*10^-5	
Prio andere St. 06a (OgewV Anl. 7 Tab 2)	56-23-5	Tetrachlorkohlenstoff	C					
WFD Prio. Stoffe 06a (2013/39/EU, Ann 2)	56-23-5	Carbon Tetrachloride	C	12	12	not appl.	not appl.	
OgewV 139 (UBA: zur Umsetzung WFD)	57-12-5	Cyanid	C	10				
WFD zKandidat	57-12-5	Cyanid (freies Cyanid)	C	0,01				
CLRTAP Originally included	60-57-1	Dieldrin	C					
Prio andere St. 09a-3 (OgewV Anl. 7 Tab 2)	60-57-1	Dieldrin (als Cyclodien Pestizid)	C					
Stockh. Conv. POPs 03 (2004)	60-57-1	Dieldrin (Annex A)	C					
CLRTAP Recognized	608-93-5	Pentachlorobenzene	C					
Stockh. Conv. POPs 19 (2009)	608-93-5	Pentachlorobenzene (Annex A and C)	C					
WFD Prio. Stoffe **26** (2013/39/EU, Ann 1)	608-93-5	Pentachlorobenzol	C	0,007	0,0007	not appl.	not appl.	
OgewV 078 (UBA: zur Umsetzung WFD)	62-73-7	Dichlorvos	C	0,0006				
WFD Prio. Stoffe 42 (2013/39/EU, Ann 1)	62-73-7	Dichlorvos	C	6*10^-4	6*10^-5	7*10^-4	7*10^-5	
WFD Prio. Stoffe 32 (2013/39/EU, Ann 1)	67-66-3	Trichlormethan	C	2,5	2,5	not appl.	not appl.	
sin list 267	71-43-2	Benzene	C					
WFD Prio. Stoffe 04 (2013/39/EU, Ann 1)	71-43-2	Benzene (Benzol)	C	10	8	50	50	
CLRTAP Originally included	72-20-8	Endrin	C					
Prio andere St. 09a-4 (OgewV Anl. 7 Tab 2)	72-20-8	Endrin (als Cyclodien Pestizid)	C					
Stockh. Conv. POPs 04 (2004)	72-20-8	Endrin (Annex A)	C					
WFD Prio. Stoffe 38 (2013/39/EU, Ann 1)	74070-46-5	Aclonifen	C	0,12	0,012	0,12	0,012	
WFD Prio. Stoffe 11 (2013/39/EU, Ann 1)	75-09-2	Dichloromethane	C	20	20	not appl.	not appl.	
CLRTAP Originally included	76-44-8	Heptachlor	C					
OgewV 087	76-44-8	Heptachlor	C	0,1				
Stockh. Conv. POPs 05 (2004)	76-44-8	Heptachlor (Annex A)	C					
Prio andere St. 29b (OgewV Anl. 7 Tab 2)	79-01-6	Trichlorethylen (Trichlorethen)	C					
REACH svhc	79-01-6	Trichlorethen	C					
WFD Prio. Stoffe 29b (2013/39/EU, Ann 2)	79-01-6	Trichlorethylen (Trichlorethen)	C	10	10	not appl.	not appl.	
OgewV Kand. 18 (inoffizielle Mitteilung)	886-50-0	Terbutryn	C					
WFD Prio. Stoffe 45 (2013/39/EU, Ann 1)	886-50-0	Terbutryn	C	0,065	0,0065	0,34	0,034	

Table 5-5: Hazardous substances, Categorie "D" – just river-specific; listed in OGewV, Table 5

Stoff-Inventar	CAS-Nr.	Stoffname			
OgewV	UQN oberird. Gew einschl. Übergangsgew. sowie Küstengew nach § 7 Abs 5 Satz 2 WHG - *Quelle:* OGewV 2011 Anlage 5 (zu § 2 Nummer 6, § 5 Absatz 4 Satz 2 und 3, § 9 Absatz 2 Satz 1)			Wasser-phase [µg/l]	Schwebst o Sed [mg/kg]
OgewV 024	100-00-5	Chlornitrobenzol hier: 1-Chlor-4-nitrobenzol (1-Cl-4-Nitrobenzen)	D	10	
OgewV 084	100-41-4	Ethylbenzol	D	10	
OgewV 006	100-44-7	Benzylchlorid (-Chlortoluol)	D	10	
sin list 337	100-44-7	a-chlorotoluene	D		
OgewV 095	10265-92-6	Methamidophos	D	0,1	
Fh-IME Ableitung UQN [µg/l]	105827-78-9	Imidacloprid	D	0,00065	
OgewV Kand. 10 (inoffizielle Mitteilung)	105827-78-9	Imidacloprid	D		
Fh-IME Ableitung UQN [µg/l]	10605-21-7	Carbendazim	D	0,05	
OgewV Kand. 02 (inoffizielle Mitteilung)	10605-21-7	Carbendazim	D		
OgewV 133	106-42-3	Dimethylbenzol hier: 1,4-Dimethylbenzol (p-Xylol)	D	10	
OgewV 038	106-43-4	Chlortoluol hier: 4-Chlortoluol (4-Chlortoluen)	D	1	
OgewV 062	106-46-7	Dichlorbenzol hier: 1,4-Dichlorbenzol (1,4-Dichlorbenzol)	D	10	
OgewV 014	106-47-8	Chloranilin hier: 4-Chloranilin	D	0,05	
sin list 416	106-47-8	4-chloroaniline	D		
OgewV 033	106-48-9	Chlorphenol hier: 4-Chlorphenol	D	10	
Fh-IME Ableitung UQN [µg/l]	106-89-8	Epichlorhydrin	D		
OgewV 083 (UBA: Verbleib in OgewV; IME)	106-89-8	Epichlorhydrin	D	10	
sin list 347	106-89-8	1-chloro-2,3-epoxypropane	D		
OgewV 051	106-93-4	Dibromethan hier: 1,2-Dibromethan	D	2	
sin list 330	106-93-4	1,2-dibromoethane	D		
OgewV 035	107-05-1	Chlorpropen hier: 3-Chlorpropen (Allylchlorid)	D	10	
OgewV 017	107-07-3	Chlorethanol hier: 2-Chlorethanol	D	10	
OgewV 132	108-38-3	Dimethylbenzol, hier 1,3-Dimethylbenzol (m-Xylol)	D	10	
OgewV 037	108-41-8	Chlortoluol 3-Chlortoluol	D	10	
OgewV 013	108-42-9	Chloranilin hier: 3-Chloranilin	D	1	
OgewV 032	108-43-0	Chlorphenol hier: 3-Chlorphenol	D	10	
OgewV 064	108-60-1	Dichlordiisopropylether	D	10	
OgewV 044	108-77-0	Cyanurchlorid (2,4,6-Trichlor-1,3,5-triazin)	D	0,1	
OgewV 116	108-88-3	Toluol	D	10	
OgewV 015	108-90-7	Chlorbenzol	D	1	
OgewV 079	109-89-7	Diethylamin	D	10	
Fh-IME Ableitung UQN [µg/l]	1113-02-6	Omethoat	D	0,000084	
OgewV 098 (UBA: Verbleib in OgewV; IME)	1113-02-6	Omethoat	D	0,1	
Fh-IME Ableitung UQN [µg/l]	111991-09-4	Nicosulfuron	D	0,00175	
OgewV Kand. 13 (inoffizielle Mitteilung)	111991-09-4	Nicosulfuron	D		
OgewV Kand. 06 (inoffizielle Mitteilung)	114-07-8	Erythromycin	D		
OgewV 077	120-36-5	Dichlorprop	D	0,1	
OgewV 072	120-83-2	Dichlorphenol hier: 2,4-Dichlorphenol	D	10	
OgewV 023	121-73-3	Chlornitrobenzol hier: 1-Chlor-3-nitrobenzol (1-Cl-3-Nitrobenzen)	D	1	
OgewV 092 (UBA: zur Umsetzung WFD)	121-75-5	Malathion	D	0,02	
OgewV 026	121-86-8	Chlor-nitrotoluol hier: 2-Chlor-4-nitrotoluol	D	1	
OgewV 085 (UBA: zur Umsetzung WFD)	122-14-5	Fenitrothion	D	0,009	
OgewV 081	124-40-3	Dimethylamin	D	10	
OgewV 118	126-73-8	Tributylphosphat (Phosphorsäuretributylester)	D	10	
OgewV 048	126-75-0	Demeton-S	D	0,1	
OgewV 034	126-99-8	Chloropren	D	10	
sin list 336	126-99-8	2-chlorobuta-1,3-diene	D		
OgewV 154 (UBA: 2011 aufgenommen)	133855-98-8	Epoxiconazol	D	0,2	
OgewV 157 (UBA: 2011 aufgenommen)	137641-05-5	Picolinafen	D	0,007	
Fh-IME Ableitung UQN [µg/l]	142459-58-3	Flufenacet	D		
OgewV Kand. 07 (inoffizielle Mitteilung)	142459-58-3	Flufenacet	D		
Fh-IME Ableitung UQN [µg/l]	14488-53-0	Dibutylzinn-Kation	D		
OgewV 052 (UBA: Verbleib in OgewV; IME)	14488-53-0	Dibutylzinn-Kation	D	0,01	0,1
sin list 35	1461-22-9	Tributyltin chloride	D		
Fh-IME Ableitung UQN [µg/l]	1461-25-2	Tetrabutylzinn	D		
OgewV 113 (UBA: Verbleib in OgewV; IME)	1461-25-2	Tetrabutylzinn	D	0,001	0,04

Stoff-Inventar	CAS-Nr.	Stoffname			
OgewV	UQN oberird. Gew einschl. Übergangsgew. sowie Küstengew nach § 7 Abs 5 Satz 2 WHG - *Quelle:* OGewV 2011 Anlage 5 (zu § 2 Nummer 6, § 5 Absatz 4 Satz 2 und 3, § 9 Absatz 2 Satz 1)			Wasser-phase [µg/l]	Schwebst o Sed [mg/kg]
Fh-IME Ableitung UQN [µg/l]	14816-18-3	Phoxim	D		
OgewV 109 (UBA: Verbleib in OgewV; IME)	14816-18-3	Phoxim	D	0,008	
Fh-IME Ableitung UQN [µg/l]	149961-52-4	Dimoxystrobin	D	0,00316	
OgewV Kand. 05 (inoffizielle Mitteilung)	149961-52-4	Dimoxystrobin	D		
OgewV 137 (UBA: zur Umsetzung WFD)	15545-48-9	Chlortoluron	D	0,4	
OgewV 124	15950-66-0	Trichlorphenol hier: 2,3,4-Trichlorphenol	D	1	
OgewV 151 (UBA: 2011 aufgenommen)	1689-84-5	Bromoxynil	D	0,5	
OgewV 111	1698-60-8	Pyrazon (Chloridazon)	D	0,1	
OgewV 050	17040-19-6	Demeton-S-methyl-sulphon	D	0,1	
Fh-IME Ableitung UQN [µg/l]	1746-81-2	Monolinuron	D		
OgewV 097 (UBA: Verbleib in OgewV; IME)	1746-81-2	Monolinuron	D	0,1	
OgewV 144 (UBA: zur Umsetzung WFD)	18691-97-9	Methabenzthiazuron	D	2	
OgewV 155 (UBA: 2011 aufgenommen)	21087-64-9	Metribuzin	D	0,2	
OgewV 158 (UBA: 2011 aufgenommen)	23103-98-2	Pirimicarb	D	0,09	
OgewV 117	24017-47-8	Triazophos	D	0,03	
OgewV 134	25057-89-0	Bentazon	D	0,1	
OgewV 003 (UBA: zur Umsetzung WFD)	2642-71-9	Azinphos-ethyl	D	0,01	
OgewV 108 (UBA: Verbleib in OgewV)	28655-71-2	PCB-180 (2,2',3,4,4',5,5'-Heptachlorobiphenyl)	D	0,0005	0,02
OgewV 101 (UBA: zur Umsetzung WFD)	298-00-0	Parathion-methyl	D	0,02	
OgewV 047	298-03-3	Demeton-O	D	0,1	
OgewV 082	298-04-4	Disulfoton	D	0,004	
Fh-IME Ableitung UQN [µg/l]	298-46-4	Carbamazepin	D	0,032	
OgewV Kand. 01 (inoffizielle Mitteilung)	298-46-4	Carbamazepin	D	0,5	
OgewV 099	301-12-2	Oxydemeton-methyl	D	0,1	
OgewV 009	302-17-0	Chloralhydrat	D	10	
OgewV 136 (UBA: zur Umsetzung WFD)	314-40-9	Bromacil	D	0,6	
OgewV 068	3209-22-1	Dichlornitrobenzol hier: 1,2-Dichlor-3-nitrobenzol	D	10	
OgewV 091	330-55-2	Linuron	D	0,1	
OgewV 152 (UBA: 2011 aufgenommen)	333-41-5	Diazinon	D	0,01	
Fh-IME Ableitung UQN [µg/l]	3380-34-5	Triclosan	D		
OgewV Kand. 19 (inoffizielle Mitteilung)	3380-34-5	Triclosan	D		
sin list 303	3380-34-5	Triclosan	D		
OgewV Kand. 12 (inoffizielle Mitteilung)	37350-58-6	Metoprolol	D		
OgewV 140 (UBA: zur Umsetzung WFD)	38260-54-7	Etrimphos, Etrimfos	D	0,004	
OgewV 028	38939-88-7	Chlor-nitrotoluol hier: 3-Chlor-4-nitrotoluol	D	1	
OgewV 145 (UBA: zur Umsetzung WFD)	51218-45-2	Metolachlor	D	0,2	
OgewV 141 (UBA: zur Umsetzung WFD)	51235-04-2	Hexazinon	D	0,07	
OgewV 119	52-68-6	Trichlorfon	D	0,002	
OgewV 030	5367-28-2	Chlor-nitrotoluol hier: 5-Chlor-2-nitrotoluol	D	1	
OgewV 067	540-59-0	Dichloroethen hier: 1,2-Dichloroethen	D	10	
OgewV 061	541-73-1	Dichlorbenzol hier: 1,3-Dichlorbenzol	D	10	
OgewV 075	542-75-6	Dichlorpropen hier: 1,3-Dichlorpropen	D	10	
OgewV 086 (UBA: zur Umsetzung WFD)	55-38-9	Fenthion	D	0,004	
OgewV 055	554-00-7	Dichloranilin hier: 2,4-Dichloranilin	D	1	
OgewV 100 (UBA: zur Umsetzung WFD)	56-38-2	Parathion-ethyl	D	0,005	
OgewV 043	56-72-4	Coumaphos	D	0,07	
CLRTAP Originally included	57-74-9	Chlordane	D		
OgewV 010	57-74-9	Chlordane (cis und trans)	D	0,003	
Stockh. Conv. POPs 02 (2004)	57-74-9	Chlordane (Annex A)	D		
OgewV 148 (UBA: zur Umsetzung WFD)	5915-41-3	Terbuthylazin	D	0,5	
OgewV 018	59-50-7	Chlor-methylphenol hier: 4-Chlor-3-Methylphenol	D	10	
OgewV 159 (UBA: 2011 aufgenommen)	60207-90-1	Propiconazol	D	1	
Fh-IME Ableitung UQN [µg/l]	60-51-5	Dimethoate	D		
OgewV 080 (UBA: Verbleib in OgewV; IME)	60-51-5	Dimethoat	D	0,1	
OgewV 054	608-27-5	Dichloranilin hier: 2,3-Dichloranilin	D	1	
OgewV 057	608-31-1	Dichloranilin hier: 2,6-Dichloranilin	D	1	

Stoff-Inventar	CAS-Nr.	Stoffname			
OgewV	UQN oberird. Gew einschl. Übergangsgew. sowie Küstengew nach § 7 Abs 5 Satz 2 WHG - *Quelle:* OGewV 2011 Anlage 5 (zu § 2 Nummer 6, § 5 Absatz 4 Satz 2 und 3, § 9 Absatz 2 Satz 1)			Wasser-phase [µg/l]	Schwebst o Sed [mg/kg]
OgewV 127	609-19-8	Trichlorphenol hier: 3,4,5-Trichlorphenol	D	1	
OgewV 070	611-06-3	Dichlornitrobenzol, hier: 1,3-Dichlor-4-nitrobenzol	D	10	
OgewV 039	615-65-6	Chlor-p-toluidin hier: 2-Chlor-p-toluidin	D	10	
OgewV 150 (UBA: 2011 aufgenommen)	62-53-3	Anilin	D	0,8	
sin list 301	62-53-3	Aniline	D		
OgewV 059	626-43-7	Dichloranilin hier: 3,5-Dichloranilin	D	1	
OgewV 143 (UBA: zur Umsetzung WFD)	67129-08-2	Metazachlor	D	0,4	
OgewV 089	67-72-1	Hexachloroethan	D	10	
OgewV 094	7085-19-0 / 93-65-2	Mecoprop	D	0,1	
OgewV 110	709-98-8	Propanil	D	0,1	
OgewV 120	71-55-6	Trichloroethane hier: 1,1,1-Trichlorethan	D	10	
OgewV Kand. 17 (inoffizielle Mitteilung)	723-46-6	Sulfamethoxazol	D	0,1	
OgewV 147 (UBA: zur Umsetzung WFD)	7287-19-6	Prometryn	D	0,5	
OgewV 161 (UBA: 2011 aufgenommen)	7440-22-4	Ag Silber	D	0,02	
OgewV 162 (UBA: 2011 aufgenommen)	7440-28-0	Thallium	D	0,2	
Fh-IME Ableitung UQN [µg/l]	7440-38-2	As Arsenic	D		
OgewV 002 (UBA: Verbleib in OgewV; IME)	7440-38-2	As Arsen	D		40
OgewV 138 (UBA: zur Umsetzung WFD)	7440-47-3	Cr Chrom	D		640
OgewV 130 (UBA: Verbleib in OgewV)	75-01-4	Vinylchlorid (Chlorethylen) , Metabolit of Tetrachlorethen	D	2	
OgewV 065	75-34-3	Dichlorethan hier: 1,1-Dichlorethan	D	10	
OgewV 066	75-35-4	Dichlorethen hier: 1,1-Dichlorethen (Vinylidenchlorid)	D	10	
OgewV 128	76-13-1	Trichlortrifluorethan hier: 1,1,2-Trichlortrifluorethan	D	10	
OgewV 160 (UBA: 2011 aufgenommen)	7782-49-2	Se Selen 5)	D	3	
OgewV 096	7786-34-7	Mevinphos	D	0,0002	
OgewV 073	78-87-5	Dichlorpropan hier: 1,2-Dichlorpropan	D	10	
OgewV 076	78-88-6	Dichlorpropen hier: 2,3-Dichlorpropen	D	10	
OgewV 121	79-00-5	Trichlorethan hier: 1,1,2-Trichlorethan	D	10	
Fh-IME Ableitung UQN [µg/l]	79-11-8	Chloressigsäure	D	0,058	
OgewV 011 (UBA: Verbleib in OgewV; IME)	79-11-8	Chloressigsäure	D	10	
OgewV 115	79-34-5	Tetrachlorethan hier: 1,1,2,2-Tetrachlorethan	D	10	
Fh-IME Ableitung UQN [µg/l]	80214-83-1	Roxythromycin	D		
OgewV Kand. 15 (inoffizielle Mitteilung)	80214-83-1	Roxythromycin	D		
OgewV 046	8065-48-3	Demeton (Summe von Demeton-O und -S)	D	0,1	
OgewV Kand. 03 (inoffizielle Mitteilung)	81103-11-9	Clarithromycin	D		
OgewV 153 (UBA: 2011 aufgenommen)	83164-33-4	Diflufenican	D	0,009	
OgewV 135 (UBA: zur Umsetzung WFD)	834-12-8	Ametryn	D	0,5	
OgewV 027	83-42-1	Chlor-nitrotoluol hier: 2-Chlor-6-nitrotoluol	D	1	
OgewV 004 (UBA: zur Umsetzung WFD)	86-50-0	Azinphos-methyl	D	0,01	
OgewV 040	87-60-5	Chlor-o-toluidin hier: 3-Chlor-o-toluidin	D	10	
OgewV 123	88-06-2	Trichlorphenol hier: 2,4,6-Trichlorphenol	D	1	
OgewV 022	88-73-3	Chlornitrobenzol hier: 1-Chlor-2-nitrobenzol (1-Cl-2-Nitrobenzen)	D	10	
OgewV 025	89-59-8	Chlor-nitrotoluol hier: 4-Chlor-2-nitrotoluol	D	10	
OgewV 029	89-60-1	Chlor-nitrotoluol hier: 4-Chlor-3-nitrotoluol	D	1	
OgewV 071	89-61-2	Dichloronitrobenzene hier: 1,4-Dichlor-2-nitrobenzol	D	10	
OgewV 021	89-63-4	Nitroanilin hier: 4-Chlor-2-nitroanilin	D	3	
OgewV 019	90-13-1	Chlornaphtalin hier: 1-Chlornaphthalin	D	1	
OgewV 020	90-13-1	Chlornaphthaline (techn. Mischung)	D	0,01	
OgewV 063	91-94-1	Dichlorbenzidin hier: 3,3-Dichlorbenzidin	D	10	
sin list 530	91-94-1	3,3'-dichlorobenzidine	D		
OgewV 049	919-86-8	Demeton-S-methyl	D	0,1	
OgewV 008	92-52-4	Biphenyl	D	1	
OgewV 005	92-87-5	Benzidin	D	0,1	
sin list 408	92-87-5	Benzidine and its salts	D		
OgewV 126	933-75-5	Trichlorphenol hier: 2,3,6-Trichlorphenol	D	1	
OgewV 125	933-78-8	Trichlorphenol hier: 2,3,5-Trichlorphenol	D	1	
OgewV 112	93-76-5	Trichlorphenoxyessigsäure hier: 2,4,5-T	D	0,1	
OgewV 093	94-74-6	MCPA (2-Methyl-4-Chlorphenoxyessigsäure)	D	0,1	

5.1.7.2 Clustering of substances: functional groups

The following subchapters contain fact sheet for individual substances, grouped by function. The fact-sheets compile the Pro and Contra for each substance, using the introduced criteria. A quick overview over the respectively relevant criteriea in each subchapter is given by the following symbols (see Figure 5-12).

Persistency (persistent, very persistent)	P vP
Toxic or Potential for endocrine disruptor	e
Potential for long-range transport and atmospheric deposition (e.g. CLRTAP)	 Long range transport
High Bioaccumulation / Biomagnification Potential	 bioaccumulation
Substances with very large production volumes	 High volumes
Substances with „marine applications“ directly discharged to sea	 Directly to sea
Marine sediments as secondary source	
Regulation status	§§§ POPs, CLRTAP EU ban; §§ HELCOM, OSPAR, WFD; § River specific
Suggested for monitoring under Descriptor 9 (first and second priorization)	D9 D9
Candidate for national monitoring under MSFD D8 (BLMP[51] (colors show suggested monitoring matrix: biota, sediment, suspended solids, water)	BLMP biota; BLMP sed.; BLMP susp.; BLMP water
Substance Category “D”	D

Figure 5-12: Illustration of the criteria.

The referenced inventories are summarized for each group.

”1“ means: listed in the mentioned inventory

”0,5“ means: listed in the mentioned inventory as a candidate substance

In the column ”others“ the compared lists are named.

[51] BLMP: Bund-Länder-Messprogramm → working group hazardous substances

In column “ABCD” the category according to the first approach (see Chapter 5.1.7.1) is documented. The substances that belong to category “C”, i.e. mentioned in WFD but not in OSPAR or HELCOM – mainly pesticides – are highlighted in light red.

The fact sheets conclude with a qualitative overview and a recommendation.

5.1.7.2.1Group ”heavy metals“

A total of 9 different heavy metals are mentioned.

Table 5-6: Group ”heavy metals“.

	others	Riv-spec	WFD	HELCOM	OSPAR	ABCD
Cadmium Cd	LRT		1	1	1	A
Mercury Hg	LRT		1	1	1	A
Lead Pb	LRT		1	1	1	A
Copper Cu				1		B
Zinc Zn				1		B
Nickel Ni			1			A
Arsenic As	IME	1				D
Chrome Cr		1				D
Selenium Se		1				D

First Priorization: Cd, Hg, Pb → potential for LRT and in focus of WFD, OSPAR and HELCOM; EC mercury strategy is currently in process[52].

HELCOM additionally includes Cu, Zn.

Arsenic is suggested to be monitored in D9. The results from UBA-Project “UQN” with Fh IME will provide further information.

Cu, Zn, As are considered second priority.

The main industrial sources for heavy metals have been regulated (IPPC[53]) → downward trend for North Sea (NS) and Baltic Sea (BS).

The relative importance of diffuse sources has increased e.g. Cd as contaminant in P-fertilizer.

Important non-regulated uses are: lead-acid batteries, ammunition and fishing equipment for Pb, dental amalgam for Hg.

Preliminary Conclusions ”heavy metals“

Pros for including Cd, Pb, Hg in MSFD-Monitoring

[52] http://ec.europa.eu/environment/chemicals/mercury

[53] Integrated Pollution Prevention and Control, EG-Richtlinie: RL 96/61/EG: IVU-Richtlinie

- Most industrial uses highly regulated, but many important sources remaining: e.g lead-acid batteries, ammunition and fishing and diving equipment (Pb), dental amalgam Hg
- Emission to air stagnant or increasing
- Potential for LRT, atmospheric pathway very important
- Very persistent (non degradable) → oceans as "final sinks "
- OSPAR cessation target 2020 will not be reached for Cd, Pb, Hg
- Suggested to be monitored under D9 (Seafood) (JRC 2010)
- Suggested to be monitored by BLMP[54] in all matrices (water, biota, sediment, suspended matter)

Cons against including Cd, Pb, Hg in MSFD-Monitoring

- Most industrial uses highly regulated
- Overall decreasing trends measured in NS and BS

Cd, Pb, Hg,	vP	Long range transport	bioaccumulation		D9	BLMP
Cu, Zn	vP	BLMP				
Ni	vP	BLMP				
As	vP	D9	Directly to sea	BLMP		
Cr	vP	BLMP				
Se	vP	BLMP				

Cd, Pb, Hg considered relevant for the marine environment, already in focus of OSPAR and/or HELCOM, Arsenic may indicate dumped munition

5.1.7.2.2Group "heavy metals - organotins"

Historically organotins were widely used as antifoulant. The application is banned by IMO[55]. They are also banned as detergents and biocides, but still in use as stabilizers in PVC, catalysts in chemical reactions as glass coatings with large production volumes.

[54] Bund Länder Messprogramm – working document

[55] IMO: International Maritime Organization

Table 5-7: Group ”heavy metals - organotins“.

	others	Riv-spec	WFD	HELCOM	OSPAR	ABCD
Organotins as group						
Tributyltin compounds (TBT)			1	1	1	A
Triphenyltin compounds (TPhT)		1		1		B
Dibutylzinn-Kation	IME	1				D
Tetrabutylzinn	IME	1				D

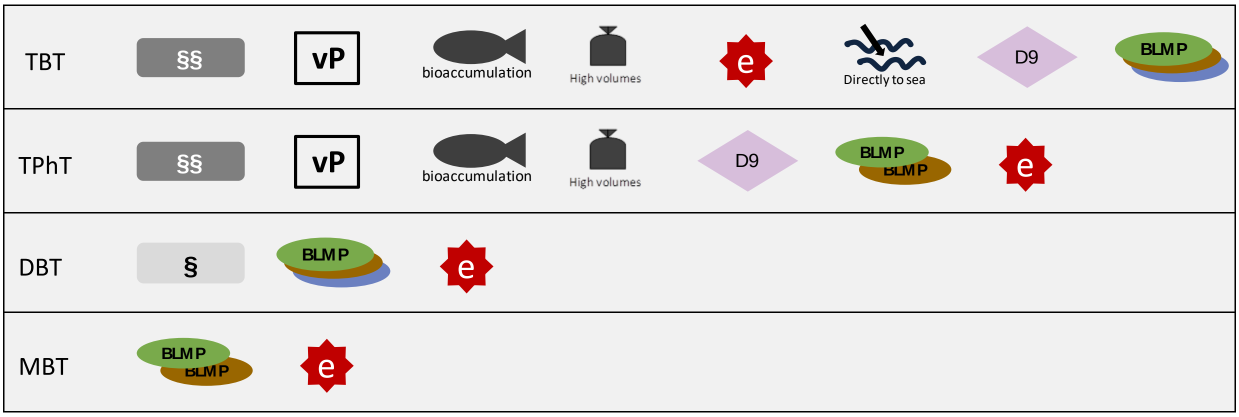
Preliminary Conclusions ”heavy metals - organotins“

Pros for including organo-tins in MSFD-Monitoring

- Largest direct pathway to the marine environment banned but direct input from marine sediments remains
- Large production volumes
- Well-documented biological effect: TBT-related imposex of gastropods, Eelpout
- OSPAR cessation target 2020 will not be reached for organo tin compounds
- TPhT not covered by WFD, particle-bound transport not covered by WFD
- Suggested to be monitored under D9 (Prio2) (Seafood) (JRC 2010)
- TBT: suggested to be monitored by BLMP (water, biota, sediment)
- TPhT: suggested to be monitored by BLMP (biota, sediment, not in water and suspended matter)

Cons against including organo-tins in MSFD-Monitoring

- Largest direct pathway to the marine environment banned (antifouling)
- Other important applications (biocides, detergents) banned
- Overall decreasing trends measured in NS and BS (TBT, TPhT)
- Riverine inputs of TBT covered by WFD (including coastal waters)
- Tetrabutyltin: stable molecules, can be metabolized to triorganotin compounds (captured by monitoring triorganotins)



Organotins as a group (*TBT, TPhT) are considered relevant for the marine environment, already in focus of OSPAR and/or HELCOM*
Tetrabutyltin is considered less relevant for the marine environment and alos not addressed by BLMP.

5.1.7.2.3Group "BFR"

The polybrominated flame retardents (BFR) are a group of large production volumes. Some of these substances already are highly regulated.

Table 5-8: Group "BFR".

	oth-ers	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
BFR as group	1			1	1			A
PBDE, Decabromodiphenyl ether (decaBDE)	svhc			1	1		1	
Alpha-Hexabromocyclododecane							1	
Beta-Hexabromocyclododecane							1	
gamma-Hexabromocyclododecane							1	
Hexabromocyclododecane (HBCDD)	svhc		1	1		1	1	A
Hexabromocyclododecan (HBCDD) und alle größeren identifizierten diastereoisomeren Verbindungen:	svhc						1	
PBDE pentaBDE PBDE Pentabromodiphenyl ether (pentaBDE)	LRT		1	1		1		
PBDE hier: Tetrabromodiphenyl ether			1			1		
PBDE OctaBDE Octabromdiphenylether (octaBDE)	LRT			1		1*)	1	
Hexabromobiphenyl (Annex A)	LRT					1		
Tetrabromobisphenol A (TBBPA)					1		1	
Hexabromodiphenyl ether (Annex A) *)			1			1		
heptabromodiphenyl ether (Annex A) *)			1			1		

Hexabromocyclododecane (HBCDD) and penta- and octaBDEs are highly regulated (Stockholm Convention on POP, CLRTAP, EU ban), whereas DecaBDE and TBBP-A show lower regulation.

TBBP-A and HBCDD are mainly used in polystyrene. DecaBDE are mainly used in plastics, textile such as upholstery fabric, and synthetic carpets.

Expectable substitution dynamics: use of DecaBDE and TBBP-A may increase due to regulation of HBCDD, penta and octaBDEs.

BFR show diffuse emission patterns and the potentially largest emission is seen at End of Life of building material and goods → yet to come and not covered by regulation.

HBCDD, penta and octaBDEs→ Diffuse sources will be assessed by WFD monitoring (including coastal waters)

BFR have a reported potential for long range transboundary transport (LRT). The Input to Sea is only partly covered by existing regulations/measures.

TBBP-A may degrade to BP-A – an endocrine disruptive chemical (EDC) in sediments. This can be relevant for the marine environment (EDC problem: upcoming EU strategy)

More data is required.

Substance in focus of OSPAR → maybe relevant for North Sea

Preliminary Conclusions "BFR"

Pros for including BFR in MSFD-Monitoring

- Potentially largest emission at End of Life of building material and goods → "yet to come" and not covered by regulation
- Diffuse emission pattern
- Potential direct input to marine environment: shredder plants, often located in harbours
- Potential for LRT (except TBBP-A)
- Substitution dynamics: use of DecaBDE and TBBP-A may increase due to regulation of other BFR
- Large production volumes
- OSPAR cessation target 2020 will not be reached for BFR
- Suggested to be monitored under D9 (Prio2) (JRC 2010)

Cons against including BFR in MSFD-Monitoring

- Use of HBCDD, octaBDE and penta-BDE are highly regulated
- Decreasing trends can be expected
- Remaining BFR less toxic

HBCDD	vP	§§§	Long range transport	bioaccumulation	e	High volumes	BLMP
PentaBDE	vP	§§§	Long range transport	bioaccumulation	e	High volumes	BLMP
OctaBDE	vP	§§§	Long range transport	bioaccumulation	e	High volumes	BLMP
BDE85, BDE66	e	BLMP					
DecaBDE (BDE 209)	e	§§	BLMP				
Tetrabromodiphenyl ether (BDE 77)	e	§§	BLMP				
Hexabromobiphenyl (Annex A)	e	§§§					
Tetrabromobisphenol A (TBBPA)	e	§§	High volumes				
Hexabromodiphenyl ether	e	§§§	BLMP				

heptabromodiphenyl ether	e	§§§	BLMP

BFR as a group is considered relevant for the marine environment, already in focus of OSPAR and/or HELCOM.
Within the group BFR DecaBDE and TBBP-A are considered most relevant for the marine environment
Due to high regulation HBCDD, octaBDE and penta-BDE are considered less relevant for the marine environment

5.1.7.2.4Group "NP OP"

Nonylphenol ethoxylates (NPE) and Octylphenol ethoxylates (OPE) in the environment degrade to NP and OP.

Table 5-9: Group "NP OP".

	others	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
Nonylphenol ethoxylates (NPE) as group				1	1			B
Octylphenol ethoxylates (OPE) as group				1				B
Nonylphenole (NP), Nonylphenols (NP)			1	1			1	A
OP Octylphenols (OP) 4-(1,1,3,3-Tetra-methylbutyl)phenol	svhc		1	1	1		1	A

Most applications banned for NPE (EU ban 2004)

- Plant protection products and biocides
- Detergents

OPE is used as an intermediate in production of phenol/formaldehyde resins (also used in the recovery of oil in offshore processes).

Preliminary Conclusions "NP OP"

Pros for including NP OP in MSFD-Monitoring

- OP may be discharged directly to the marine environment (resin for recovery of oil in offshore processes)
- Large production volumes
- Particle bound transport not covered by WFD
- OSPAR cessation target 2020 will not be reached for OP
- OP does not fulfill B criteria for REACH, but equivalent concern for OSPAR and HELCOM

Cons against including NP OP in MSFD-Monitoring

- Use of NPE is highly regulated, with roll-on effects on OPE
- No reported potential for LRT
- Decreasing trends can be expected
- Riverine inputs covered by WFD (including coastal waters)

- OP does not fulfill B criteria for REACH
- OSPAR cessation target 2020 will be reached for NPE

NP	bioaccumulation	BLMP	§§§	e	
NPE	High volumes	bioaccumulation	§§§	e	
OP	Directly to sea	bioaccumulation	BLMP	§§	e
OPE	High volumes	bioaccumulation	§§	e	

NP/OP as group is considered relevant for the marine environment, already in focus of OSPAR and/or HELCOM
OP may be more relevant than NP given the regulatory status

5.1.7.2.5Group "PFC"

Per- or Polyflourated Chemicals are known as very persistent, exclusively of anthropogenic origin and not degradable under environmentally relevant conditions.

Table 5-10: Group "PFC".

	others	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
PFOS Perfluorooctanesulfonic acid (PFOS), its salts and perfluorooctanesulfonyl fluoride (PFOSF) (Annex B)	LRT + cand. svhc					1		A
PFOS Perfluoroctansulfonat (PFOS), Perfluorooctane sulfonate (PFOS)	LRT	0,5	1	1	1		1	A
PFOA Perfluoroctansäure (PFOA), Perfluorooctanoic acid (PFOA)				1	1		1	B

Hence PFOS is highly regulated (EU ban, POPs, CLRTAP).

Historically PFOS was widely used for industrial applications (metal plating, photolithography) and household products (impregnation of textiles, carpets, food wrappings). Today urban stock, landfills etc. can be expected as secondary (diffuse) sources.

For PFOA a voluntary agreement for phase out exists. It is a small number of producers.

PFOA does not fulfill B criteria, but is of equivalent concern for OSPAR and HELCOM.

Preliminary Conclusions "PFC"

Pros for including PFOS, PFOA in MSFD-Monitoring

- highly persistent
- Despite regulation diffuse sources for PFOS (and PFOA) remain, many precursor substances → diffuse emission pattern

- Substitution dynamics: use of other polyfluorinated substances may increase → Shorter chained and less toxic, but very mobile substitutes
- PFOS still produced in e.g. China (rising volumes!)
- Potential for LRT (PFOS; PFOA suspected)
- OSPAR cessation target 2020 will not be reached for PFOS
- Suggested to be monitored under D9 (Prio2) (Seafood) (JRC 2010)

Cons against including PFOS, PFOA in MSFD-Monitoring

- Use of PFOS is highly regulated
- Decreasing trends can be expected
- Riverine inputs of PFOS covered by WFD (including coastal waters)
- PFOA does not fulfill B criteria for REACH

PFOS	vP	§§§	Long range transport	bioaccumulation	High volumes	D9	BLMP
PFOA	vP	§	Long range transport	bioaccumulation	D9	BLMP	

PFC as a group is considered relevant for the marine environment, already in focus of OSPAR and/or HELCOM As an alternative to PFOS/PFOA, a sum parameter for PFC can be monitored → captures substitution dynamics

5.1.7.2.6Group "Chloralkanes"

The important substances of this group are the short-chain chlorinated paraffins (SCCP or chloroalkanes, C10-13) and the medium-chain chlorinated paraffins (MCCP or chloroalkanes, C14-17).

SCCP are highly regulated, but not reported for LRT. MCCP in focus of HELCOM (BSAP 15).

Table 5-11: Group "Chloralkanes".

	others	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
SCCP Short-chain chlorinated paraffins (SCCP or chloroalkanes, C10-13)			1	1	1		1	A
MCCP Medium-chain chlorinated paraffins (MCCP or chloroalkanes, C14-17)				1				B

Preliminary Conclusions "Chloralkanes"

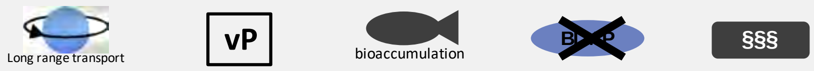
Pros for including SCCP, MCCP in MSFD-Monitoring

- Potential for LRT (SCCP)
- Air pathway and pathway via suspended particles not covered by WFD
- MCCP not included in WFD
- MCCP considered relevant by HELCOM

Cons against including SCCP, MCCP in MSFD-Monitoring

- Use of SCCP is highly regulated
- Decreasing trends can be expected
- Riverine inputs of SCCP covered by WFD (including coastal waters)
- OSPAR cessation target 2020 will be reached for SCCP
- Considerd not relevant by BLMP
- MCCP: point sources covered by IPPC, diffuse emissions less important

SCCP



Chlor-alkanes as group is considered relevant for the marine environment, already in focus of OSPAR and/or HELCOM
Due to high regulation, low importance of diffuse sources and potential for LRT, SCCP is considered less relevant for the marine environment.
For MCCP is considered relevant by HELCOM but not by OSPAR
→ maybe relevant for Baltic Sea (BS), more data is required.

5.1.7.2.7Group "Plasticizer"

Plasticizer refers to Dibutylphthalate (DBP) in particular, Diethylhexylphthalate (DEHP) and certain phthalates. They show a high production volume and potential for endocrine disruption.

Table 5-12: Group "Plasticizer".

	others	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
dibutylphthalate (DBP), diethylhexylphthalate (DEHP) certain phthalates					1			A
Dibutylphthalate (DBP)	svhc							B
Bis(2-ethylhexyl) phthalate; DEHP	svhc		1				1	A
Aniline		1					1	D
1,2-dichloroethane = ethylenedichloride	svhc						1	

Preliminary Conclusions "Plasticizer "

Pros for including plasticizer in MSFD-Monitoring

- DEHP potential for endocrine disruption
- High production volume
- Diffuse emission pattern (from products)
- Diffuse emissions from products not regulated → potentially more important than point sources
- OSPAR cessation target 2020 will not be reached for DBP and DEHP

Cons against including plasticizer in MSFD-Monitoring

- Point sources covered by IPPC
- Riverine inputs of DEHP covered by WFD (including coastal waters)
- EDC problem will be adressed by EU strategy
- no reported potential for LRT

DEHP	High volumes	e	§§	D9	BLMP
DBP	High volumes	e			

Plasticizer as agroup is considered relevant for the marine environment, already in focus of OSPAR and HEL-COM

5.1.7.2.8Group "Process Chemicals"

Table 5-13: Group „Process Chemicals".

	others	Riv-spec	WFD	HELCOM	OSPAR	SIN	ABCD
Hexachlorobutadien 2)			1		1	1	A
Trichlormethan	IKSR		1				C
Dichlormethane			1				C
Trichlorbenzene (var.)	cand svhc		1				C
Trichlorobenzene hier: 1,2,4-trichloroben-zene	vT				1	1	B
Trichlorobenzene hier:1,3,5-trichloroben-zene	vT				1		B
Trichlorobenzene hier: 1,2,3-trichloroben-zene					1	1	B
Benzylchlorid (-Chlortoluol)		1				1	D
Chloranilin hier: 4-Chloranilin		1				1	D
Epichlorhydrin	IME	1				1	D
Chloropren		1				1	D
Tetrachlorethylen (tetrachlorethen, Per, Tetrachlorethylen, Perchlorethylen)	vT	1	1			1	C
Carbon tetrachloride	IKSR	1					D
Trichloroethylene	svhc	1	1				C
Dichlorbenzidin hier: 3,3-Dichlorbenzidin		1				1	D
Trichlorophenoxyaceticacid hier: 2,4,5-Tri-chlorophenoxyaceticacid		1					D
1,2-dichlorobenzene		1				1	D
1,3-dichloropropan-2-ol		1				1	D
Dichlorbenzol hier: 1,4-Dichlorbenzol (1,4-Dichlorbenzol)		1					D
Dichloroethen hier: 1,2-Dichloroethen		1					D
Dichlorbenzol hier: 1,3-Dichlorbenzol		1					D
Hexachlorethan		1					D
Trichloroethane hier: 1,1,1-Trichlorethan		1					D
Vinylchlorid (Chlorethylen) , Metabolit of Tetrachlorethen		1					D
Dichlorethan (DCA) hier: 1,1-Dichlorethan		1					D
Dichlorethan hier: 1,2-Dichlorethan		1	1				C
Dichlorethen hier: 1,1-Dichlorethen (Vinyli-denchlorid)		1					D
Trichlorethan hier: 1,1,2-Trichlorethan		1					D
Tetrachlorethan hier: 1,1,2,2-Tetrachlor-ethan		1					D
Carbon Tetrachloride			1				C
Benzol			1			1	C

In the following Trichlorobenzene (TCB) is taken as a highly relevant chemical out of this group.

- TCB mainly used as intermediate for the production of herbicides, pigments and dyes (80%)
- not readily biodegradable and very toxic
- bioaccumulation potential very high
- Potential for endocrine disruption
- High production volume (produced in Germany as intermediate for herbicides)
- TCB medium regulation (WFD, OSPAR)

Preliminary Conclusions "process chemicals – here: TCB"

Pros for including TCB in MSFD Monitoring

- Potential for endocrine disruption
- High production volume
- OSPAR cessation target 2020 will not be reached for TCB
- Production may still occur in e.g. China and Russia

Cons against including TCB in MSFD Monitoring

- EDC problem will be adressed by EU strategy
- High production volume but many closed applications
- Diffuse sources less important → Point sources covered by IPPC
- Riverine inputs of TCB covered by WFD (including coastal waters)
- No reported potential for LRT

TCB	bioaccumulation	High volumes	e	§§
1,3,5-TBC	~~BLMP~~			
1,2,4-TBC	BLMP			
Dichloroethane (EDC)	High volumes	P		
Tetrachlorethylene	High volumes	P	~~BLMP~~	
Carbon tetrachloride	~~BLMP~~	D	§	
Hexachlorobenzene	§§§	BLMP		
Hexachloroutadiene	§§§	Bl BLMP		

Epichlorohydrinb	~~BLMP~~			
Benzyl chloride α-chlorotoluene	High volumes	~~BLMP~~	D	§
Trichloromethane (Chloroform)	High volumes	~~BLMP~~		
Dichlorobenzidin	~~BLMP~~			
Benzene	High volumes	~~BLMP~~	Directly to sea	
Tributyl phosphate	§	D	BLMP	
Chloroacetic acid	High volumes	D	§	

TCB is considered relevant for the marine environment

5.1.7.2.9Group "PAH"

The characteristics of Polycyclic aromatic hydrocarbons (PAH) are:

- consisting of three or more fused benzene rings
- toxic, persistent and bioaccumulative (especially in invertebrates, metabolized in higher organisms)
- potential for LRT
- natural components of tar, coal and oil (oil contains 0.2 - 7% PAH) → produced water from offshore installations is a direct pathway to the sea
- PAH has only few intended uses, e.g.
  - Naphthalene as intermediate for insecticides, stabilisators, pharmaceuticals, cosmetic additives and plasticiser
  - Anthracen as intermediate product for paints and plastics
  - Creosote (mixture with PAHs) is used in wood preservatives
  - Tar used in many applications e.g. roofs, floors

Table 5-14: Group ”PAH“ and “PNC”.

PAH	others	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
PAH as group	LRT		(1)	1	1			A
Benzo[a]pyrene / Benzo[def]chrysene			1	1	1		1	A
Naphthalene			1	1	1		1	A
Anthracene	svhc		1	1	1		1	A
Benzo[ghi]perlyene			1	1	1			A

PAH	others	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
Indeno[1,2,3-cd]pyrene			1	1	1			A
Fluoranthene			1	1	1			A
Benzo(e)acephenanthry-lene/Benzo[b]fluoranthene			1	1			1	A
Benzo(k)fluoranthene			1	1			1	A
Chrysen (Benzo[a]phenanthren)				1	1		1	
Benz[a]anthracene				1	1		1	B
Phenanthrene		1		1	1			B
Pyren				1	1			B
Dibenzothiophene					1			B
Hydroxypyren hier: 1-hydroxypyrene				1				B
Dibenz[a,h]anthracene				1			1	B
Acenapthene				1				B

PAH – PCN (polyclorinated naphtalenes)	others	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
Chlorinated naphtalene	LRT	1			1	0,5		B
pentachloronaphthalene*	LRT				1	0,5		B
trichloronaphthalene*	LRT				1	0,5		B
hexachloronaphthalene*	LRT				1	0,5		B
tetrachloronaphthalene*	LRT				1	0,5		B
octachloronaphthalene*	LRT				1	0,5		B
heptachloronaphthalene*	LRT				1	0,5		B
naphthalene, chloro derivs. *	LRT				1	0,5		B
Chlornaphtalin hier: 1-Chlornaphthalin	LRT	1				0,5		

Preliminary Conclusions ”PAH‟

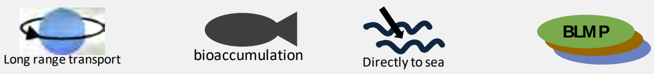
Pros for including PAH in MSFD-Monitoring

- reported potential for LRT, atmospheric pathway very important
- direct pathway to the marine environment via produced water from offshore installations
- OSPAR cessation target 2020 will not be reached for PAH
- “By product” of oil, coal and tar → diffuse emission pattern with a variety of sources (e.g. heating with coal)
- Suggested to be monitored under D9 (Seafood) (JRC 2010)

Cons against including PAH in MSFD-Monitoring

- Industrial uses highly regulated
- Monitoring deposition rates covers air pathway (CLRTAP)
- WFD covers riverine input

PAH



PAH as group is considered relevant for the marine environment, already in focus of OSPAR and/or HELCOM Indicator for oil spills and background loads

5.1.7.2.10 Group "Dx"

Dioxins (PCDD), furans (PCDF) and dioxin-like polychlorinated biphenyls (dl PCB)

- Dioxins are non-polar, lipophilic and persistent organic pollutants (POPs), high potential for biomagnification and bioconcentration in the food web
- toxic equivalent scheme (TEQ)
- not manufactured intentionally
- formed as unintentional by-products in heating and combustion processes (organic matter, chlorine compounds and a catalyst, dioxin window at 300-600°C)
- PCB phase-out by 2025 (POPs)
- Toxic, hydrophobic, strong bioconcentration
- Production in EU stopped in the 80`s, but may still continue in e.g. Russia and China

Table 5-15: Group "PAH" and "PNC".

	others	Riv.spec.	WFD	HELCOM	OSPAR	POP	SIN	ABCD
Dx (PCDD), Fur (PCDF), dl PCB as group	LRT		1	1				A
PCBs as group	LRT		1	1	1	1		A
PCDD as group	LRT		1	1	1	1		A
PCDF as group	LRT		1	1	1	1		A
PCB-118 (2,3',4,4',5-Pentachlorobiphenyl)		1			1			
PCB-105, 2,3,3',4,4'-Pentachlorobiphenyl					1			
PCB-153, C12H4Cl6, 2,2',4,4',5,5'-Hexachlorobiphenyl		1			1			
PCB-138, C12H4Cl6, 2,2',3,4,4',5'-Hexachlorobiphenyl		1			1			
PCB-180, C12H3Cl7, 2,2',3,4,4',5,5'-Heptachlorobiphenyl		1			1			
PCB-52, C12H6Cl4, 2,2',5,5'-Tetrachlorobiphenyl		1			1			
PCB-101, C12H5Cl5, 2,2',4,5,5'-Pentachlorobiphenyl		1			1			
PCB-156, 2,3,3',4,4',5-Hexachlorobiphenyl					1			
PCB-28, C12H7Cl3, 2,4,4'-Trichlorobiphenyl		1			1			

Preliminary Conclusions “Dx“

Pros for including Dx in MSFD-Monitoring

- reported potential for LRT
- High toxicity and bioaccumulation
- Produced unintentionally from a variety of sources → diffuse emission pattern (e.g. heating with wood and coal)
- OSPAR cessation target 2020 will not be reached for PCDDs, PCDFs, PCBs
- Suggested to be monitored under D9 (Seafood) (JRC 2010)

Cons against including Dx in MSFD-Monitoring

- Monitoring deposition rates covers air pathway (CLRTAP)
- Water pathway small compared to air
- WFD candidate substances → covers riverine input and coastal waters

PCDD	§§§	vP	Long range transport	bioaccumulation	D9
PCDF	§§§	vP	Long range transport	bioaccumulation	
dl-PCB	§§§	vP	Long range transport	bioaccumulation	D9

Dx as group is considered relevant for the marine environment, already in focus of OSPAR and/or HELCOM Monitoring deposition rates (CLRTAP) and higher trophic levels of food web (MSFD D9) may be sufficient

5.1.7.2.11 Group “Pesticides“

A total of ~50 pesticides are mentioned in the inventories. Many of these are highly regulated.

Table 5-16: Group ”Pesticides”.

	others	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
Endosulfan		1	1	1	1	1		A
DDT	LRT	1	1			1		C
Alpha-hexachlorocyclohexane (alpha-HCH)	LRT		1		1	1		A
Beta-HCH	LRT		1		1	1		A
Gamma-HCH (Lindane)	LRT		1		1	1		A
Hexachlorobenzene	LRT		1		1	1	1	A
Pentachlorobenzene	LRT		1			1		C
Aldrin	LRT	1	1			1		C
Chlordane	LRT	1				1		
Dieldrin	LRT	1	1			1		C
Endrin	LRT	1	1			1		C
Heptachlor	LRT	1				1		
Chlordecone	LRT					1		

	others	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
Mirex	LRT					1		
Toxaphene (Annex A)	LRT					1		

All Pesticides listed above have potential for long range transport by air pollution, and already are highly regulated as POPs.

	others	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
Pentachlorophenol			1		1		1	A
Trifluralin			1		1			A
Terbutryn		1	1					C
Simazin			1					C
Alachlor			1					C
Atrazine			1					C
Chlorpyrifos			1					C
Diuron			1					C
Isoproturon			1					C
Chlorfenvinphos			1					C
Dichlorvos		1	1					C
Heptachlor epoxide		1	1					C
Cybutryn		1	1					C
Quinoxifen			1					C
Bifenox			1					C
Cypermethrin			1					C
Aclonifen			1					C
Dicofol			1		1			C
Isodrin		1	1		1			A
Methoxychlor					1			B
ethyl O-(p-nitrophenyl) phenyl phos-phonothionate (EPN)*					1			B
Azinphos-ethyl		1						D
Azinphos-methyl		1						D
Biphenyl		1						D
Chlorethanol hier: 2-Chlorethanol		1						D
Coumaphos		1						D
Dichlorphenoxyessigsäure hier: 2,4-D	IME	1						D
Demeton (Summe von Demeton-O und -S)		1						D
Demeton-O		1						D
Demeton-S		1						D
Demeton-S-methyl		1						D
Demeton-S-methyl-sulphon	IME	1						D
Dibromethan hier: 1,2-Dibromethan		1						D
Dichlorphenol hier: 2,4-Dichlorphenol		1						D
Dichlorpropen hier: 1,3-Dichlorpropen		1						D
Dichlorprop		1						D
Dimethoat	IME	1						D
Disulfoton		1						D
Fenitrothion		1						D
Fenthion		1						D
Linuron		1						D

	others	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
Malathion		1						D
Mecoprop		1						D
Methamidophos		1						D
Mevinphos		1						D
Monolinuron		1						D
Omethoat	IME	1						D
Oxydemeton-methyl		1						D
Parathion-ethyl		1						D
Parathion-methyl		1						D
Phoxim		1						D
Propanil		1						D
Pyrazon (Chloridazon)		1						D
Triazophos		1						D
Trichlorfon		1						D
Bentazon		1						D
Ametryn		1						D
Bromacil		1						D
Chlortoluron		1						D
Etrimphos, Etrimfos		1						D
Hexazinon		1						D
Metazachlor		1						D
Methabenzthiazuron		1						D
Metolachlor		1						D
Prometryn		1						D
Terbuthylazin		1						D
Bromoxynil		1						D
Diazinon		1						D
Diflufenican		1						D
Epoxiconazol		1						D
Metribuzin		1						D
Picolinafen		1						D
Pirimicarb		1						D
Propiconazol		1						D
Dimoxystrobin	IME	0,5						D
Flufenacet	IME	0,5						D
Flurtamone	IME	0,5						D
Imidacloprid	IME	0,5						D
Nicosulfuron	IME	0,5						D
Triclosan	IME	0,5					1	D
Sulcotrion	IME	0,5						D

Preliminary Conclusions “Pesticides “

Pros for including Pesticides in MSFD-Monitoring

- Potential for LRT
- Partly large production volumes
- Pathway: emission from agriculture and e.g. building stock / house facades (diffuse emission patterns)

Cons against including Pesticides in MSFD-Monitoring

- The substances with potential for LRT are listed as POPs → highly regulated
- EU: positive listfor plant protection products (see footnote [20]) and biocides gives relatively good control in this sector
- OSPAR cessation target 2020 are expected to be reached for Dicofol, Endosulfan, HCH isomers (lindane), Methoxychlor, Trifluralin, Pentachlorophenol

DDT	Long range transport	§§§	BLMP
alpha-HCH	Long range transport	§§§	BLMP
beta HCH	Long range transport	§§§	BLMP
gamma-HCH	Long range transport	§§§	BLMP
Hexachlorobenzene	Long range transport	§§§	BLMP
Pentachlorobenzene	Long range transport	§§§	
Aldrin	Long range transport	§§§	
Chlordane	Long range transport	§§§	
Dieldrin	Long range transport	§§§	BLMP
Endrin	Long range transport	§§§	
Heptachlor	Long range transport	§§§	
Chlordecone	Long range transport	§§§	
Mirex	Long range transport	§§§	
Toxaphene	Long range transport	§§§	

Heptachlor	P	bioaccumulation		§§§	BLMP
heptachlore epoxide	P	bioaccumulation	§§§	BLMP	
Dicofol	P	bioaccumulation	§§	BLMP	
Trifluralin	Long range transport	e	bioaccumulation	§§	BLMP
2,4-D	BLMP	§	D	High volumes	
Diuron	BLMP	§§			
2-Methyl-4-chloro-phenoxyace-tic acid	BLMP	§	D		
Isoproturon	BLMP	§§			
Terbuthylazine	BLMP	§	D		
Metolachlor	BLMP	§	D		
Cybutryne	BLMP	§§			
Bentazone	BLMP	§	D		
Dichlorvos	BLMP	§§			
Chlorpyrifos	BLMP	§§			
Simazine	BLMP	§§			
Alachlor	BLMP	§§			
Atrazine	BLMP	§§			
Mecoprop	BLMP	§	D		
Terbutryn	BLMP	§§			
Azinphos ethyl	~~BLMP~~	§	D		
Azinphos methyl	~~BLMP~~	§	D		

Dimethoate	~~BLMP~~	§	D
Epichlorohydri	~~BLMP~~	§	D
Fenitrothion	~~BLMP~~	§	D

The following pesticides are considered not relevant by BLMP:

Dimethoate, Epichlorohydrin, Fenitrothion, Azinphos methyl, Azinphos ethyl

The following pesticides are listed in Stockholm POP, but not yet mentionend neither by OSPAR, HELCOM nor by WFD or river specific:

Mirex, Toxaphene, Chlordedone

Listed in Stockholm Convention on POP and beyond this only in river specific context:

Adrine, Chlordane, Dieldrin, Endrin, Heptachlor

Pesticides *considered more relevant for rivers and ground water than for the marine environment. Nevertheless, the monitoring of substances of category "C" must be mandatory in coastal waters.*

5.1.7.2.12 Group "Pharmaceuticals"

A total of 14 pharmaceuticals is mentioned in the different inventories.

Table 5-17: Group "Pharmaceuticals".

	others	Riv-spec	WFD	HELCOM	OSPAR	ABCD
Diclofenac	vT		1	1		A
Ibuprofen	vT					
Clotrimazole					1	B
Diosgenin					1	B
Carbendazim	vT + IME	0,5				D
Erythromycin		0,5				D
Carbamazepin	vT + IME	0,5				D
Chloralhydrat		1				D
Metoprolol		0,5				D
Sulfamethoxazol		0,5				D
Roxythromycin	IME	0,5				D
Clarithromycin		0,5				D
17-α-ethinylestradiol	vT		0,5	1		B
17-β-estradiol	vT		0,5			

Preliminary Conclusions "Pharmaceuticals "

Pros for including Pharamceuticals in MSFD-Monitoring

- High relevance as micro pollutants in an aquatic environment

Cons against including Pharamceuticals in MSFD-Monitoring

- Less relevant for the marine environment due to use pattern (households, hospitals)
- Higher relevance for rivers (receiving household and hospital wastewater) → WFD

17alpha-ethinylestradiol	e	BLMP		
17beta-estradiol	e	BLMP		
EE2 (+E1,E2, E3)	e	BLMP		
Diclofenac	High volumes	P	BLMP	§§
Clotrimazole	~~BLMP~~			

Group less relevant for the marine environment due to use pattern
Pharmaceuticals considered more relevant for rivers and ground water than for the marine environment.

5.1.7.2.13 Group "others"

In this group different other substance groups are subsumed.

Table 5-18: Group "others".

	others	Riv-spec	WFD	HELCOM	OSPAR	POP	SIN	ABCD
Musk xylene					1		1	B
cyanide		1	1					(C)
cesium BLMP				1				B

According to the Umweltbundesamt of Austria the Consumption of musk xylenes in the EU declined; which raises the question whether action is required or not[56].

OSPAR cessation target 2020 will not be reached for musk xylene.

Background for radionuclides: BSAP – radioactivity at pre-Tchernobyl level → special importance for the Baltic Sea (BS)

Suggested to be monitored under D9 (Prio2) (Seafood) (JRC 2010)

Cyanides and musk xylenes are considered to be medium relevant for marine waters

[56] http://www.umweltbundesamt.at

Radionuclides are considered relevant for marine waters

5.1.7.3 Direct emissions to sea

Additional substances may be relevant for the deep sea, which are not listed in the considered inventories. Possible sources include:

- ocean shipping
- offshore developments (oil and gas platforms, pipelines, wind farms)
- atmospheric deposition (e.g. due to long-range transport)
- nutrients - input from land-based emitters such as agriculture
- chemical munitions - a legacy of World War II; possible release of contaminants such as arsenic compounds.

The substances, which are to be identified in the ongoing project, will complement the final MSFD-list.

In Chapter 5.1.6 two substances resp. substance groups are already mentioned as relevant for direct emissions:

- PAH as natural component of oil (oil contains 0.2 - 7% PAH) → produced water from offshore installations is a direct pathway to the sea
- OP may be discharged directly to the marine environment (resin for recovery of oil in offshore processes)

5.1.7.3.1 Direct emissions to sea (offshore and shipping)

All phases of oil and gas production activities can have an impact on the marine environment. Besides accidential oil and chemical spills, the produced waters are constant source of discharge of hazardous substances. In addition, legal and illegal discharges from ships as well as their exhaust are relevant sources for emissions of hazardous substances.

Within the OSPAR work area “Offshore Oil & Gas Industry” some information and data concerning direct emissions to sea are worked out, published as a report in the OSPAR offshore industry series 2012[57]. The composition of discharged chemicals is differentiated in the following categories:

PLONOR	Substance on OSPAR List of Substances Used and Discharged Offshore which are Considered to Pose Little or no Risk to the Environment (PLONOR) (Agreement Number: 2004-10, update 2008).
LCPA	Substance listed in the OSPAR List of Chemicals for Priority Action (LCPA) (including its updates) (Agreement Number: 2004-12).
Inorganic LC50 or EC50 > 1mg/l	Inorganic substance with LC50 or EC50 less than 1 mg/l.

[57] http://www.ospar.org/v_publications/browse.asp?menu=01330305830000_000000_000000

Biodegradation <20%	Biodegradation of the substance is less than 20% in OECD 306, Marine BODIS or any other accepted marine protocols; or less than 20% during 28 days in freshwater (ready test).
Substance meets two of three Criteria	Substance meets two of the following three criteria: marine protocol); or in the absence of valid results for such tests; less than 60% 301E); II. bioaccumulation: BCF > 100 or log Pow >= 3 and molecular weight <700; III. toxicity: LC50 < 10mg/l or EC50 < 10mg/l; if toxicity values <10 mg/l are derived from limit tests to fish, actual fish LC50 data should be submitted
Inorganic LC50 or EC50 > 1mg/l	Inorganic substance with LC50 or EC50 over 1 mg/l.

The following figures are based on the data of OSPAR "Offshore Oil & Gas Industry". This data is collected from the abutting countries. The fact, that this includes the official data from Germany was confirmed by the Landesamt für Bergbau, Energie und Geologie of Niedersachsen (LBEG), which is responsible for the two German offshore installations[58].

In Figure 5-13 and Figure 5-14 the yearly oil production and the corresponding spill and discharge of dispersed oil is shown (data from Denmark, Germany, Ireland, Netherlands, Norway, Spain and the United Kingdom). Spills and discharges account for about 0.01 to 0.02 ‰ of the oil production. Oil production declined in the past decade.

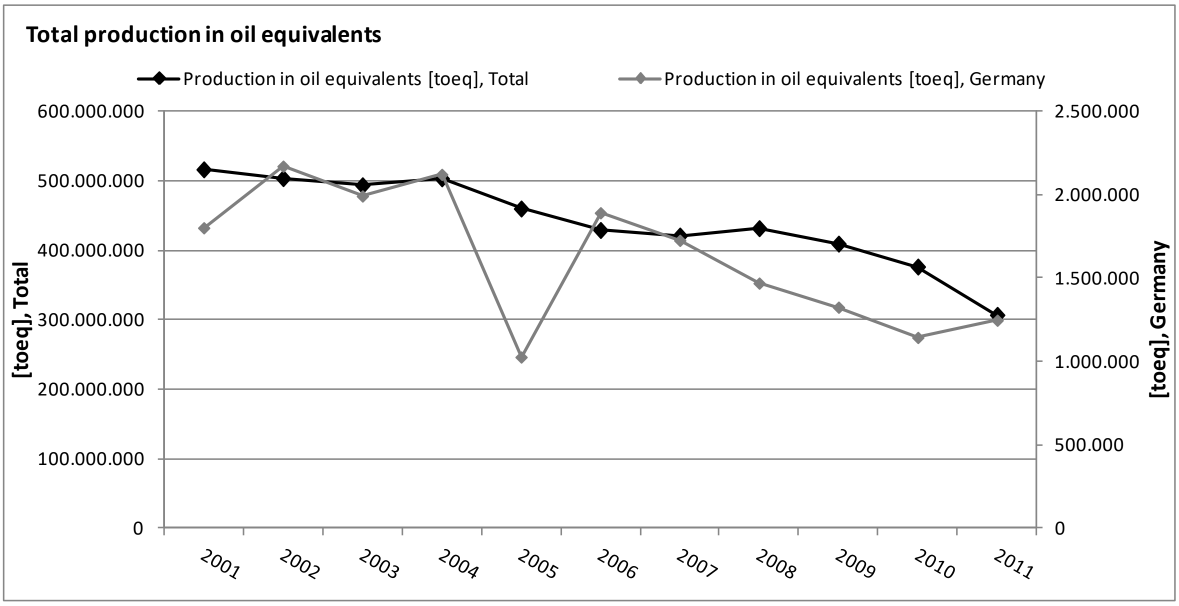


Figure 5-13: Oil production in the OSPAR region and Germany (Mittelplate) (OSPAR Offshore Industry Series, 2011).

[58] communicatd by E-Mail on November 12, 2013

In Figure 5-15 the yearly (2001-2010) amount of used and discharged chemicals is shown. The discharge ratio is more or less constant.

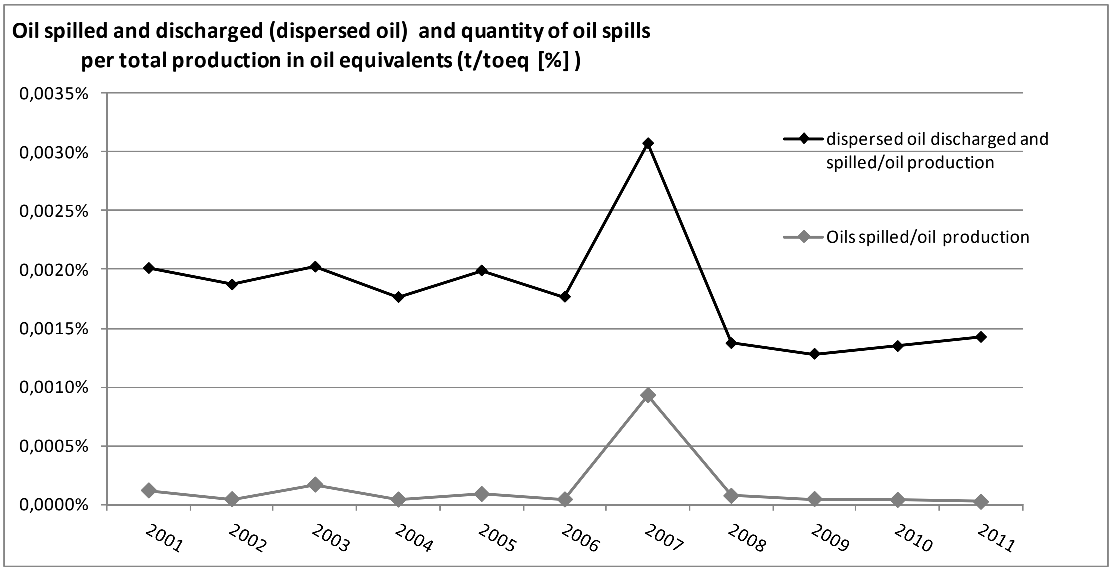


Figure 5-14: Ratios of spilled oil and of the sum of discharged and spilled oil per oil production (t/toeq) (OSPAR Offshore Industry Series, 2011).

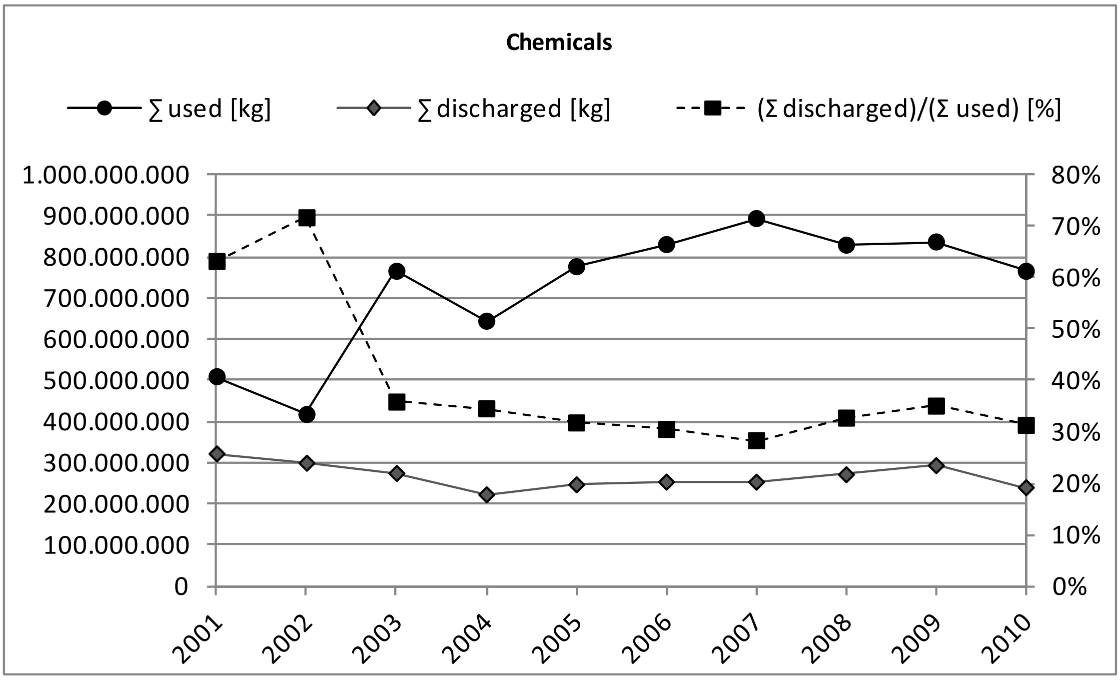


Figure 5-15: Amount of used and discharged chemicals (2001-2010) (OSPAR Offshore Industry Series, 2011)[59].

[59] http://www.ospar.org/v_publications/browse.asp?menu=01330305830000_000000_000000

The Plonor chemicals make up the majority of chemicals used (see Figure 5-16).

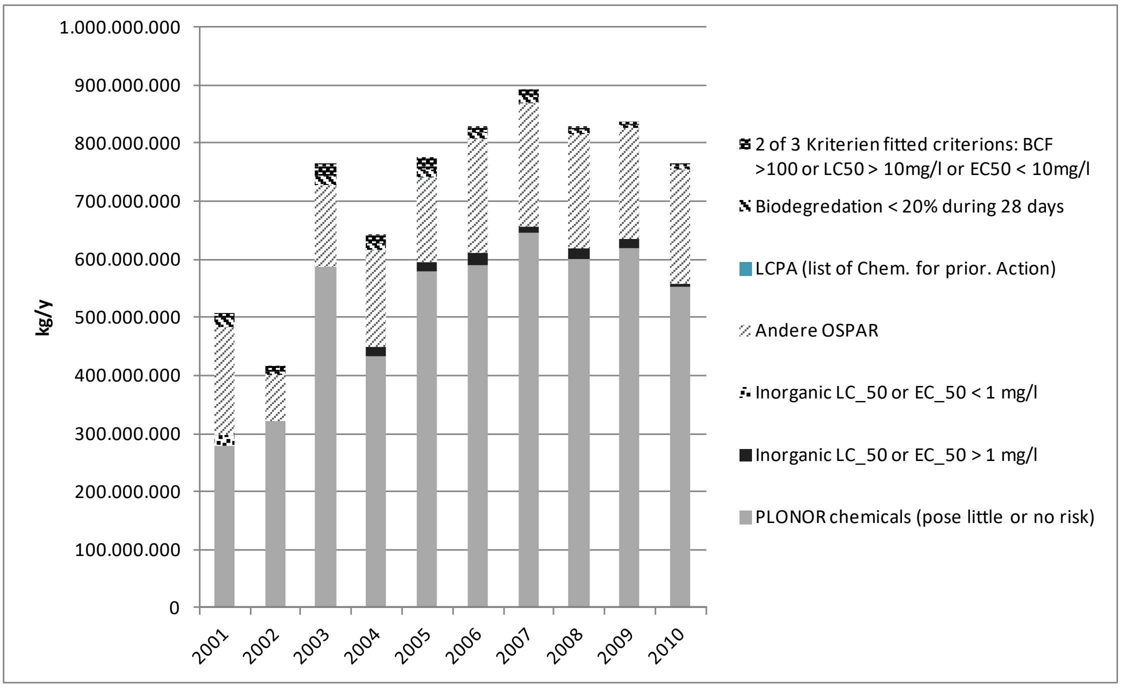


Figure 5-16: Composition of used chemicals (2001-2010) (OSPAR Offshore Industry Series, 2011)

Figure 5-17 shows the composition of discharged chemicals and the specific ratios of discharge. The discharge of LCPA Chemicals declined from 2001 to 2010.

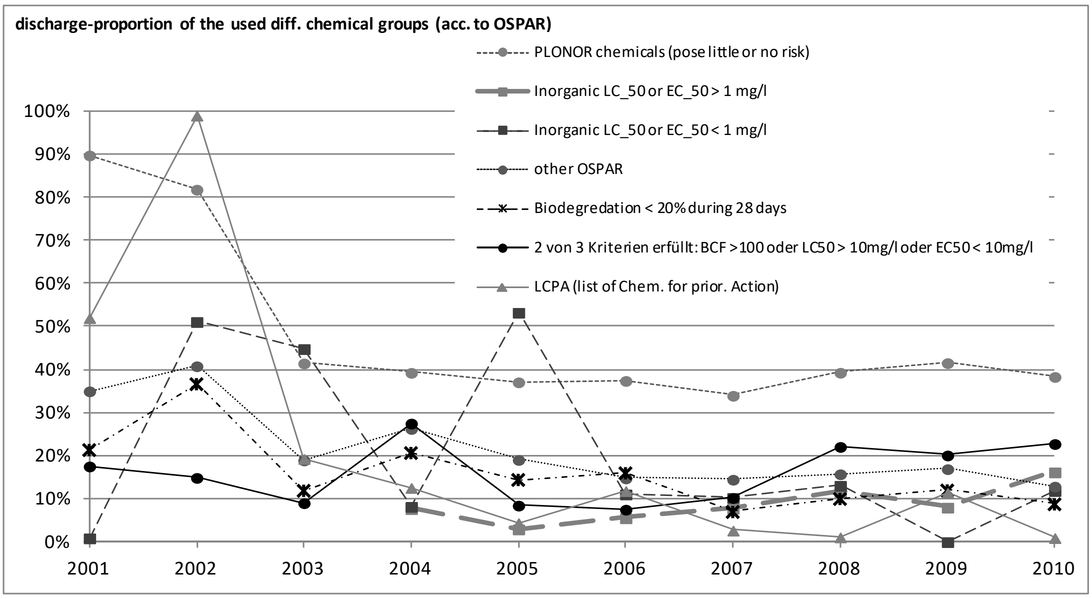


Figure 5-17: Discharge ratios of the used chemicals; (OSPAR Offshore Industry Series, 2011).

The most critical substances are summarized in the the OSPAR list of chemicals for priority action (LCPA).

The amount of substances of LCPA is relatively small between 1000 kg/y and 5000 kg/y and the discharge declined over the past decade (see Figure 5-16).

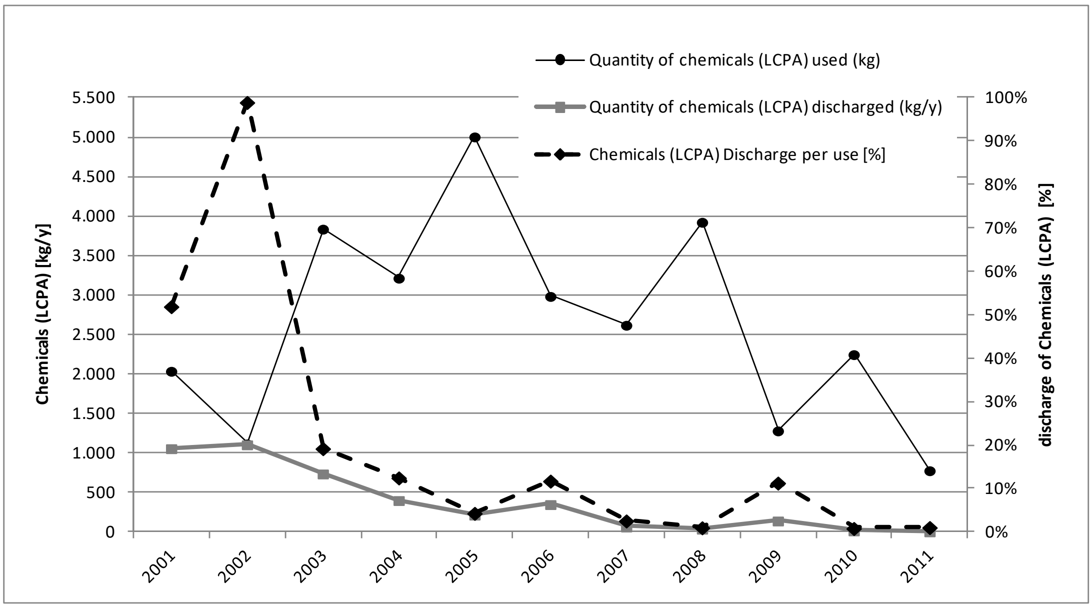


Figure 5-18: Used and discharged of LCPA [kg/y] (OSPAR Offshore Industry Series, 2011).

The LCPA (substances listed in the OSPAR List of Chemicals for priority action) account for about 10 ppm (average) of the total emission. That describes an amount of 21 kg in the year 2010 in the OSPAR region.

Emissions to air

The quantity of relevant air pollution according to OSPAR "Offshore Oil & Gas industry" is shown in Figure 5-19.

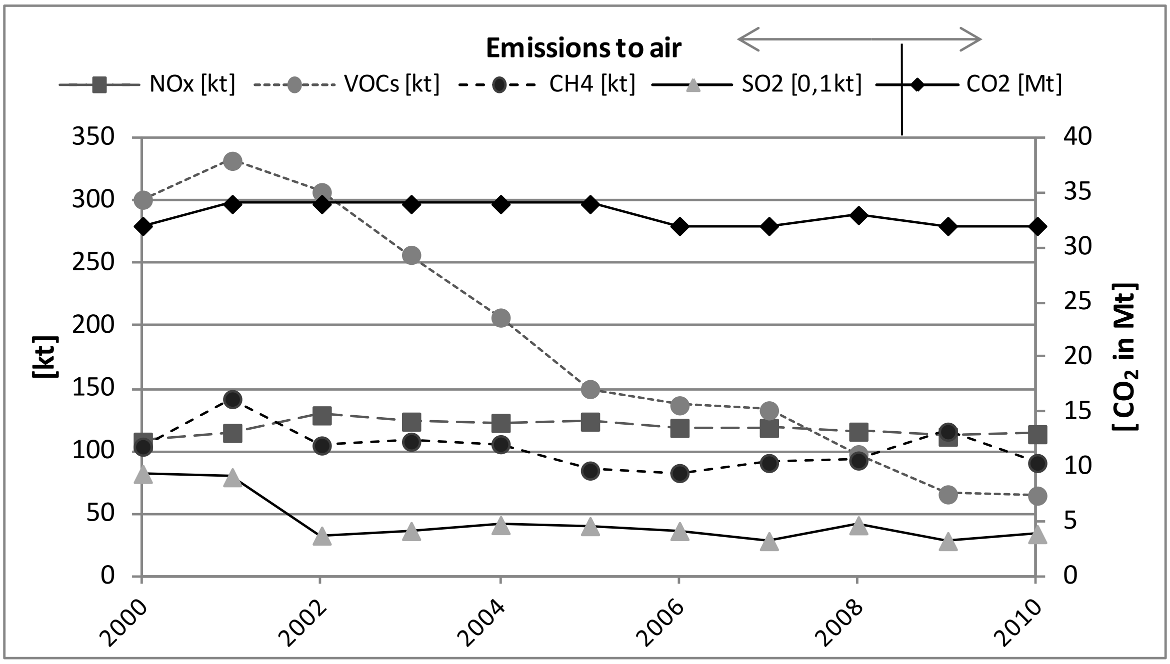


Figure 5-19: Emissions to air; OSPAR 2010 „Offshore Oil & Gas Industry".

The declining curve of volatile organic compounds (VOC) is remarkable. According to the Norwegian report there was a substantial reduction over the last years due to nmVOC recovery requirements on tankers.

The figures show that there have been efforts since the millennium, to reduce the emissions of HS.

Munition

Munition can be found in every ocean in the world. Sea dumping of chemical and conventional weapons and ammunition took place in the North and Baltic Seas especially after World War II up to the mid-1970s. The munition shells may break open during dumping, if not, corrosion processes may lead to toxic agents leaking out of over time. There is a lack of reliable information on what types of weapons are dumped and where they are lying.

According to HELCOM[60] about 40,000 tonnes of chemical munitions containing some 15,000 tonnes of chemical warfare agents were dumped into the Baltic Sea after World War II. Within the OSPAR maritime area 151 dumping sites and an increasing number of encounters with munition are documented[61].

The main hazardous substances mentioned which are discharged by leakage are organo-arsenic agents, mustard gas (sulphur and nitrogen) and organo-phosphorus.

[60] http://helcom.fi/baltic-sea-trends/hazardous-substances/sea-dumped-chemical-munitions

[61] OSPAR assessment sheet „Encounters with Chemical and Conventional Munitions" (2013-1)

5.1.8 Conclusions

The sheer presence of a multitude of substances in the marine environment, and other remote environments like the arctic, is a large reason for concern. These substances, as shown by their sheer presence, are persistent and mobile enough to reach remote environments as final sinks. It is virtually impossible to remove these substances once they are emitted to the environment, so whatever their effect on the ecosystem – chronic effects, as cocktail of chemicals or together with other stressors – it is irreversible. The risk associated with chemical pollution is characterised by large uncertainties regarding the probability and the extent of damage. Give the large risk, the complexity of the chemical pollution problem and the knowledge gaps should not be used as an excuse for not taking action.

More than 500 substances are potentially relevant for the marine environment. This includes substances that are in focus of WFD but not of OSPAR or HELCOM. Due to the spatial overlap of WFD and MSFD, these substances have to be monitored in coastal waters. These substances are summarized in Table 5-19.

Table 5-19: WFD substances, not yet in focus of OSPAR or HELCOM → Category "C", thus mandatory for coastal water.

<u>Pesticides</u>:

Alachlor, Atrazine, Chlorfenvinphos, Chlorpyrifos , Cyclodiene pesticides (Aldrien, Dieldrin, Endrin, Isodrin), DDT and para DDT, Diuron, Isoproturon, Pentachlorobenzol, Simazin, Quinoxyfen, Aclonifen, Bifenox, Cybutryn (Substitute for TBT), Cypermethrin, Dichlorvos, Heptachlor/-epoxid, Terbutryn

<u>Industrial Chemicals</u>:

Benzene (Benzol), Carbon Tetrachloride, Dichloroethan (hier: 1,2-Dichloroethan = Ethylendichlorid), Dichloromethane, Tetrachlorethylene, Trichlorethylen (Trichlorethen), Trichlorbenzole (Trichlorbenzene), Trichlormethan

<u>Pharmaceutical</u>:

17-β-Estradiol (watch list)

Given the multitude of substances, this project takes a pragmatic approach, clustering the substances based on their use pattern and compiling the arguments for inclusion or exclusion of a substance for monitoring under the MSFD from the available data.

To recap, there are large uncertainties. While in theory, it is simple to find the most harmful substances, in practice, the question is very complex as reliable values for PNEC and PEC in the marine environment are not available.

On the ecotoxicological side (PNEC) the abundance of possible target species and ecotoxicological effects contributes to the complexity, let alone the effect of chemical cocktails (concerted effects of chemicals) and the cross effect with other stressors, such as climate change and eutrophication.

On the load side (PEC), the situation is similarily complex. The concentration in the environment may be subject to considerable changes in time and space (hot spots and hot times of emission).

For the marine environment, which represents a final sink, the amount of a substance X that reaches the marine environment depends on two aspects. Firstly, on the persistence and the mobility of the substance in the ecosphere (transport and environmental fate processes). Secondly, on the source pattern, technical barriers and existing regulation in the anthroposphere, as laid out in the introduction.

The conclusions on the functional groups are compiled below, ranking the substances with high priority, second priority and lower priority for the marine environment based on the assessment criteria compiled in the fact sheets. The red star denotes substances that are highly regulated and can be expected to show a decreasing trend.

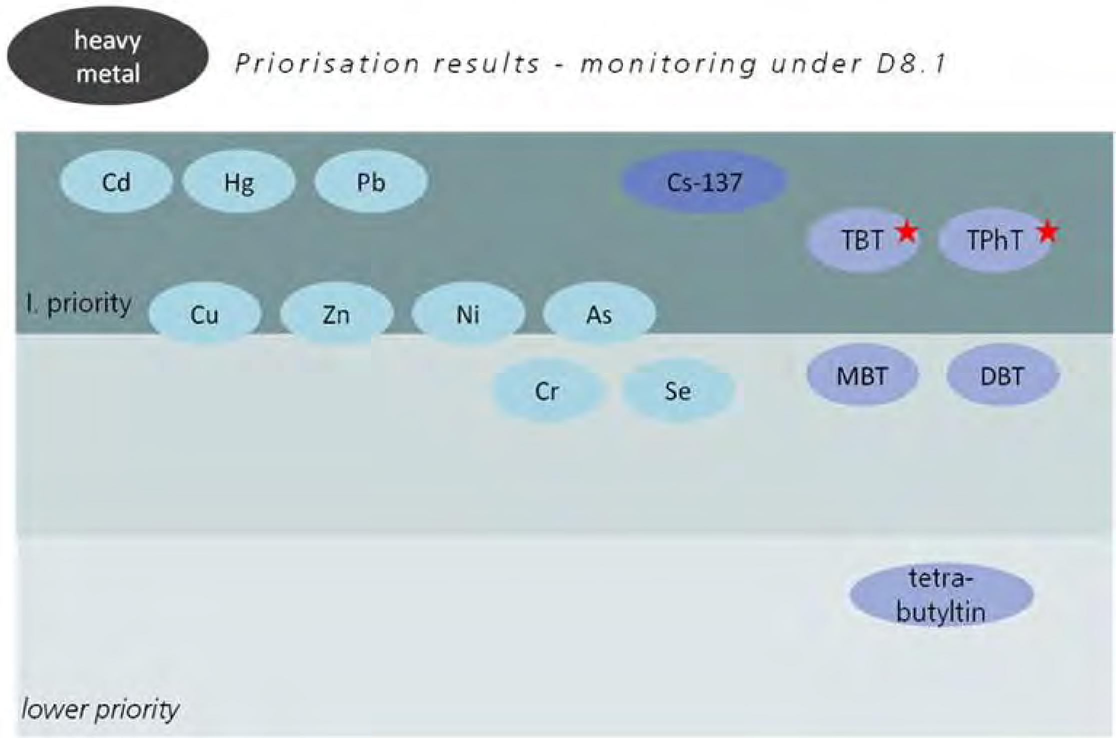


Figure 5-20: Substances proposed for monitoring under D8 – “heavy metals”.

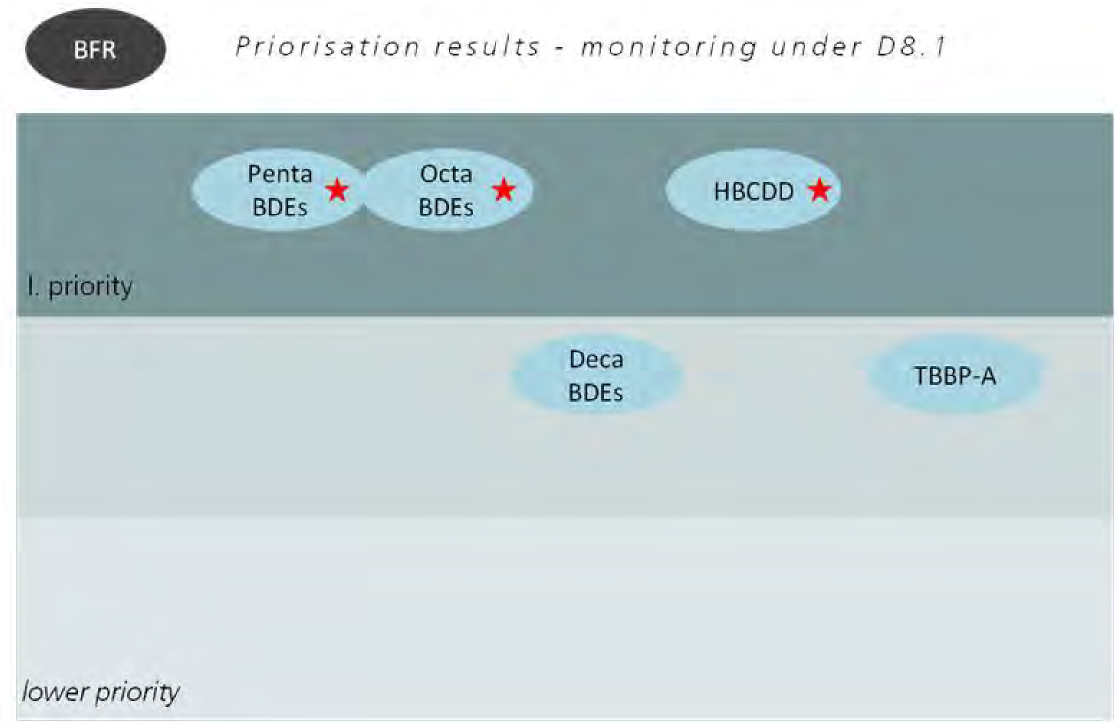


Figure 5-21: Substances proposed for monitoring under D8 – "BFR".

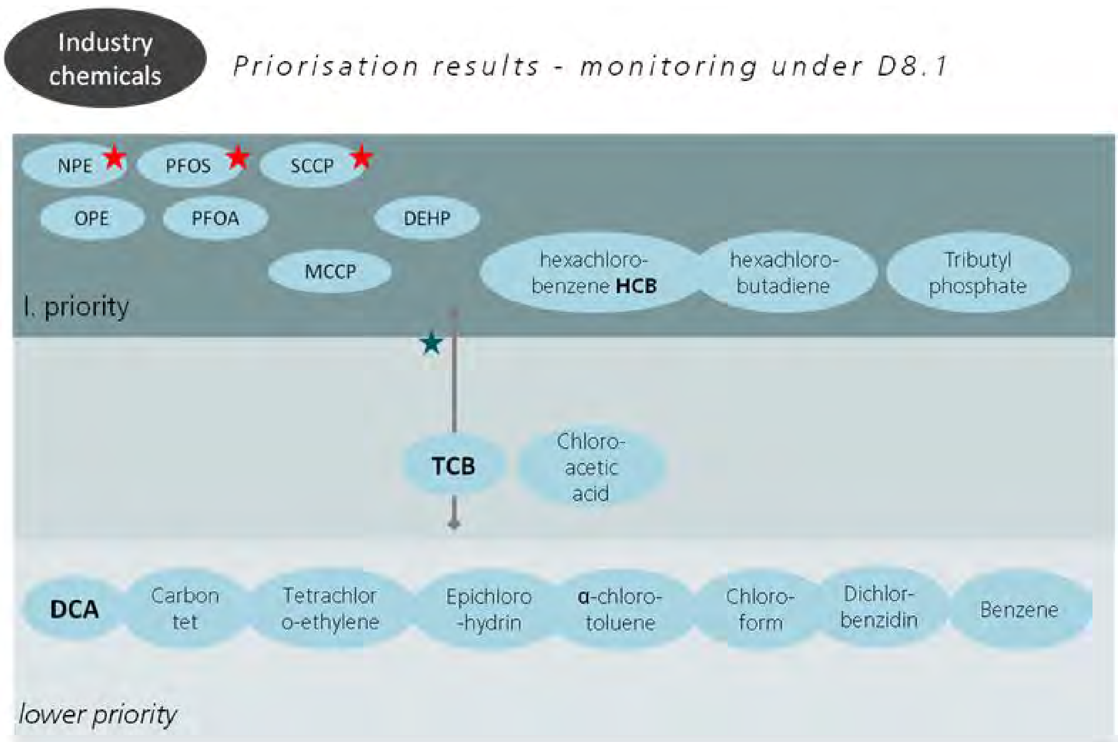


Figure 5-22: Substances proposed for monitoring under D8 – "Industry chemicals" Note: TCB - Volatilization from water surfaces is expected to be an important fate process, but may be attenuated by adsorption to suspended solids and sediment in the water column. OSPAR cessation target 2020 will not be reached for TCB.

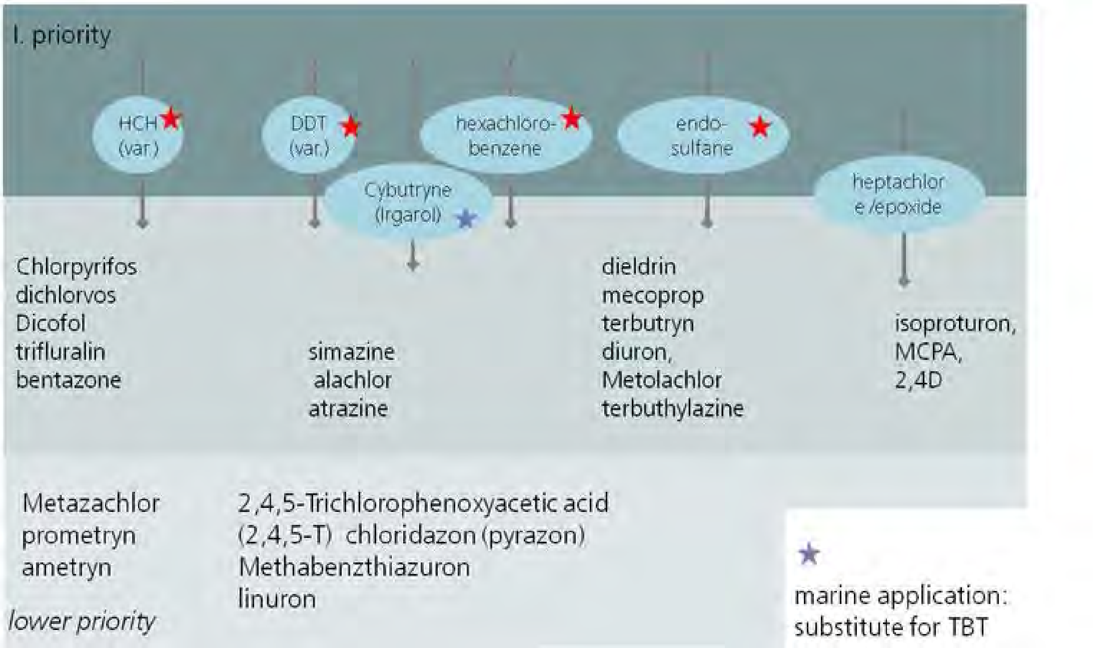


Figure 5-23: Substances proposed for monitoring under D8 – “Pesticides”.

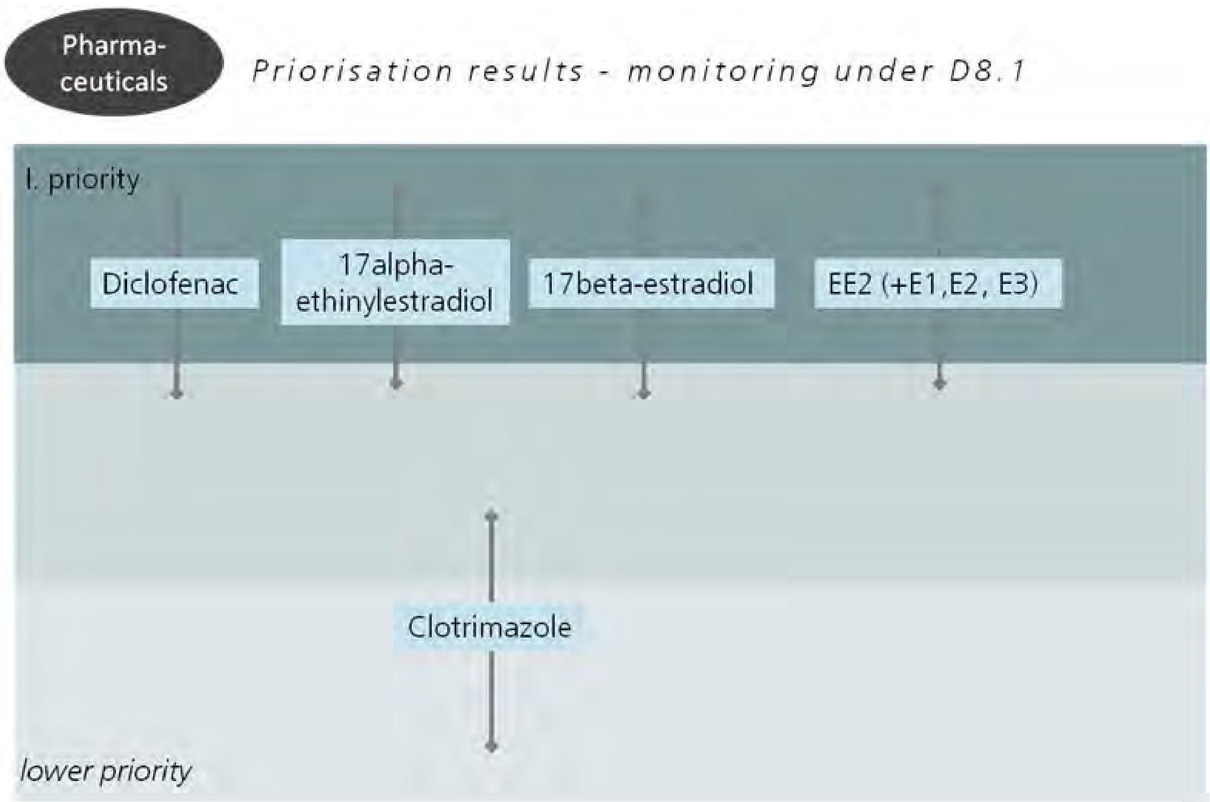


Figure 5-24: Substances proposed for monitoring under D8 – “Pharmaceuticals”.

For substances denoted with the red star, which are highly regulated and can be expected to show a decreasing trend, it is an option to run a sort of “fade out monitoring” with lower number of samples (time intervall, spatial coverage, only biota monitoring for top predators). These substances may be excluded from regular monitoring under the MFSD D8.1, given the limited resources for monitoring and the lack of possible actions against the emission of these substances (as they are already very highly regulated). This “fade out monitoring” as an alternative to regular monitoring under the MFSD D8.1 can use synergies with the arctic monitoring and CLTRAP.

These alternative approaches are proposed as a base for discussion, rather than as recommendation. For the organotins (TBT, TPhT) the monitoring of the biological effects (imposex on snails) under D8.2 (see the following section) may be sufficient, as the link is well established.

For dioxins, monitoring only in biota (top predators, using potential synergies with D9) and monitoring atmospheric deposition (using potential synergies with CLTRAP) is an alternative approach.

While these alternative approaches would free resources for other substances it has to be stressed that the high regulation is due to the very harmful characteristics of these substances. Generally spoken, the most notorious hazardous substances are the most highly regulated. Very often, these substances are still found in environmental samples years and decades after the regulation. But from their presence, it is not possible to find the sources and pathways or appropriate measures.

The question discussed in the present report is: Which substances are most relevant for the marine environment? With a wider perspective, the question to answer for each substance of relevance would be: At what point of the network of sources and pathways can this substance best be regulated and monitored? The answer has to be based on a full picture of the sources and pathway pattern as shown in Figure 5-1 and Figure 5-2, taking into account the dynamics illustrated in

Figure 5-3. These compilation leads to an integrated approach to management of hazardous substances while the regulation and monitoring landscape is rather fragmented at present. Looking at the WFD and MSFD, the question is whether the MSFD is the continuation of WFD (as they already overlap in coastal waters) or complementary to it.

The ranking of the groups, taking into account the highest priority substance of each group, gives the following Figure 5-25. The highest priority groups are heavy metals and organotins, brominated flame retardents (BFR), Polycyclic aromatic hydrocarbons (PAH) and the large group of industry chemicals with Polyflourated Chemicals PFC, Chloralkanes and Phenols; followed by dioxins and Pesticides; followed by pharmaceuticals.

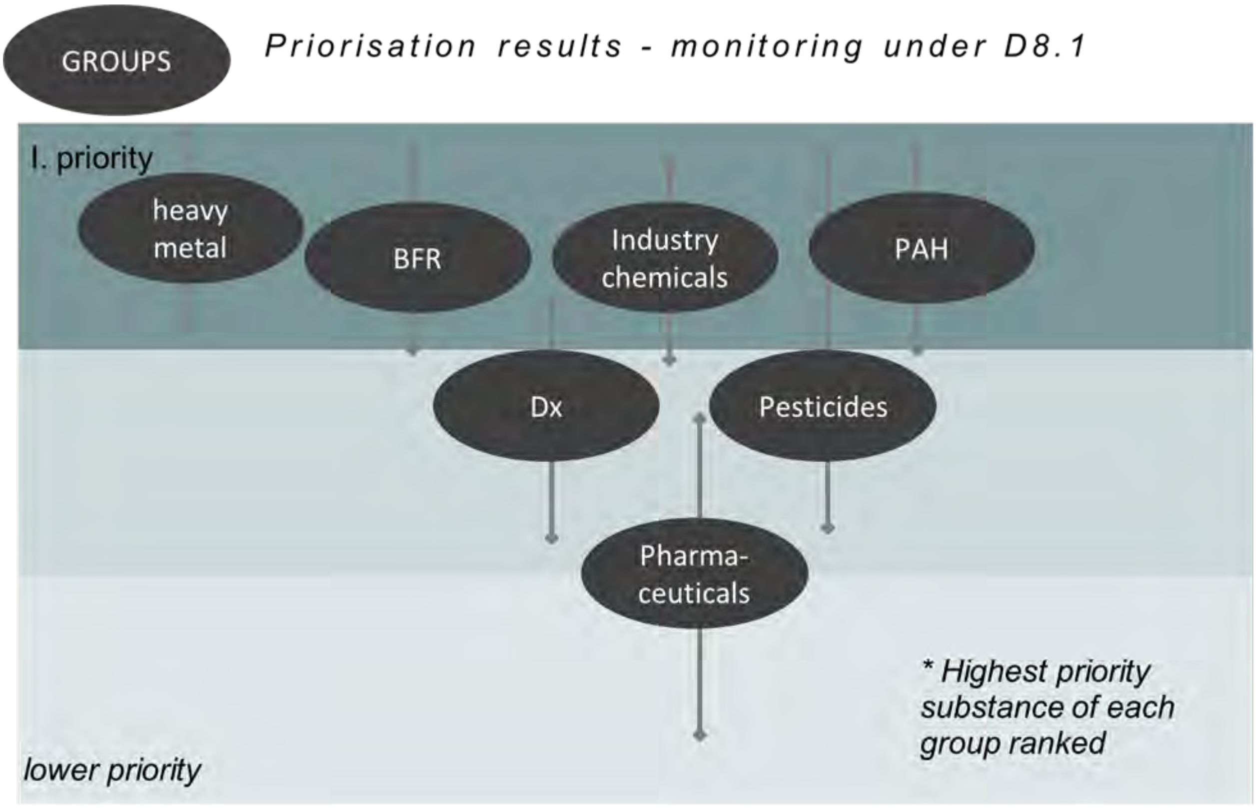


Figure 5-25: Substances proposed for monitoring under D8 – “Overview groups”.

The OSPAR list of chemicals for priority action includes chemicals that are directly discharged into the sea through offshore activities. Oil spills and discharge of produced water are reported by the abutted countries. As PAHs are natural components of tar, coal and oil (oil contains 0.2 - 7% PAH), produced water from offshore installations and other direct discharge is a direct pathway to the sea. PAHs and alkyl phenols (NP/OP) are discharged together with oil hazardous substances such as heavy metals, via produced water. Drilling fluids contain traces of heavy metals such as lead.

Beyond this there are more sources for direct emission into the sea like TBT, still used for antifouling paint of ships, aquacultures as a source of veterinary pharmaceuticals or munition dumped into the sea in large amounts which is an uncertain source of contaminants and difficult to calculate.

Monitoring and assessment are fundamental to a sound policy. Science-based assessment and policy ideally should act in an iterative way. Monitoring and assessment must be strictly implemented and at the same time - where new findings are available - be evaluated through feedback mechanisms that allow timely modification. The ESF Marine Board[62] calls this mechanism the “plan-do-assess-revise cycle” (see Figure 5-26).

Within the CHASE tool each substance is assessed against a threshold level and the results of the substances are then combined to obtain the status for each of the BSAP ecological objectives (concentrations of hazardous substances close to natural levels, all fish safe to eat, healthy wildlife, radioactivity at pre-Chernobyl-level).

[62] http://www.esf.org/marineboard

As a result of the calculation mode of the CR (dividing by the squareroot) the envrionmental status is not absolutely dependent on the number of indicators, but the cumulative effects of several smaller stressors are taken into account.

The CHASE tool gives each of these objectives a status (bad, poor, moderate, good, high). Supplemented by the OSPAR Assessment criteria, geographically structured and based on a confidential database, the CHASE tool in its expanded form as CHASE 2.0 seems to be a good and relatively easily applicable assessment tool.

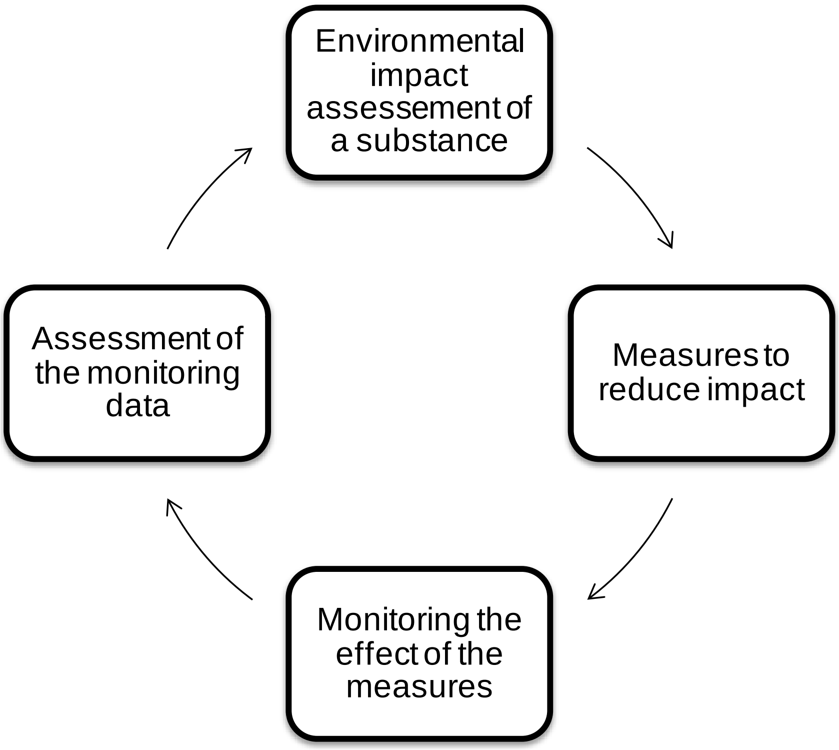


Figure 5-26: Monitoring as iterative process: „plan-do-assess-revice cycle“ (source: ESF 2011).

In any case, the differences between the North Sea and the Baltic Sea should be considered. While the North Sea is open to the Atlantic Ocean, the Baltic Sea is a brackish sea with little connection to the ocean. The water exchange is accordingly very low; the water is renewed on average only every 25 to 35 years. The Baltic Sea is therefore vulnerable and sensitive to all kinds of pollution to a greater extent.

5.2 Environmental quality targets with particular attention to river-basin-specific pollutants

In this project substances considered significant in the meaning of the MSRL and the “Water Framework Directive (WFD)” were identified. The selection process is explained in the chapter above and substances are listed in Table 5-20.

Additionally, relevant substances of the list of Priority Substances under the WFD (2000/60/EC) and the subsequent “Priority Substances Directive” (Directive 2008/105/EC), both recently updated in Directive 2013/39/EU, and the OGewV were selected by the German Environment Agency. The substances are listed in the Table 5-21 and Table 5-22.

Table 5-23 lists substances currently assessed in a project of the German Environment Agency (UBA project FKZ 3712 28 232 (Wenzel et al., 2014): “Derivation of proposals for EQS”). The proposed Environmental Quality Standards (EQS) for the marine environment derived so far in the project are shown in Table 5-28 and provide additional information regarding the protection of the marine ecosystem.

Table 5-20: Relevant substances identified in the framework of the project

Substance (n=28)	CAS-Nr.		Substance	CAS-Nr.
17-β-Estradiol	50-28-2		Dichloromethane	75-09-2
17-alpha-Ethinylestradiol	57-63-6		Dichlorvos	62-73-7
Aclonifen	74070-46-5		Diuron	330-54-1
Alachlor	15972-60-8		Heptachlor/-epoxid	1024-57-3
Atrazine	1912-24-9		Hexachlorocyclohexan	608-73-1
Benzene (Benzol)	71-43-2		Ibuprofen	15687-27-1
Bifenox	42576-02-3		Isoproturon	34123-59-6
Chlorfenvinphos	470-90-6		Pentachlorobenzol	608-93-5
Chlorpyrifos (Chlorpyrifos-ethyl)	2921-88-2		Pentachlorphenol	87-86-5
Cyanid (freies Cyanid)	57-12-5		Quinoxyfen	124495-18-7
Cybutryn	28159-98-0		Simazin	122-34-9
Cypermethrin	52315-07-8		Terbutryn	886-50-0
DEHP Bis(2-ethyl-hexyl) phthalat	117-81-7		Trichlormethan	67-66-3
1,2-Dichloroethan	107-06-2		Trifluralin	1582-09-8

Table 5-21: Substances, included in Annex 5 of OGewV 2011 and proposed to remain in Annex 5

Substance (n=12)	CAS-Nr.		Substance	CAS-Nr.
Anilin	62-53-3		Picolinafen	137641-05-5
Bromoxynil	1689-84-5		Pirimicarb	23103-98-2
Diazinon	333-41-5		Propiconazol	60207-90-1
Diflufenican	83164-33-4		Selen	7782492
Epoxiconazol	133855988		Silber	7440-22-4
Metribuzin	21087-64-9		Thallium	7440280

Table 5-22: Substances of Annex 5 of the OGewV (2011) previously listed in VO-WRRL (2005) Annex 4.

Substance (n=19)	CAS-Nr.		Substance	CAS-Nr.
Azinphos-ethyl	2642719		Chlortoluron	15545-48-9
Azinphos-methyl	86-50-0		Etrimphos	38260547
Dichlorvos	62-73-7		Hexazinon	51235042
Fenitrothion	122-14-5		Metazachlor	67129082

Fenthion	55389		Methabenzthiazuron	18691979
Malathion	121755		Metolachlor	51218452
Parathion-ethyl	56382		Prometryn	7287196
Parathion-methyl	298000		Terbuthylazin	5915-41-3
Ametryn	834128		Nitrobenzol	98953
Bromacil	314409			

Table 5-23: UBA project FKZ 3712 28 232 (Wenzel et al., 2014). Derivation of proposals for EQS.

Substance (n=30)	CAS-Nr.		Substance	CAS-Nr.
2,4-D	94-75-7		Phoxim	14816-18-3
Arsenic	7440-38-2		Roxythromycin	80214-83-1
Carbamazepin	298-46-4		Sulcotrion	99105-77-8
Carbendazim	10605-21-7		Tetrabutyltin	1461-25-2
Chloroacetic acid	79-11-8		Triclosan	3380-34-5
Dibutyltin cation	14488-53-0		Metoprolol	37350-58-6
Dimethoate	60-51-5		Bezafibrat	41859-67-0
Dimoxystrobin	149961-52-4		Erythromycin	114-07-8
Epichlorhydrin	106-89-8		Chrom	7440-47-3
Flufenacet	142459-58-3		Copper, Copper II sulphate pentahy-drate, Copper(i)oxide, Copper(ii)oxide, Dicopper chloride trihydroxide	7440-50-8, 7758-98-7, 1317-3-1, 1317–38–0, 1332-65-6
Flurtamone	96525-23-4		Zink	7440-66-0
Imidacloprid	105827-78-9		Uranium	7440-61-1
Monolinuron	1746-81-2		Sulfamethoxazole	723-46-6
Nicosulfuron	111991-09-4		Monobutylzinn	78763-54-9
Omethoat	1113-02-6		Phosphorsäure-triphenylester	115-86-6

According to EC (2010) quality standards for the substances considered significant for the marine environment are required. To support the derivation of environ¬mental quality standards (EQS) for priority substances and for river-basin-specific pollutants that need to be regulated under the WFD the “Technical Guidance for Deriving Environmental Quality Standards” (Guidance Document No. 27, 2011) has been developed. An approach for deriving EQS for the marine environment is included in this approach. Depending on the quantity of data and the taxonomic groups the uncertainty varies accordingly and different assessment factors (AF) are applied to obtain an EQS. The assessment scheme and the respective foot notes shown in Table 3.3 of the Guidance document are presented below in Table 5-24. The AFs (assessment factors) for the protection of saltwater organisms are generally higher than the AFs used for the protection of freshwater organisms reflecting the larger uncertainty especially with regard to the general underrepresentation of specific marine taxa in the dataset and possibly greater species diversity in the marine ecosystem compared to the freshwater ecosystem.

Concerning the quality standards (QS) required for the assessment under the MSRL the question came up whether the approach of the Guidance Document No. 27 can be followed. The procedure of the Guidance Document No. 27 should be used for deriving EQS values for the selected substances and the suitability of this approach for deriving quality standards under the MSRL should be discussed.

Table 5-24: Assessment factors to be applied to aquatic toxicity data for deriving a $QS_{SW, eco}$ (Table 3.3 of the Guidance Document No. 27, 2011).

Data set	Assessment factor
Lowest short-term L(E)C50 from freshwater or saltwater representatives of three taxonomic groups (algae, crustaceans and fish i.e. base set) of three trophic levels	10,000 [a)]
Lowest short-term L(E)C50 from freshwater or saltwater representatives of three taxonomic groups (algae, crustaceans and fish) of three trophic levels, plus two additional marine taxonomic groups (e.g. echinoderms, molluscs)	1000 [b)]
One long-term result (e.g. EC10 or NOEC) (from freshwater or saltwater crustacean reproduction or fish growth studies)	1000 [b)]
Two long-term results (e.g. EC10 or NOEC) from freshwater or saltwater species representing two trophic levels (algae and/or crustaceans and/or fish)	500 [c)]
Lowest long-term results (e.g. EC10 or NOEC) from three freshwater or saltwater species (normally algae and/or crustaceans and/or fish) representing three trophic levels	100 [d)]
Two long-term results (e.g. EC10 or NOEC) from freshwater or saltwater species representing two trophic levels (algae and/or crustaceans and/or fish) plus one long-term result from an additional marine taxonomic group (e.g. echinoderms, molluscs)	50
Lowest long-term results (e.g. EC10 or NOEC) from three freshwater or saltwater species (normally algae and/or crustaceans and/or fish) representing three trophic levels + two long-term results from additional marine taxonomic groups (e.g. echinoderms, molluscs)	10 [e)]

Notes:

General note: Evidence for varying the assessment factor should in general include a consideration of the availability of data from a wider selection of species covering additional feeding strategies/ life forms/ taxonomic groups other than those represented by the algal, crustacean and fish species (such as echinoderms or molluscs). This is especially the case, where data are available for additional taxonomic groups representative of marine species.

More specific recommendations with regard to issues to consider in relation to the data available and the size and variation of the assessment factor are indicated below.

When there are indications that a substance may cause adverse effects via disruption of the endocrine system of mammals, birds, aquatic or other wildlife species, it should be considered whether the assessment factor would also be sufficient to protect against effects caused by such a mode of action, or whether an increase of the factor would be appropriate.

a) The use of a factor of 10,000 on short-term toxicity data is a conservative and protective factor and is designed to ensure that substances with the potential to cause adverse effects are identified. It assumes that uncertainties identified above make a significant contribution to the overall uncertainty. For any given substance there may be evidence that this is not so, or that one particular component of the uncertainty is more important than any other. In these circumstances it may be necessary to vary this factor.

This variation may lead to a raised or lowered assessment factor depending on the evidence available. Except for substances with intermittent release, as defined in ECHA (2008), under no circumstances should a factor lower than 1000 be used in deriving a QSsw, eco from short-term toxicity data. Evidence for varying the assessment factor could include one or more of the following:

- evidence from structurally similar compounds which may demonstrate that a higher or lower factor may be appropriate.

- knowledge of the mode of action as some substances by virtue of their structure may be known to act in a non-specific manner. A lower factor may therefore be considered. Equally a known specific mode of action may lead to a higher factor.

- the availability of data from a variety of species covering the taxonomic groups of species across at least three trophic levels. In such a case the assessment factors may only be lowered if multiple data points are available for the most sensitive taxonomic group (i.e. the group showing acute toxicity more than 10 times lower than for the other groups). Variation from an assessment factor of 10,000 should be fully reported with accompanying evidence.

b) An assessment factor of 1000 is applied where data from a wider selection of species are available covering additional taxonomic groups (such as echinoderms or molluscs) other than those represented by algal, crustacean and fish species; if data are at least available for two additional taxonomic groups representative of marine species.

An assessment factor of 1000 is applied to a single long-term result (e.g. EC10 or NOEC) (freshwater or saltwater crustacean or fish) if this result was generated for the taxonomic group showing the lowest L(E)C50 in the short-term algal, crustacean or fish tests.

If the only available long-term result (e.g. EC10 or NOEC) is from a species which does not have the lowest L(E)C50 in the short-term tests, applying an assessment factor of 1000 is not regarded as protective of other more sensitive species. Thus, the hazard assessment is based on the short-term data with an assessment factor of 10,000 applied. However, normally the lowest QSsw, eco should prevail.

An assessment factor of 1000 can also be applied to the lowest of the two long-term results (e.g. EC10 or NOEC) covering two trophic levels (freshwater or saltwater algae and/or crustacean and/or fish) when such results (e.g. EC10 or NOEC) have not been generated for the species showing the lowest L(E)C50 of the short-term tests.

This should not apply in cases where the acutely most sensitive species has an L(E)C50-value lower than the lowest long-term value. In such cases the QSsw, eco might be derived by applying an assessment factor of 1000 to the lowest L(E)C50 of the short-term tests.

c) An assessment factor of 500 applies to the lowest of two long term results (e.g. EC10 or NOEC) covering two trophic levels (freshwater or saltwater algae and/or crustacean and/or fish) when such results have been generated covering those trophic levels showing the lowest L(E)C50 in the short-term tests with these species. Consideration can be given to lowering this factor in the following circumstances:

- It may sometimes be possible to determine with a high probability that the most sensitive species covering fish, crustacea and algae has been examined, that is that a further longer-term result (e.g. EC10 or NOEC) from a third taxonomic group would not be lower than the data already available. In such circumstances an assessment factor of 100 would be justified;

- a reduced assessment factor (to 100 if only one short-term test, to 50 if two short-term tests on marine species are available) applied to the lowest long-term result (e.g. EC10 or NOEC) from only two species may be appropriate where:

- short-term tests for additional species representing marine taxonomic groups (for example echinoderms or molluscs) have been carried out and indicate that these are not the most sensitive group, and;

- it has been determined with a high probability that long-term results (e.g. EC10 or NOEC) generated for these marine groups would not be lower than that already obtained. This is particularly important if the substance does not have the potential to bioaccumulate.

An assessment factor of 500 also applies to the lowest of three long term results (e.g. EC10 or NOEC) covering three trophic levels, when such results have not been generated from the taxonomic group showing the lowest L(E)C50 in short-term tests. This should, however, not apply in the case where the acutely most sensitive species has an L(E)C50 value lower than the lowest long-term result (e.g. EC10 or NOEC) value. In such cases the QSsw, eco might be derived by applying an assessment factor of 1000 to the lowest L(E)C50 in the short-term tests.

d) An assessment factor of 100 will be applied when longer-term toxicity results (e.g. EC10 or NOEC) are available from three freshwater or saltwater species (algae, crustaceans and fish) across three trophic levels. The assessment factor may be reduced to a minimum of 10 in the following situations:

- where short-term tests for additional species representing marine taxonomic groups (for example echinoderms or molluscs) have been carried out and indicate that these are not the most sensitive group, and it has been determined with a high probability that long-term results (e.g. EC10 or NOEC) generated for these species would not be lower than that already obtained;

- where short-term tests for additional taxonomic groups (for example echinoderms or molluscs) have indicated that one of these is the most sensitive group acutely and a long-term test has been carried out for that species. This will only apply when it has been determined with a high probability that additional long-term results (e.g. EC10 or NOEC) generated from other taxa will not be lower than the long-term results already available.

e) A factor of 10 cannot be decreased on the basis of laboratory studies only. It may be permitted if justified by mesocosm or field data.

It is important to note that a thorough verification and updating of the existing EQS or the derivation of new EQS would be a time- and labour-consuming task due to the large number of the selected substances (n=59). Therefore, it was not possible to perform this work in the framework of the presented project.

However, to check the procedure described in the Technical Guidance Document 27 (2011) and to obtain a rough estimate of saltwater EQS values for the selected substances a limited data search was performed as described below.

- The comprehensive ECOTOX Database of U.S. EPA was used, search modus “Simple search” (ECOTOX, 2013).
  As stated at the home page of the database “ECOTOX is a comprehensive database, which provides information on adverse effects of single chemical stressors to ecologically relevant aquatic and terrestrial species. ECOTOX includes more than 400,000 test records covering 5,900 aquatic and terrestrial species and 8,400 chemicals. The primary source of ECOTOX data is the peer-reviewed literature, with test results identified through comprehensive searches of the open literature. All pertinent information on the species, chemical, test methods, and results presented by the authors are abstracted into the ECOTOX database. ECOTOX also includes third-party data collections from the EPA, U.S. Geological Survey, Russia, and OECD (Organization for Economic Cooperation and Development) member nations summarizing research that is either published in non-English journals or not available in the open literature.”

Due to the large data pool the distribution of substance specific data regarding taxonomic groups is considered to be representative for an overall literature search.

- Substance search was performed using the CAS number.
- It was checked for the individual substances Substance specific test data were collected for "aquatic animals" and "aquatic plants" and the effect parameters behaviour, ecosystem, growth, mortality, physiology, population und reproduction were used.
- Validity of the cited studies was not verified.
- Data were separated in fresh- und saltwater data.
- Data were filtered according to the relevant endpoints:
  LC_{10}, EC_{10}, NOEC und NOEL for long-term effects
  LC_{50}, EC_{50} for short-term effects.
- whether the acute basic dataset consisting of data for algae, invertebrates (i.e crustaceans) and fish is available for freshwater and/or for saltwater organisms.
- It was checked whether at least one long-term value for crustaceans or fish is available.
- The test durations were reviewed in order to ensure the differentiation of acute and chronic tests.
- A long-term $QS_{\text{SW eco marine}}$ was derived using only the saltwater data according to Table 3.3 of the Guidance Document No. 27.
- A long-term $QS_{\text{SW eco pooled}}$ was derived using the pooled freshwater and saltwater data according to Table 3.3 of the Guidance Document No. 27. An evaluation with freshwater data only was not performed.

The original data of the database search are not included in the report due to the large amount; they are available from the ETOX database of the German Environment Agency. The relevant original data and the derivation of the QS are published in the UBA Texte 47/2015 (Wenzel et al., 2015). The complete report of the project (Wenzel et al. 2014) is also made publicly available.

The outcome of the evaluation for the different substance groups is summarised in the tables below.

The general results are:

- The current data situation for the selected substances regarding saltwater organisms is generally poor.
- Therefore, a reliable assessment and derivation of EQS values based on marine data only is not possible (except a few substances, e.g. copper).
- Mostly, neither a complete acute base set for algae, crustaceans and fish nor a NOEC of at least one long-term study is available for saltwater organisms.
- Within the pooled data from fresh- and saltwater a freshwater species is the most sensitive. However, this is regarded a result of the larger database for freshwater organisms and the higher number of taxonomic groups. Therefore, the probability to identify a very sensitive organism is higher compared to the limited marine data set.
- Due to the limited data set for marine organisms for most of the substances and the lack of additional taxonomic groups specific for the marine environment there is a high uncertainty deriving EQS only using marine data.

Therefore, the procedure of the Guidance Document 27 (2011) for using pooled data (if the requirements for combining data are fulfilled, i.e. data not statistically different, please refer to chapter 3.2.3 of the Guidance Document 27) and applying higher assessment factors is considered correspondingly applicable for the MSRL, unless a large amount of marine data is available.

Conclusion regarding the derivation of quality standards (QS) under the MSRL: The QS for the selected substances considered significant for the marine environment has to be derived based on a thorough data search and study verification. The procedure of the Guidance Document 27 (2011) for using combined fresh- and saltwater data (if the requirements for combining data are fulfilled and applying higher assessment factors is considered correspondingly applicable for the MSRL, unless a large amount of marine data for a substance is available.

Table 5-25: Relevant substances identified in the framework of the project.

Substance	CAS-Nr.	Pooled freshwater (FW) and marine (SW) data				Marine data		
		$QS_{SW,eco\ pooled}$ [µg/L]	AF	Lowest short/long term result	Used dataset	$QS_{SW,eco\ marine}$ [µg/L]	AF	Used dataset
17-β-Estradiol	50-28-2	0.0000107	100	FW Fish	Three long-term results from three trophic levels	0.000032	500	Two long -term results from two trophic levels
17-alpha-Ethinylestradiol	57-63-6	0.0000008	50	FW Fish	Two long-term results + One long-term result from additional marine taxonomic group	0.000002	50	Two long-term results + One long-term result from additional marine taxo-nomic group
Aclonifen	74070-46-5	no data				no data		
Alachlor	15972-60-8	0.0035	100	FW Algae	Three long-term results from three trophic levels	0.22	500	Two long -term results from two trophic levels
Atrazine	1912-24-9	0.05	10	FW Fish	Three long-term results from three trophic levels + Two long-term results from two additional marine taxonomic groups	0.35	10	Three long-term results from three trophic levels + Two long-term results from two additional marine taxonomic groups
Benzene (Benzol)	71-43-2	0.374	10000	SW Crustacean	Three short-term results from three trophic levels	0.374	10000	Three short-term results from three trophic levels
Bifenox	42576-02-3	No three short-terms of three trophic levels				No three short-terms of three trophic levels		
Chlorfenvinphos	470-90-6	No three short-terms of three trophic levels				No three short-terms of three trophic levels		
Chlorpyrifos (Chlorpyrifos-ethyl)	2921-88-2	0.001	10	FW Crustacean	Three long-term results from three trophic levels + Two long-term results from two additional marine taxonomic groups	0.006	10	Three long-term results from three trophic levels + Two long-term results from two additional marine taxonomic groups
Cyanid (freies Cyanid)	57-12-5	0.058	500	SW Crustacean	Two long -term results from two trophic levels	0.058	500	Two long -term results from two trophic levels
Cybutryn	28159-98-0	0.00001	50	FW Algae	Two long-term results + One long-term result from additional marine taxonomic group	0.0002	50	Two long-term results + One long-term result from additional marine taxo-nomic group
Cypermethrin	52315-07-8	0.000041	100	SW Crustacean	Three long-term results from three trophic levels	0.000041	100	Three long-term results from three trophic levels
DEHP Bis(2-ethyl-hexyl) phthalat	117-81-7	0.154	500	FW Crustacean	Two long -term results from two trophic levels	1	1000	Three short-term results from three trophic levels + Two additional marine taxonomic groups

Substance	**CAS-Nr.**	**Pooled freshwater (FW) and marine (SW) data**				**Marine data**		
		$QS_{SW,eco\ pooled}$ [µg/L]	**AF**	**Lowest short/long term result**	**Used dataset**	**$QS_{SW,eco\ marine}$ [µg/L]**	**AF**	**Used dataset**
1,2-Dichloroethan	107-06-2	29	1000	FW Fish	One long-term result	0.69279	10000	Three short-term results from three trophic levels
Dichloromethane	75-09-2	1.23	10000	SW Crustacean	Three short-term results from three trophic levels	1.23	10000	Three short-term results from three trophic levels
Dichlorvos	62-73-7	0.0148	100	SW Crustacean	Three long-term results from three trophic levels	0.00148	1000	One long-term result
Diuron	330-54-1	0.0042	50	SW Algae	Two long-term results + One long-term result from additional marine taxonomic group	0.0042	50	Two long-term results + One long-term result from additional marine taxo-nomic group
Heptachlor/-epoxid	1024-57-3	0.000004	10000	SW Crustacean	Three short-term results from three trophic levels	No three short-terms of three trophic levels		
Hexachlorocyclohexan	608-73-1	0.0004	1000	SW Crustacean	One long-term result	0.0004	1000	One long-term result
Ibuprofen	15687-27-1	0.001	100	FW Fish	Three long-term results from three trophic levels	No three short-terms of three trophic levels		
Isoproturon	34123-59-6	No three short-terms of three trophic levels				No three short-terms of three trophic levels		
Pentachlorobenzol	608-93-5	0.019	500	SW Fish	Two long -term results from two trophic levels	0.019	500	Two long -term results from two trophic levels
Pentachlorphenol	87-86-5	0.5	10	FW Algae	Three long-term results from three trophic levels + Two long-term results from two additional marine taxonomic groups	0.0044	1000	Three short-term results from three trophic levels + Two additional marine taxonomic groups
Quinoxyfen	124495-18-7	0.01272	500	FW Algae	Two long -term results from two trophic levels	0.0743	1000	Three short-term results from three trophic levels + Two additional marine taxonomic groups
Simazin	122-34-9	0.3	100	FW Algae	Three long-term results from three trophic levels	0.6	1000	Three short-term results from three trophic levels + Two additional marine taxonomic groups
Terbutryn	886-50-0	0.27348	500	FW Algae	Two long -term results from two trophic levels	0.00031	10000	Three short-term results from three trophic levels
Trichlormethan	67-66-3	12.4	100	FW Fish	Three long-term results from three trophic levels	2.8	10000	Three short-term results from three trophic levels

Substance	CAS-Nr.	Pooled freshwater (FW) and marine (SW) data				Marine data		
		$QS_{SW,eco\ pooled}$ [µg/L]	AF	Lowest short/long term result	Used dataset	$QS_{SW,eco\ marine}$ [µg/L]	AF	Used dataset
Trifluralin	1582-09-8	0.028	500	SW Crustacean	Two long -term results from two trophic levels	0.028	500	Two long -term results from two trophic levels

Table 5-26: Substances, included in Annex 5 of OGewV 2011 and proposed to remain in Annex 5.

Substance	CAS-Nr.	Pooled freshwater and marine data				Marine data		
		$QS_{SW,eco\ pooled}$ [µg/L]	AF	Lowest short/long term result	Used dataset	$QS_{SW,eco\ marine}$ [µg/L]	AF	Used dataset
Anilin	62-53-3	0.04	100	FW Fish	Three long-term results from three trophic levels	No three short-terms of three trophic levels		
Bromoxynil	1689-84-5	6.02	500	FW Fish	Two long -term results from two trophic lev-els	No three short-terms of three trophic levels		
Diazinon	333-41-5	0.0000001	50	FW Crustacean	Two long-term results + One long-term result from additional marine taxonomic group	0.042	50	Two long-term results + One long-term re-sult from additional marine taxonomic group
Diflufenican	83164-33-4	no data				no data		
Epoxiconazol	133855988	No three short-terms of three trophic levels				No three short-terms of three trophic levels		
Metribuzin	21087-64-9	0.00466	500	FW Algae	Two long -term results from two trophic lev-els	0.0116	500	Two long -term results from two trophic levels
Phenanthren	85-01-8	0.1	50	FW Fish	Two long-term results + One long-term result from additional marine taxonomic group	1.7823	50	Two long-term results + One long-term re-sult from additional marine taxonomic group
Picolinafen	137641-05-5	no data				no data		
Pirimicarb	23103-98-2	0.00065	10000	FW Crustacean	Three short-term results from three trophic levels	No three short-terms of three trophic levels		
Propiconazol	60207-90-1	1	500	FW Fish	Two long -term results from two trophic lev-els	0.0021	10000	Three short-term results from three trophic levels
Selen	7782492	0.08	500	FW Fish	Two long -term results from two trophic lev-els	No three short-terms of three trophic levels		

Substance	CAS-Nr.	Pooled freshwater and marine data				Marine data		
		$QS_{SW,eco\ pooled}$ [µg/L]	AF	Lowest short/long term result	Used dataset	$QS_{SW,eco\ marine}$ [µg/L]	AF	Used dataset
Silber	7440-22-4	0.33	1000	SW Crustacean	One long-term result	0.33	1000	One long-term result
Thallium	7440280	0.0012	10000	FW Crustacean	Three short-term results from three trophic levels	No three short-terms of three trophic levels		

Table 5-27: Substances of Annex 5 of the OGewV (2011), previously listed in VO-WRRL (2005) Annex 4.

Substance	CAS-Nr.	Pooled freshwater and marine data				Marine data		
		$QS_{SW,eco\ pooled}$ [µg/L]	AF	Lowest short/long term result	Used dataset	$QS_{SW,eco\ marine}$ [µg/L]	AF	Used dataset
Azinphos-ethyl	2642719	No three short-terms of three trophic levels				No three short-terms of three trophic levels		
Azinphos-methyl	86-50-0	0.0002	100	SW Crustacean	Three long-term results from three trophic levels	0.0002	100	Three long-term results from three trophic levels
Dichlorvos	62-73-7	0.0148	100	SW Crustacean	Three long-term results from three trophic levels	0.00148	1000	One long-term result
Fenitrothion	122-14-5	0.00009	100	FW Crustacean	Three long-term results from three trophic levels	0.000087	1000	One long-term result
Fenthion	55389	0.000074	500	SW Crustacean	Two long -term results from two trophic levels	0.000074	500	Two long -term results from two trophic levels
Malathion	121755	0.006	100	SW Crustacean	Three long-term results	0.00012	500	Two long -term results from two trophic levels
Parathion-ethyl	56382	0.00004	50	FW Crustacean	Two long-term results + One long-term result from additional marine taxonomic group	0.6	50	Two long-term results + One long-term result from additional marine taxonomic group
Parathion-methyl	298000	0.00022	500	SW Crustacean	Two long -term results from two trophic levels	0.00022	500	Two long -term results from two trophic levels
Ametryn	834128	0.003	1000	FW Algae	Three short-term results from three trophic levels + Two additional marine taxonomic groups	0.01	1000	Three short-term results from three trophic levels + Two additional marine taxonomic groups
Bromacil	314409	0.00068	10000	FW Algae	Three short-term results from three trophic levels	0.688	10000	Three short-term results from three trophic levels
Chlortoluron	15545-48-9	No three short-terms of three trophic levels				No three short-terms of three trophic levels		
Etrimphos	38260547	0.0001	1000	FW Crustacean	One long-term result	No three short-terms of three trophic levels		
Hexazinon	51235042	0.035	100	FW Algae	Three long-term results from three trophic levels	No three short-terms of three trophic levels		
Metazachlor	67129082	No three short-terms of three trophic levels				No three short-terms of three trophic levels		
Methabenzthiazuron	18691979	No three short-terms of three trophic levels				No three short-terms of three trophic levels		
Metolachlor	51218452	0.0014	500	FW Algae	Two long -term results from two trophic levels	0.0034	500	Two long -term results from two trophic levels
Prometryn	7287196	0.003	100	FW Algae	Three long-term results from three trophic levels	0.0222	100	Three long-term results from three trophic levels

Substance	CAS-Nr.	Pooled freshwater and marine data				Marine data		
		$QS_{SW,eco\ pooled}$ [µg/L]	AF	Lowest short/long term result	Used dataset	$QS_{SW,eco\ marine}$ [µg/L]	AF	Used dataset
Terbuthylazin	5915-41-3	0.00032	10000	FW Algae	Three short-term results from three trophic lev-els	No three short-terms of three trophic levels		
Nitrobenzol	98953	5.2	500	FW Crustacean	Two long -term results from two trophic levels	0.668	10000	Three short-term results from three trophic levels

Table 5-28: UBA project FKZ 3712 28 232 (Wenzel et al. 2014). Derivation of proposals for EQS.

Substance Name	CAS_No	Marine Data	AA-EQS sw
2,4-D	94-75-7	Short-term result one taxonomic group	0.02
Arsenic	7440-38-2	Short-term results from three taxonomic groups, long-term results from two taxonomic groups	1.3
Bezafibrate	41859-67-0	no	0.23
Carbamazepin	298-46-4	Long-term results from two taxonomic groups	0.05
Carbendazim	10605-21-7	Short-term results from three taxonomic groups	0.015
Chloroacetic acid	79-11-8	no	0.058
Chromium	7440-47-3	Short-term results from four taxonomic groups	0.6
Copper	7440-50-8	Short-term results from at least three taxonomic groups, long-term results from eight taxonomic groups	0.7
Dibutyltin cation	14488-53-0	Short-term results from two taxonomic groups and long-term results from two taxonomic groups	0.2
Dimethoate	60-51-5	Short-term results from three taxonomic groups	0.007
Dimoxystrobin	149961-52-4	no	0.003
Epichlorhydrin	106-89-8	Short-term result from one taxonomic group	1.1
Erythromycin	114-07-8	no	0.02
Flufenacet	142459-58-3	Short-term results from three taxonomic groups and long-term results from two taxonomic groups	0.004
Flurtamone	96525-23-4	no	0.023
Imidacloprid	105827-78-9	Short-term results from two taxonomic groups	0.00024
Metoprolol	37350-58-6	no	4.3
Monobytyltin cation*	78763-54-9	Short-term result from two taxonomic groups, long-term results from three taxonomic groups	0.00056/0.075
Monolinuron	1746-81-2	no	0.015
Nicosulfuron	111991-09-4	Short-term result from one taxonomic group	0.00087
Omethoate	1113-02-6	Short-term result from two taxonomic groups	0.00042
Phosphorsäure-triphe-nylester	115-86-6	Short-term result from two taxonomic groups	0.37
Phoxim	14816-18-3	Short-term result from three taxonomic groups	0.000074
Roxythromycin	80214-83-1	no	0.0047
Sulcotrion	99105-77-8	Short-term result from one taxonomic group	0.01
Sulfamethoxazole	723-46-6	no	0.06
Tetrabutyltin	1461-25-2	Short-term result from one taxonomic group and long-term result from one taxonomic group	0.014
Triclosan	3380-34-5	Short-term result from three taxonomic groups and long-term result from one taxonomic group	0.002
Uranium	7440-61-1	no	0.042
Zink (dissolved)	7440-66-0	Short-term results from two taxonomic groups, long-term results from six taxonomic groups	3.0

AA-EQS sw = Annual Average concentration of the Environmental Quality Standard (long-term EQS).

* due to the lack of reliable date the derivation of a final, reliable QS is not found useful at present. The shown values are covering a concentration range.

5.3 Biological Effects - Bioindicators

5.3.1 Introduction

The monitoring of biological effects in the North Sea and Baltic Sea is an issue of major concern for the new EU marine strategy framework directive (MSFD). Descriptor 8 requires that concentrations of contaminants are at levels not giving rise to pollution effects." Therefore, biological indicators are valuable tools to identify and verify the adverse environmental effects of contaminants and to attribute the changes to specific contaminants.

A literature review was conducted to identify available bioindicators that could provide a warning of the environmental impact. Biomarkers, are already integrated in international monitoring programmes such as OSPAR/CEMP; and HELCOM/COMBINE (OSPAR 2008; HELCOM 2007) were listed and further promising bioindicators suggested and discussed with regard to their potential use in the monitoring of the addressed contaminants.

5.3.2 Biological effects and biomarker

The selected pollutants relevant for monitoring under the MSFD were examined with respect to their biological effects (see Chapter 5.1). For these substances, information on their modes of action (MoA) related to similar physiological responses in exposed aquatic organisms were collected and the substances were grouped according to their MoA (

In the following tables the information on the substances were taken from databases (e.g. IRAC http://www.irac-online.org/modes-of-action/; PAN Pesticides Database - Chemicals http://www.pesticideinfo.org/Search_Chemicals.jsp) reports (e.g. Wenzel et al., 2014) and commonly used textbooks and manuals.

Table 5-33).

Biomarkers are used to indicate an exposure to substances (biomarker of exposure) or indicate a hazardous effect of substances on organisms (biomarker of effect). Biomarkers can give information on the health status of an ecosystem. However, if biological effects (biomarker response) are to be attributed to the presence of specific pollutants in the environment, knowledge on the MoA of the substances of concern is essential.

A broad range of biomarkers (see Table 5-30, Table 5-31, Table 5-32, Chapter 5.3.2.1) is available. The marine conservation conventions HELCOM (CORESET), OSPAR (Common indicators) define a range of biomarkers which have partly been used as part of the integrated monitoring of marine ecosystems or might be used in the future.

The selected biomarkers were grouped as biomarkers of exposure (Table 5-29) or as biomarkers of effect, ranked according to their level of biological organisation: molecular/cellular (Table 5-30), organ/organism (Table 5-31) and population (Table 5-32). As a further step it was tried to link the molecular/cellular and the organ/organism biomarkers (hereinafter called "specific biomarker") to specific MoA (

Table 5-34).

The attribution of defined biological effects to the presence of specific pollutants in the environment was difficult or even impossible due to the low compound specificity of most specific biomarkers. Still, few specific biomarkers were identified that can be used for an integrated assessment of the status of the marine environment in relation to hazardous substances.

The substances selected above as potentially relevant for the marine environment are characterised by a broad range of MoAs and effects. Most of the effects can be detected by the set of biomarkers currently used in the OSPAR CEMP and HELCOM Combine. These biomarkers include biomarker of effects: molecular/cellular marker (EROD activity, micronuclei, lysosomal stability), organ/organism marker (macroscopic liver neoplasms, liver histophathology, externally visible signs of deseases and imposex) and population marker related to reproduction and are supplemented byexposure information (PAK metabolites in organisms). Biomarkers responding to substance induced effects are complemented by biomarkers of exposure, such as detection of PAK metabolites in organisms.

However, several substances of the selected potential marine pollutions within the project show specific MoAs which are not covered by the commonly used biomarker set, as described above.

Organophosphates and carbamates are characterised by AChE inhibition in animals which can be measured by the AChE test. Some of these compound groups are readily hydrolysed in the aquatic environment depending on the pH but may still pose a hazard in coastal waters.

Herbicides are mostly characterised by inhibition of photosynthesis and enter the aquatic environment following use on agricultural land. A biomarker reflecting this MoA (e.g. photosynthesis efficiency) has not been applied so far but might be considered useful in case of relevant contaminatin levels, especially in coastal waters.

In both cases, the use of additional biomarkers should be considered for monitoring of contamination in the marine environment.

The information provides a comprehensive overview of selected pollutants and their biological effects and will help to improve the use of biomarker as part of an integrated assessment of the marine ecosystem.

5.3.2.1 Biological indicators of contamination

The selected biomarkers were grouped as biomarkers of exposure (Table 5-29) or as biomarkers of effect, ranked according to their level of biological organisation: molecular/cellular (Table 5-30), organ/organism (Table 5-31) and population (Table 5-32). As a further step it was tried to link the molecular/cellular and the organ/organism biomarkers (hereinafter called “specific biomarker”) to specific MoA (

Table 5-34).

The tables describe the biological effects and the issues addressed (e.g. group of chemicals) by each biomarker as well as the key organisms they are applied to. Experiences and limitations of the individual biomarkers are described and key references are provided. The tables allow a comprehensive analysis of the bioindicators of contamination which have been used as part of monitoring programmes and alternative biomarkers which may be used in the future.

5.3.2.1.1 Biomarkers for exposure

Table 5-29: Exposure indicators and biomarker characteristics.

PAH metabolites in bile (indicator of exposure)	
Biological effects:	Measures (chemical analysis) exposure to and metabolism of PAHs
Issues addressed:	PAHs
Organism:	Fish (bile)

Experiences:	preCEMP
Literature:	JAMP (2008), Kammann 2007; Vuorinen et al. 2006
Alkylphenol-bile metabolites (indicator of exposure)	
Biological effects:	Measures (chemical analysis) exposure to and metabolism of alkylated phenols
Issues addressed:	Alkylated phenols
Organism:	Fish (Cod)
Experiences:	Requires further research before use in monitoring programmes
Literature:	Reed et al. 2011
YES (Yeast Estrogen Receptor Assay)	
Biological effects:	Reporter gene assay for (anti)estrogenic substances
Issues addressed:	Oestrogen receptor-active compounds
Organism:	Yeast
Experiences:	Commonly used in monitoring studies for estrogenic substances
Limitations:	Estrogenic activity in a sample
Literature:	Reed et al. 2011; Matsumoto et al. 2004
YAS (Yeast Androgen Receptor Assay)	
Biological effects:	Reporter gene assay for (anti)androgenic substances
Issues addressed:	Androgen receptor-active compounds
Organism:	Yeast
Experiences:	Commonly used in monitoring studies for androgenic substances
Limitations:	Androgenic activity in a sample
Literature:	Reed et al. 2011

5.3.2.1.2Biomarker for effects: Molecular and cellular level

Table 5-30: Molecular and cellular level biomarker characteristics.

Lysosomal membrane stability (LMS)	
Biological effects:	Damage of biological membranes; Impact on liver function. It will be most important to have an assessment of the disease status of each individual fish sampled. This biomarker provides useful supporting information if EROD measurements are being made.
Issues addressed:	Not contaminant specific but responds to a wide variety of xenobiotic contaminants and metals.
Organism:	Fish, mussels
Advantages:	Early detectable; computational modelling in addition to bio monitoring possible
Limitations:	Possible tolerance in fluctuating environment (estuaries)

Experiences:	Applied as first-tier screening parameter Monitoring programme Mediterranean Sea, UNEP 2007; preCEMP; CORESET
Literature:	JAMP (1997), Köhler et al. 2002; Moore and Lowe 2004;
Lipid content	
Biological effects:	Energy reserves, storage lipids
Issues addressed:	Not contaminant specific
Organism:	Fish, mussels
Advantages:	Early marker
Limitations:	Seasonal dependencies (dependent on food availability)
Literature:	Grote et al. 2011; Carro et al. 2012; Fraser 1989
Acid phosphatase activity of macrophage aggregates (M-ACT)	
Biological effects:	Immunotoxicity
Issues addressed:	Not contaminant specific
Organism:	Fish, mussels
Limitations:	Seasonal dependencies (dependent on food availability)
Literature:	Boeg and Lehtonen 2006: Reed et al. 2010
Micronuclei (MN)	
Biological effects:	Mutagenicity; exposure to aneugenic and clastogenic contaminants
Issues addressed:	Non contaminant specific;
Organism:	Fish, mussels
Limitations:	Requires further research
Experiences:	Has been used in monitoring programmes
Literature:	Barsiene et al. 2006
Glutathione-S-transferase (GST)	
Biological effects:	Measures expsoure and the capacity of the major group of phase II enzymes. Considered most promising for isoenzyme specific measurements
Issues addressed:	Predominantly organic xenobiotics; PAHs
Organism:	Fish, mussels
Limitations:	Requires further research before use in monitoring programs
Experiences:	BIOMAR programme
Literature:	Reed et al. 2011
Acetylcholinesterase; (AChE) and catalase enzyme activities (CAT, GR)	
Biological effects:	Measures exposure
Issues addressed:	Organophosphates and carbamates or similar molecules
Organism:	Fish, mussels, crustaceans
Limitations:	Temperature effects

Literature:	Livingstone et al. 1993; Bocquené and Galgani 1998;
Bulky DNA adduct formation	
Biological effects:	Measures genotoxic effects. Possible predictor of pathology through mechanistic links. Sensitive indicator of past and present exposure
Issues addressed:	PAHs, other synthetic organics, e.g. nitro-organics, amino triazine pesticides (triazines)
Organism:	Fsh, mussesl, invetebrates
Limitations:	Requires further research for optimised substance identification
Literature:	JAMP (2008), Reichert et al. 1999; Reed et al. 2010
Metallothione induction (MT)	
Biological effects:	Measures exposure and disturbance of copper and zinc metabolism. Measures induction of metallothionein protein
Issues addressed:	Cu, Zn, Cd and inorganic Hg
Organism:	Fish, mussels
Limitations:	Dependent on salinity
Experiences:	preCEMP
Literature:	JAMP (2008), Hylland 1999; George et al. 2004; Kammann et al. 1996
Ethoxyresorufin-O-deethylase activity (EROD) /Cyp 1A	
Biological effects:	Possible predictor of pathology through mechanistic links. Sensitive indicator of past and present exposure
Issues addressed:	Measures indiction of enzymes which metabolize planar organic contaminants (e.g. PAHs, planar PCBs, dioxins)
Organism:	Fish (liver)
Advantages:	Highly sensitive, one of the first detectable
Limitations:	Influenced by abiotic and biotic factors: temperature, pH, age, reproductive stage
Experiences:	Tested on more the 150 species of fish; used in monitoring programs: North Sea Task force, Mediterranean Pollution Network, French National Observation Network; preCEMP; COMBINE
Literature:	JAMP (2008), Stagg and McIntosh 1998; Kammann et al. 2008;
Oxidative stress	
Biological effects:	Measures the presence of free radicals: 1) superoxide dismutase (SOD) which converts O2- to H2O2; 2) catalase which converts H2O2 to water: 3) glutathione peroxidase (GPX) which also converts H2O2 to water; 4) glutathione reductase (GR) which maintains cellular reduced glutathione; 5) malonedialdehyde which is a measure of lipid peroxidation.
Issues addressed:	Not contaminants specific, will respond to a wide range of environmental contaminants
Organism:	Fish, invertebrates
Limitations:	Requires further research

Experiences:	BIOMAR programme
Literature:	JAMP (2008), Sheader et al., 2006
Aminolevulinic acid dehydrase (ALAD)	
Biological effects:	Index of exposure: Lead causes a dose-dependant inhibition of -aminolevulinic acid dehydratase (ALA-D) which is an enzyme essential for the synthesis of haemoglobin in the haemopoietic tissue. ALA-D inhibition is therefore a good indicator of Pb exposure which is maximally inhibited before other signs of Pb toxicity become apparent
Issues addressed:	Hg, Pb
Organism:	Fish
Experiences:	BEEP
Literature:	JAMP (2008), Reed et al. (2011), Dwyer et al. 1988; Johansson-Sjöbeck and Larsson 1978; Johansson-Sjöbeck and Larsson 1979; Schmitt et al. 1993.
Peroxisomal proliferation	
Biological effects:	Potential alteraations in lipid metabolism, non-genotoxic carcinogenesis
Issues addressed:	Contaminant specific
Organism:	Fish, invertebrates
Experiences:	Requires further research before use in monitoring programmes
Literature:	Reed et al. 2010
BaP hydroxylase-like enzymes	
Biological effects:	Measures exposure to organic contaminants
Issues addressed:	Induced enzyme response to PAHs, planar PCBs, dioxins and/or furans
Organism:	Invertebrates
Experiences:	Requires further research before use in monitoring programmes
Literature:	Reed et al. 2011
MDR/MXR	
Biological effects:	Induction/inhibition of multidrug (MDR)/multixenobiotic (MXR) resistance transporters in membranes; response to xenobiotic stress: disturbance of integrity of membrane
Issues addressed:	Multiple contaminants (organics and metals)
Organism:	Fish, mussels, invertebrates
Literature:	Bard, S.M. 2000; Achard et al. 2004

5.3.2.1.3Biomarker of effects: Organ and organism level

Table 5-31: Organ and organism level biomarker characteristics.

Liver histopathology	
Biological effects:	Histological changes in liver tissue; indicative of nonspecific and specific contaminant effects at cellular or tissue level
Issues addressed:	Carcinogenic and non-carcinogenic contaminants; PAHs
Organism:	Fish
Experiences:	CORESET; preCEMP
Literature:	JAMP (2008), Hinton, D.E. and Lauren, D.J. 1990; Feist et al. 2004
Macroscopic liver neoplasms	
Biological effects:	Indicative of contaminant associated liver carcinogenesis. Diagnosis of pathological changes and enzymatic markers of carcinogenesis associated with exposure to genotoxic and nongenotoxic carcinogenesis.
Issues addressed:	Effects of carcinogenic substances. PAHs, other synthetic organics, e.g. nitro-organics.
Organism:	Fish
Experiences:	preCemp; CORESET
Literature:	Bucke et al. 1996; Feist et al. 2004;
Vitellogenin induction	
Biological effects:	Measures feminization of male fish, indicator for reproductive impairment, by determining Vtg induction in fish by measuring Vtg protein concentrations in blood plasma
Issues addressed:	Oestrogenic substances / anti-androgen
Organism:	Adult and juvenile fish
Literature:	JAMP (2008), Scott and Hylland 2002;
Intersex, gonadal	
Biological effects:	Measures feminization of male fish and reproductive impairment
Issues addressed:	Oestrogenic substances
Organism:	Fish
Literature:	JAMP (2008), Oehlmann 2004
Imposex /Intersex TBT specific	
Biological effects:	Reproductive interference; direct indications of the effects of TBT at the individual organism level. Imposex (superimposition of penis and/or vas deferens on prosobranch females) and intersex condition (pathological alterations in the oviduct of littorinids and replacement of female by male organs)
Issues addressed:	Organotins, especially TBT
Organism:	Neogastropods, Mollusks

Advantages:	Sensitive biomarkers for the determination of the degree of environmental organotin and especially tributyltin (TBT) pollution
Experiences:	CEMP, COMBINE, CORESET
Literature:	JAMP (2008), Gibbs 1999; Strand and Jacobsen 2002;
Condition index	
Biological effects:	Influence on health and fitness
Issues addressed:	Not contaminant specific
Organism:	Fish
Literature:	Jakob et al. 1996
Visible diseases/abnormalities	
Biological effects:	Diseases incl. paracitosis
Issues addressed:	Not contaminant specific
Organism:	Fish
Limitations:	Influenced by other factors apart from contaminants
Literature:	Bucke et al. 1996
Behavioural responses	
Biological effects:	Feeding and swimming habits, sencory responses
Issues addressed:	Not contaminant specific
Organism:	Fish
Experiences:	Requires further research before use in monitoring programmes
Literature:	Shrivastava et al. 2010

5.3.2.1.4Biomarker of effects: Population level

Table 5-32: Population level biomarker characteristics.

Benthic community analysis	
Biological effects:	Ecosystem level, structure, species distribution
Issues addressed:	Responds to a wide variety of contaminants, particularly those resulting in organic enrichment
Organism:	Macro-, meio-, and epibenthos
Advantages:	Particularly useful for point sources. Most appropriate for deployment when other monitoring methods indicate that a problem may exist
Literature:	Reed et al. 2011
Whole sediment bioassays	
Biological effects:	General toxicity in a biotest with sediment samples
Issues addressed:	Not contaminant specific, will respond to a wide range of
Organism:	Sediment organisms, sediment-dwelling taxa

Advantages:	May enable retrospective interpretation of community changes
Literature:	Thain and Bifield 2001
Reproductive success in fish	
Biological effects:	Measures reproductive output and survival of eggs and fry in relation to contaminants.
Issues addressed:	Endocrine substances
Organism:	Fish
Experiences:	preCEMP, CORESET
Literature:	Gercken et al. 2006
Abnormalities in wild fish embryos and larvae	
Biological effects:	Measures frequency of probably lethal abnormalities in fish larvae.
Issues addressed:	Not linked unequivocally to contaminants
Organism:	Fish, including demersal and pelagic species
Experiences:	Requires further research before use in monitoring programmes
Literature:	Sundelin et al. 2008

In the following tables the information on the substances were taken from databases (e.g. IRAC http://www.irac-online.org/modes-of-action/; PAN Pesticides Database - Chemicals http://www.pesticideinfo.org/Search_Chemicals.jsp) reports (e.g. Wenzel et al., 2014) and commonly used textbooks and manuals.

Table 5-33: Mode of action of the selected substances.

Substance	CAS number	Product group / Use type	Substance group	Mode of action
1,2-Dichloroethane	107-06-2	Chemical	Chlorinated hydrocarbon	Exact mechanism not clear: carcinogenic, hepatic, renal, neurological effects u.a.. Possibly reactive intermediates formed by conjugation with glutathione
17-alpha-Ethinylestradiol	57-63-6	Hormone	Estrogen	ER agonist
17-β-Estradiol	50-28-2	Hormone	Estrogen	ER agonist
2,4-D, 2,4-(dichlorophenoxy) acetic acid	94-75-7	Herbicide	Phenoxy compound	Plant hormone, synthetic auxin = uncontrolled growth
Aclonifen	74070-46-5	Herbicide	Diphenylether	Carotenoid synthesis and protoporphyrinogen oxidase inhibitor (chlorophyll synthesis)

Substance	CAS number	Product group / Use type	Substance group	Mode of action
Alachlor	15972-60-8	Herbicide	Chloracetanilide	Elongase inhibition, and inhibition of geranylgeranyl pyrophosphate (GGPP) cyclisation enzymes, part of the gibberellin pathway, *PS Inhibitor*
Ametryn	834128	Herbicide	Triazine	PS II inhibitor
Anilin	62-53-3	Chemical	Aromatic amine	Exact mechanism not clear: Carcinogenic, after metabolic activation
Arsenic	7440-38-2	Metal	Metal	Thiol affinity, enzyme inhibition. Uncoupler of oxidative phosphorylation.
Atrazine	1912-24-9	Herbicide	Triazine	PS II inhibitor
Azinphos-ethyl	2642719	Insecticide	Organophosphate	AChE inhibitor
Azinphos-methyl	86-50-0	Insecticide	Organophosphate	AChE inhibitor
Benzene (Benzol)	71-43-2	Chemical	Hydrocarbon, aromatic	Carcinogen (DNA breaks?)
Bezafibrate	41859-67-0	Pharmaceutical, Human	Phenoxyacetates	Agonist of PPARα: peroxisome proliferator and hypolipidaemic agent
Bifenox	42576-02-3	Herbicide	Diphenylether	Protoporphyrinogen oxidase inhibitior (chlorophyll synthesis) *PS Inhibitor*
Bromacil	314409	Herbicide	Uracil	PS II inhibitor
Bromoxynil	1689-84-5	Herbicide, Metabolite	Nitrile herbicide	PSII inhibitor
Carbamazepine	298-46-4	Pharmaceutical	Dibenzazepine; Analgesics, Non-Narcotic	Sodium channel blocker
Carbendazim	10605-21-7	Fungicide, Metabolite soil	Benzimidazole	Mitosis and cell division inhibitor
Chlorfenvinphos	470-90-6	Insecticide	Organophosphate	AChE Inhibitor
Chloroacetic acid	79-11-8	Chemical	Organochlorine	Corrosive. MCA blocks the cell energy supply probably by decreasing the activity of pyruvate dehydrogenase and to some extent of ketoglutarate dehydrogenase
Chlorpyrifos (Chlorpyrifos-ethyl)	2921-88-2	Insecticide	Organophosphate	AChE Inhibitor
Chlortoluron	15545-48-9	Herbicide	Urea	PS II inhibitor

Substance	CAS number	Product group / Use type	Substance group	Mode of action
Chromium	7440-47-3	Metal	Metal	Cr (VI) cancerogenic. Cr (III): During the reduction process of Chromium (VI) to Chromium (III) in the body, along with reactive intermediates, chromium adducts with proteins, deoxyribonucleic acid (DNA), and secondary free radicals are also formed
Copper (2+): Copper II sulphate pentahydrate, Copper(i)oxide, Copper(ii)oxide, Dicopper chloride trihydroxide	7440-50-8, 7758-98-7, 1317-3-1, 1317–38–0, 1332-65-6	Metal Fungicide	Metal, heavy	Proteins, nonspecific denaturation (disruption) of cellular proteins (enzymes). Link to various chemical groups (imidazoles, phosphates, , sulfhydryls, hydroxyls groups.
Cyanid (freies Cyanid)	57-12-5	Chemical	Anorganic	Cytochrome c oxidase aa_3 inhibitor (electrone transport, respiration chain)
Cybutryne (Irgarol)	28159-98-0	Herbicide, Biocide (Algicide)	Triazine	PS II inhibitor
Cypermethrin	52315-07-8	Insecticide	Pyrethroid	Sodium channel modulator
DEHP, Bis(2-ethyl-hexyl) phthalate	117-81-7	Chemical	Phthalate	Endocrine disruptor
Diazinon	333-41-5	Insecticide	Organophosphate	AChE inhibitor
Dibutyltin	14488-53-0	Chemical	Organotin	Oxidative phosphorylation inhibitor; suspending maturation of immature thymocytes by inhibiting their interaction/binding with thymic epithelial cells
Dichloromethane	75-09-2	Chemical	Halogenated aliphatic	Carcinogenic, DNA damage after metabolic activation by GST
Dichlorvos	62-73-7	Insecticide	Organophosphate	AChE Inhibitor
Dichlorvos	62-73-7	Insecticide	Organophosphate	AChE Inhibitor
Diflufenican	83164-33-4	Herbicide	Carboxamide	Carotinoid biosynthesis inhibitor
Dimethoate	60-51-5	Insecticide	Organophospate	AChE Inhibitor
Dimoxystrobin	149961-52-4	Fungicide	Strobilurin	Respiration inhibitor (QoL fungicide)
Diuron	330-54-1	Herbicide	Urea, Phenylurea	PS II inhibitor
Epichlorhydrin	106-89-8	Chemical	Organochlorine	Strong irritant, highly reactive, damage genetic material, cancerogenic

Substance	CAS number	Product group / Use type	Substance group	Mode of action
Epoxiconazol	133855988	Fungicide	Triazole	Sterol Biosynthesis Inhibitors, Demethylation of C-14 during ergosterol biosynthesis, affects formation of cell walls of fungi
Erythromycin	114-07-8	Pharmaceutical, Antibiotc	Macrolide	Protein synthesis in bacteria, reversibly binding to 50 S subunit of bacterial ribosomes
Etrimfos	38260547	Insecticide	Organophosphate	AChE Inhibitor
Fenitrothion	122-14-5	Insecticide	Organophosphate	AChE Inhibitor
Fenthion	55389	Insecticide	Organophosphate	AChE Inhibitor
Flufenacet	142459-58-3	Herbicide	Oxyacetamide	Mitosis and cell division inhibitor
Flurtamone	96525-23-4	Herbicide	Pyridazinone	Carotinoid biosynthesis inhibitor
Heptachlor/-epoxid	1024-57-3	Insecticide	Organochlorine	Chloride channel-blocker
Hexachlorocyclohexan (Lindane)	608-73-1	Insecticide	Chlorinated cyclic hydrocarbon. HCH, isomere mixture	Sodium channel blocker (acute) - narcotic. Receptor agonist: e.g. Aryl hydrocarbon, estrogen, androgen
Hexazinone	51235042	Herbicide	Triazinone	PS II inhibitor
Ibuprofen	15687-27-1	Pharmaceutical, Human	Phenylacetate	Exact mechanism unkown, non-selective inhibitor of cyclooxygenase=> prostaglandin synthesis
Imidacloprid	105827-78-9	Insecticide	Neonicotinoid	Acetylcholine receptor (nAChR) agonist
Isoproturon	34123-59-6	Herbicide	Urea	PS II inhibitor
Malathion	121755	Insecticide	Organophosphate	AChE Inhibitor
Metazachlor	67129082	Herbicide	Chloroacetamide	Exact MOA not known, inhibition of protein, fatty acid and lignin synthesis
Methabenzthiazuron	18691979	Herbicide	Urea	PS II inhibitor
Metolachlor	51218452	Herbicide	Chloroacetamide	Mitosis and cell division inhibitor
Metoprolol	37350-58-6	Pharmaceutical	Phenyl Esters, Anisol	Beta(1) -adrenergic receptor blocker in the heart
Metribuzin	21087-64-9	Herbicide	Triazinone	PSII inhibitor
Monobutyltin	78763-54-9	Chemical	Organotin	Oxidative phosphorylation inhibitor. Effect at the level of the cell membrane and/or cytoskeleton, resulting in disturbances of inter and intracellular communication processes, which are of crucial importance to thymocyte maturation

Substance	CAS number	Product group / Use type	Substance group	Mode of action
Monolinuron	1746-81-2	Herbicide	Urea	PS II inhibitor
Nicosulfuron	111991-09-4	Herbicide	Urea, sulfonylurea	Acetolactate synthase (ALS) inhibitor (protein synthesis inhibition)
Nitrobenzene	98953	Chemical	Nitro aromatic	Carcinogen, methaemoglobin formation, cyanosis, neurotoxic effects, spermatotoxicity
Omethoate	1113-02-6	Insecticide	Organophosphate	AChE Inhibitor
Parathion-ethyl	56382	Insecticide	Organophosphate	AChE Inhibitor
Parathion-methyl	298000	Insecticide	Organophosphate	AChE Inhibitor
Pentachlorobenzene	608-93-5	Chemical	Chlorinated aromatic hydrocarbon	Unspecific, nonpolar narcotic
Pentachlorophenol	87-86-5	Chemical	Chlorinated aromatic hydrocarbon	Uncoupler of oxidative phosphorylation
Phosphorsäure-triphenylester (Triphenyl phosphate, TPP)	115-86-6	Chemical	Phosphoric acid ester (flame retartant,	AChE Inhibitor
Phoxim	14816-18-3	Insecticide	Organophosphate	AChE Inhibitor
Picolinafen	137641-05-5	Herbicide	Pyridine compound	Carotenoid biosynthesis inhibitor (phytoene desaturase)
Pirimicarb	23103-98-2	Insecticide	Carbamate	AChE inhibitor
Prometryn	7287196	Herbicide	Triazine	PSII Inhibitor
Propiconazol	60207-90-1	Fungicide	Triazole	Sterol Biosynthesis Inhibitors, Demethylation of C-14 during ergosterol biosynthesis, affects formation of cell walls of fungi
Quinoxyfen	124495-18-7	Fungicide (Azanaphthalene fungicide)	Aryloxyquinoline,	Multi-site MOA, disruption of early cell signaling events, Dihydroorotate-DH ?, serine esterase?, (exact MOA unknown)
Roxithromycin	80214-83-1	Pharmaceutical, Antibiotic	Macrolide	Protein synthesis in bacteria, reversibly binding to 50 S subunit of bacterial ribosomes
Selenium (selenite, selenate	7782492	Metal	Metal, trace element	Prooxidant effects. Oxidative stress. Reaction with tissue thiols by redox catalysis resulting in formation of reactive oxygen species (superoxide anion). Substitutes for sulfur in biomolecules and in many biochemical reactions

Substance	CAS number	Product group / Use type	Substance group	Mode of action
Silver ion	7440-22-4	Metal, Antimicrobial	Metal, precious	Proteins reaction with thiol groups. Effects on bacterial respiration.
Simazin	122-34-9	Herbicide	Triazine	PSII Inhibitor
Sulcotrione	99105-77-8	Herbicide	Triketone	Carotinoid biosynthesis inhibitor
Sulfamethoxazole	723-46-6	Pharmaceutical, Antibiotc	Sulfanilamides	Dihydrofolate synthetase inhibitor, bacterial synthesis of dihydrofolic acid
Terbuthylazine	5915-41-3	Herbicide, Microbiocide	Triazine	PSII Inhibitor
Terbutryn	886-50-0	Herbicide	Triazine	PSII Inhibitor
Tetrabutyltin	1461-25-2	Chemical	Organotin	Oxidative phosphorylation inhibitor
Thallium (+I)	7440280	Metal	Metal	Exact mechanism unknown. Competitor of potassium in enzymes, ion channels, pumps. Suspected cancerogen
Trichlormethan	67-66-3	Chemical	Chlorinated Hydrocarbon	Cancerogenic, possibly, in humans; sufficent evidence in exp. Animals. Toxicity after metabolism
Triclosan	3380-34-5	Biocide	Polychloro phenoxy phenol.	Fatty acid synthesis inhibitor, binds to bacterial enoyl-acyl carrier protein reductase enzyme (ENR)
Trifluralin	1582-09-8	Herbicide	Dinitroaniline	Microtubule assembly inhibition
Uranium (Uranyl salts)	7440-61-1	Metal	Metal, heavy	Protein etc. - Uranyl cations bind tenaciously to protein, nucleotides. Canerogenic
Zink	7440-66-0	Metal, Antimicrobial	Metal, essential trace element	Prooxidant effects. Reaction with tissue thiols by redox catalysis resulting in formation of reactive oxygen species (superoxide anion)

Table 5-34: Attribution of specific biomarkers to substance groups and modes of action.

Biomarker	Substance group /Use type	Mode of action
?	Urea, sulfonylurea	Acetolactate synthase (ALS) inhibitor (protein synthesis inhibition)
?	Neonicotinoid	Acetylcholine receptor (nAChR) agonist
AChE inhibiton	Carbamate	AChE inhibitor
AChE inhibiton	Organophosphate	AChE inhibitor

Biomarker	Substance group /Use type	Mode of action
AChE inhibiton	Phosphoric acid ester (flame retartant,	AChE Inhibitor
Liver histopathology Peroxisomal proliferation	Phenoxyacetates	Agonist of PPARα: peroxisome proliferator and hypolipidaemic agent
?	Phenyl Esters, Anisol	Beta (1) -adrenergic receptor blocker in the heart
DNA adduct formation Macroscopic liver neoplasms ?	Chlorinated Hydrocarbon	Carcinogen, possibly, in humans; sufficent evidence in exp. Animals. Toxicity after metabolism
DNA adduct formation Macroscopic liver neoplasms Micronuclei ?	Hydrocarbon, aromatic	Carcinogen (DNA breaks)
DNA adduct formation Macroscopic liver neoplasms Micronuclei ?	Nitro aromatic	Carcinogen, methaemoglobin formation, cyanosis, neurotoxic effects, spermatotoxicity
DNA adduct formation Macroscopic liver neoplasms Micronuclei ?	Halogenated aliphatic	Carcinogenic, DNA damage after metabolic activation by GST
? Photosynthesis inhibiton (no biomarker defined)	Pyridine compound	Carotenoid biosynthesis inhibitor
? Photosynthesis inhibiton (no biomarker defined)	Diphenylether	Carotenoid synthesis and protoporphyrinogen oxidase inhibitor
? Photosynthesis inhibiton (no biomarker defined)	Carboxamide	Carotinoid biosynthesis inhibitor
? Photosynthesis inhibiton (no biomarker defined)	Pyridazinone	Carotinoid biosynthesis inhibitor
? Photosynthesis inhibiton (no biomarker defined)	Triketone	Carotinoid biosynthesis inhibitor
General marker, e.g. EROD activity Glutathione-S-transferase (GST) Liver histopathology	Chlorinated hydrocarbon, aliphatic	Chloride channel-blocker
Macroscopic liver neoplasms DNA adducts (?) Micronuclei	Chlorinated hydrocarbon, aliphatic	Exact mechanism not clear: carcinogenic, hepatic, renal, neurological effects u.a. Possibly reactive intermediates formed by conjugation with glutathione
Lysosomal membrane stability Macroscopic liver neoplasms DNA adducts Liver histopathology	Chlororganic compound, aliphatic, epoxid	Strong irritant/corrosive, highly reactive, damage genetic material, cancerogenic
Lysosomal membrane stability	Chlororganic acid	Corrosive. MCA blocks the cell energy supply probably by decreasing the activity of pyruvate dehydrogenase and to some extent of ketoglutarate dehydrogenase

Biomarker	Substance group /Use type	Mode of action
General marker, e.g. EROD activity	Cyanids	Cytochrome c oxidase aa_3 inhibitor (electrone transport, respiration chain), cardiac, neurological, and metabolic dysfunctions
Bacteria specific (no biomarker defined)	Sulfanilamides	Dihydrofolate synthetase inhibitor, bacterial synthesis of dihydrofolic acid
Photosynthesis inhibiton (no biomarker defined)	Chloracetanilide	Elongase inhibition, and inhibition of geranylgeranyl pyrophosphate (GGPP) cyclisation enzymes, part of the gibberellin pathway, *PS Inhibitor*
Peroxisomal proliferation Vitellogenin induction in fish Macroscopic liver neoplasms DNA adducts (?)	Phthalate	Endocrine disruptor, possible carcinogen, PPAR α: peroxisome proliferator etc.
Vitellogenin induction in fish	Estrogen	ER agonist
Macroscopic liver neoplasms DNA adducts (?) Micronuclei	Aromatic amine	Exact mechanism not clear: Carcinogenic, after metabolic activation
DNA adducts Micronuclei formation Oxidative stress Macroscopic liver neoplasms Lysosomal membrane stability MDR/MXR	Metal (Cr)	Cr (VI) cancerogenic. Cr (III): During the reduction process of Chromium (VI) to Chromium (III) in the body, along with reactive intermediates, chromium adducts with proteins, deoxyribonucleic acid (DNA), and secondary free radicals are also formed
Macroscopic liver neoplasms Lysosomal membrane stability MDR/MXR	Metal (Tl)	Exact mechanism unknown. Competitor of potassium in enzymes, ion channels, pumps. Suspected cancerogen.
Oxidative stress Lysosomal membrane stability MDR/MXR	Metal (Se) essential trace element	Prooxidant effects. Oxidative stress. Reaction with tissue thiols by redox catalysis resulting in formation of reactive oxygen species (superoxide anion). Substitutes for sulfur in biomolecules and in many biochemical reactions
Oxidative stress Metallothione induction Lysosomal membrane stability MDR/MXR	Metal (Zn) essential trace element	Prooxidant effects. Reaction with tissue thiols by redox catalysis resulting in formation of reactive oxygen species (superoxide anion).
Bacteria specific (no biomarker defined) Lysosomal membrane stability MDR/MXR	Metal, precious (Ag)	Proteins reaction with thiol groups. Effects on bacterial respiration.
Metallothione induction Lysosomal membrane stability MDR/MXR	Metal, heavy (Cu) essential trace element	Proteins, nonspecific denaturation (disruption) of cellular proteins (enzymes). Link to various chemical groups (imidazoles, phosphates, , sulfhydryls, hydroxyls groups.

Biomarker	Substance group /Use type	Mode of action
Macroscopic liver neoplasms DNA adducts Lysosomal membrane stability MDR/MXR	Metal, Uranyl ion	Protein etc. - Uranyl cations bind tenaciously to protein, nucleotides. Carcinogenic
?	Phenylacetate	Exact mechanism unkown, non-selective inhibitor of cyclooxygenase=> prostaglandin synthesis
? (Photosynthetic activity?)	Chloroacetamide	Exact MOA not known, inhibition of protein, fatty acid and lignin synthesis.
Bacteria specific (no biomarker defined) Peroxisomal proliferation	Polychloro phenoxy phenol	Fatty acid synthesis inhibitor, binds to bacterial enoyl-acyl carrier protein reductase enzyme (ENR)
Micronuclei	Dinitroaniline	Mitosis and cell division inhibitor Microtubule assembly inhibition
Micronuclei	Benzimidazole	Mitosis and cell division inhibitor
Micronuclei	Chloroacetamide	Mitosis and cell division inhibitor
Micronuclei	Oxyacetamide	Mitosis and cell division inhibitor
?	Aryloxyquinoline	Multi-site MOA, disruption of early cell signaling events, e.g. dihydroorotate-DH, serine esterase, (exact MOA unknown)
Imposex	Organotin	Oxidative phosphorylation inhibitor. Effect at the level of the cell membrane and/or cytoskeleton, resulting in disturbances of inter and intracellular communication processes, which are of crucial importance to thymocyte maturation.
?	Phenoxy compound	Plant hormone, synthetic auxin = uncontrolled growth
Bacteria specific (no biomarker defined)	Macrolide	Protein synthesis in bacteria, reversibly binding to 50 S subunit of bacterial ribosomes
? Photosynthesis inhibiton (no biomarker defined)	Diphenylether	Protoporphyrinogen oxidase inhibitior (chlorophyll synthesis) *PS Inhibitor*
? Photosynthesis inhibiton (no biomarker defined)	Triazine	PS II inhibitor
? Photosynthesis inhibiton (no biomarker defined)	Triazinone	PS II inhibitor
? Photosynthesis inhibiton (no biomarker defined)	Uracil	PS II inhibitor
? Photosynthesis inhibiton (no biomarker defined)	Urea	PS II inhibitor
? Photosynthesis inhibiton (no biomarker defined)	Nitrile herbicide	PSII inhibitor
?	Strobilurin	Respiration inhibitor (QoL fungicide)
General marker, e.g. EROD activity	Dibenzazepine; Analgesics, Non-Narcotic	Sodium channel blocker

Biomarker	Substance group /Use type	Mode of action
General marker, e.g. EROD activity M-ACT test Vitellogenin induction in fish	Chlorinated cyclic hydro-carbon. HCH, isomere mixture	Sodium channel blocker (acute) - nar-cotic. Receptor agonist: e.g. Aryl hydro-carbon, estrogen, androgen, immune modulator
General marker, e.g. EROD activity	Pyrethroid	Sodium channel modulator
Vitellogenin induction in fish	Triazole	Sterol Biosynthesis Inhibitors, Demethyl-ation of C-14 during ergosterol biosyn-thesis, affects formation of cell walls of fungi
Oxidative stress Lysosomal membrane stability MDR/MXR	Metal (As)	Uncoupler of oxidative phosphorylation. Thiol affinity, enzyme inhibition.
General marker, e.g. EROD activity	Chlorinated aromatic hy-drocarbon	Uncoupler of oxidative phosphorylation
General marker, e.g. EROD activity	Chlorinated aromatic hy-drocarbon	Unspecific, nonpolar narcotic

Conclusions

This chapter provides a comprehensive overview of biomarkers for exposure and effects on different biological levels. The survey shows that most of the biomarkers are not substance-specific and thus do not allow the iderntification of specific contaminants. However, the combination of different biomarkers in a monitoring programme allows to narrow down the group of contaminants present in the marine environment responsible for biological effects. Many of the available biomarkers need further validation and development to guarantee reliable results when used in monitoring programmes.

5.4 Effects of harmful substances on marine mammals

5.4.1 Introduction to marine mammal species from German waters

5.4.1.1 Harbour porpoise (*Phocoena phocoena*)

The harbour porpoise (*Phocoena phocoena*) is the most common cetacean in German waters (Benke et al., 1998; Siebert et al., 2006a). Adult animals reach a body mass between 45 and 75 kg (maximum 90 kg) and a body length of 150-165 cm (maximum 185 cm) (Benke et al., 1998; Siebert et al., 2012a). The males are smaller and lighter than the females. Sexual maturity is reached by age three or four. Their reproductive cycle is strongly associated with the seasons. Peak birth rate occurs between the months of June and August in the North Sea and one month later in the Baltic Sea (Hasselmeier et al., 2004). Adult females of both the North and Baltic seas are usually lactating and pregnant during the same time of year. In German waters they can reach up to 25 years of age, but mean life span is about 10 years (Read and Hohn, 1995; Siebert et al., 2012a).

Harbour porpoises occur most often in coastal water habitats of Europe, North America and Northern Africa with water temperatures from approximately 0.5°C to 24°C (Hammond et al., 2008; Siebert et al., 2012a). Harbour porpoises in German waters belong to three different subpopulations: Southern and Central North Sea, Western Baltic and Baltic Proper (Tiedemann et al., 1996; Wiemann et al., 2010; Huggenberger et al., 2002). The aerial abundance estimates in the German North Sea between 2002 and 2006 ranged from 55,048 in spring to 15,394 in autumn (Gilles et al., 2009). In Figure 5-27 the distribution of harbour porpoises between 2002 and 2010 is shown. Their distribution depends on the season. In the Western Baltic Sea 23,227 animals were estimated to be present in the area from Skagerrak up to the island of Rügen in 2005 (Hammond et al., 2013). In summer 2012 a ship-based survey for harbour porpoises density and abundance was conducted in Kattegat, Beltsea and Western Baltic (Gap Area, ASCOBANS, 2012). Due to the results the abundance for the observed area of 51.511 km² was estimated at 40,475 animals (95% CI: 25,614 – 65,041, CV (Coefficient of Variation) = 0.235), the density was estimated at 0,786 animals/km² (95% CI: 0.498 – 1.242, CV = 0.235) (Viquerat et al., 2013). For German waters the highest density within the Western Baltic population is found around the Kiel Bight by aerial surveys (Scheidat et al., 2008). The Baltic proper population is considered to be seriously depleted (BfN, 2007) and it is estimated that the size of the population has decreased to a few hundred animals (Berggren et al., 1995; Koschinski, 2002). Therefore since 2006 no aerial surveys are performed. But porpoise detector (POD) based surveys were made (Gallus et al., 2012).

Harbour porpoises in the North Sea primarily subsist on sandeel and sole, and in the Baltic Sea on goby, herring and cod (Benke et al., 1998; Lockyer and Kinze, 2003). There is, however, both seasonal and annual fluctuation (Gilles et al., 2009).

Apart from bycatch, which is a major cause of death (Kock and Benke, 1996; Vinther and Larsen, 2004), noise and chemical pollution are permanent stressors for harbour porpoises and impact their health status (Richardson et al., 1995; Siebert et al., 1999; Das et al., 2006; Beineke et al., 2005; Lucke et al., 2009), but information is still deficient. Altogether, cumulative effects of different anthropogenic activities are poorly understood.

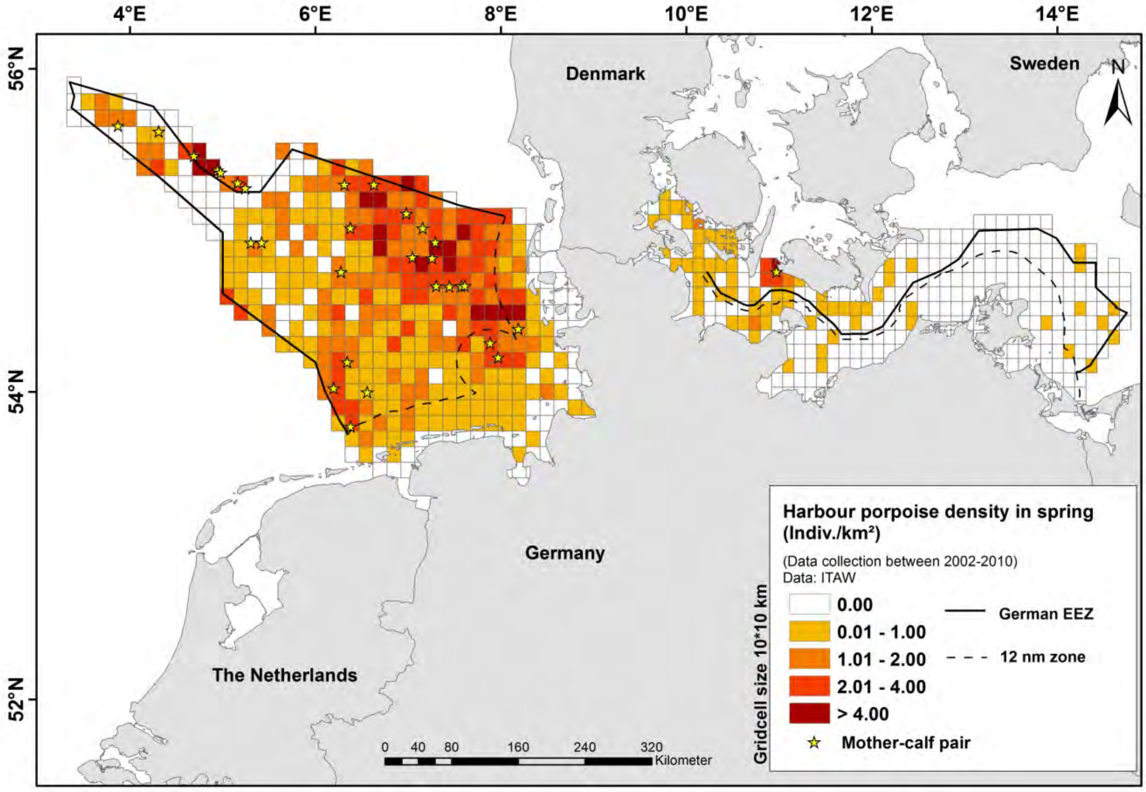


Figure 5-27: Spring distribution of harbour porpoises (Individuals/km²) in the German North and Baltic Sea derived from line transect aerial surveys and positions of mother-calf pairs sighted during March - May in 2002-2010. Note: Balic proper were monitored by aerials surveys until 2006, (modified after Siebert et al., 2012a).

5.4.1.2 Harbour seal (*Phoca vitulina*)

The subspecies *Phoca vitulina vitulina* (Figure 5-28) is resident in German waters. They inhabit the southern North Sea and are also found in the German Baltic Sea as vagrants where they do not reproduce (Reijnders et al., 2005; Siebert et al. 2012a).

Males can reach a body length of 180 cm and a weight of up to 130 kg, while females can reach a body length of 150 cm and a maximum weight of 105 kg (King, 1983). They can live up to 30- 35 years (Siebert et al., 2012a). Sexual maturity of female seals is reached between the 3rd and 4th years (Burns, 2002), and by males between the 4th to 5th years. Physical maturity is reached by females in six to nine years (Burns, 2002) and by males in seven to nine years. The annual birth peak in the German Wadden Sea is in June (95%) (Abt, 2002). Harbour seals are solitary animals but they use sandbanks and undisturbed beaches for resting, moulting, pupping and lactation in groups (Reijnders et al., 2005). Harbour seals are opportunistic feeders so they feed on a variety of fish species, but primarily flatfish and other demersal fish (Burns, 2002; Gilles et al., 2008).

Figure 5-28: Harbour seals in the German Wadden Sea, (Picture: Abbo van Neer; ITAW)

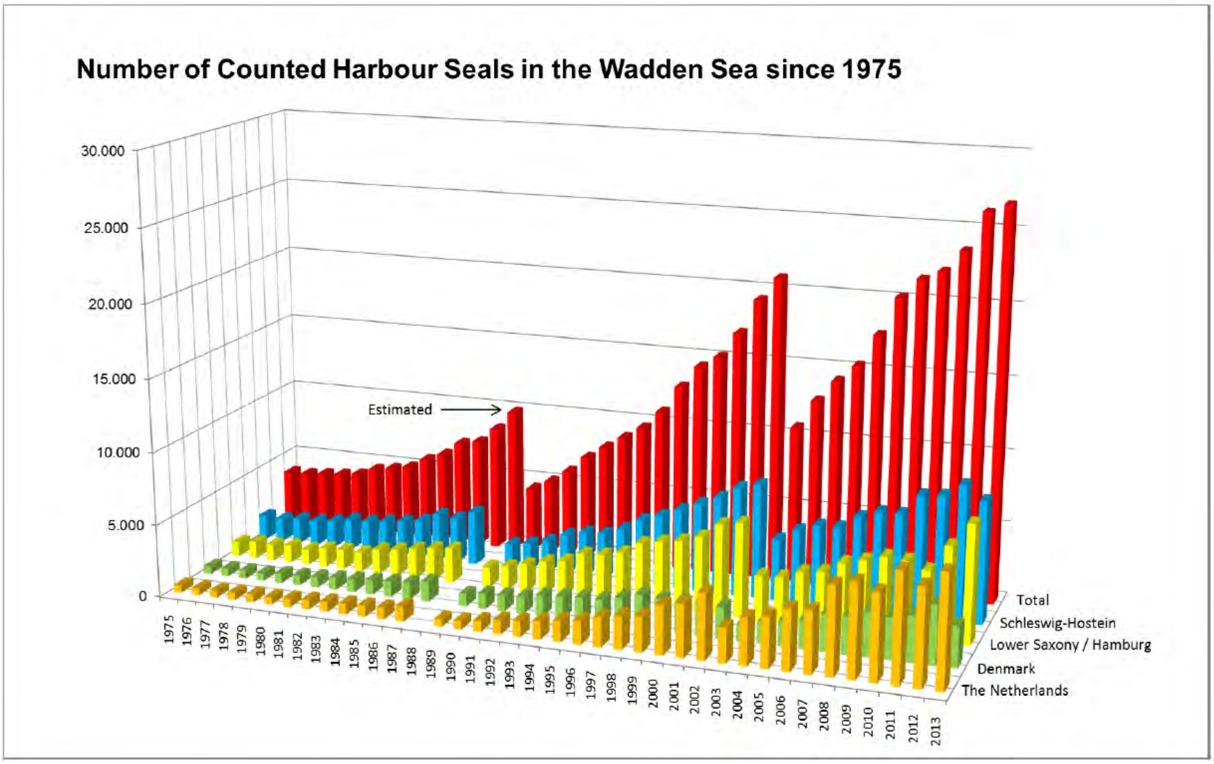


Figure 5-29: Number of counted harbour seals in the Wadden Sea (Netherland, Denmark, Germany) (Common-Wadden-Sea-Secretariat, 2013a).

In the years 1988 and 2002 the phocine distemper virus (PDV) caused mass mortalities. Several thousand harbour seals died from this infection (Müller et al., 2004; Härkönen et al., 2006). Current research of the year 2013 lead to a population size estimation of 39,400 harbour seals, due to 26,788 counted individuals (Common-Wadden-Sea-Secretariat, 2013a). This allows us to conclude that the population has recovered in the Wadden Sea. In Figure 5-29 the number of counted harbours seals by aerial surveys is shown. The seals were counted during low tide in the Wadden Sea, while moulting out during midday (Common-Wadden-Sea-Secretariat, 2013a). In the Baltic Sea colonies of harbour seals are found on the Danish and Southern Swedish coast. Occasionally vagrant younger harbour seals can also occur in German waters of the Baltic. But harbour seals do not reproduce in German waters of the Baltic Sea (Siebert et al., 2012a).

5.4.1.3 Grey seal (Halichoerus grypus)

Grey seals (Figure 5-30) show sexual dimorphism in size, colour and head shape. Pups are born with white lanugos, a body length of 90-110 cm and a weight of 13-18 kg (Hall, 2002; Siebert et al., 2012a). Adult Eastern Atlantic males reach 200 cm and females 180 cm in length and the maximum weight of an East Atlantic male is 310 kg (Hall, 2002; Siebert et al., 2012a). Males are darker in colour than females. Sexual maturity in males begins at the age of six. Females reach sexual maturity at the age between three and five. All three breeding colonies in German waters are all located in the North Sea (TSEG, 2012) with the largest colony on Helgoland. The pupping time is in winter (November to January) (TSEG, 2012).

Figure 5-30: Grey seals at Helgoland, (Picture: Abbo van Neer, ITAW)

Grey seals are also domiciled in the Baltic Sea but only a few sightings were reported in the German Baltic Sea (Herrmann et al., 2007). During surveys in the North Sea in 2012 and 2013 while spring moult, 2,785 individuals were counted in all areas of the Wadden Sea (Common-Wadden-Sea-Secretariat, 2013b). This is 31% lower than last year (Common-Wadden-Sea-Secretariat, 2013b) (Figure 5-31 and Figure 5-32). Grey seals prefer rocky shores, coastal areas, caves and ice in the north as habitat. In German waters they settle down on sandy islands and beaches. Their longevity has been estimated to be between 35-40 years (Siebert et al., 2012a). Grey seals, like harbour seals, are opportunistic feeders. In German waters their main food source is demersal and benthic fish.

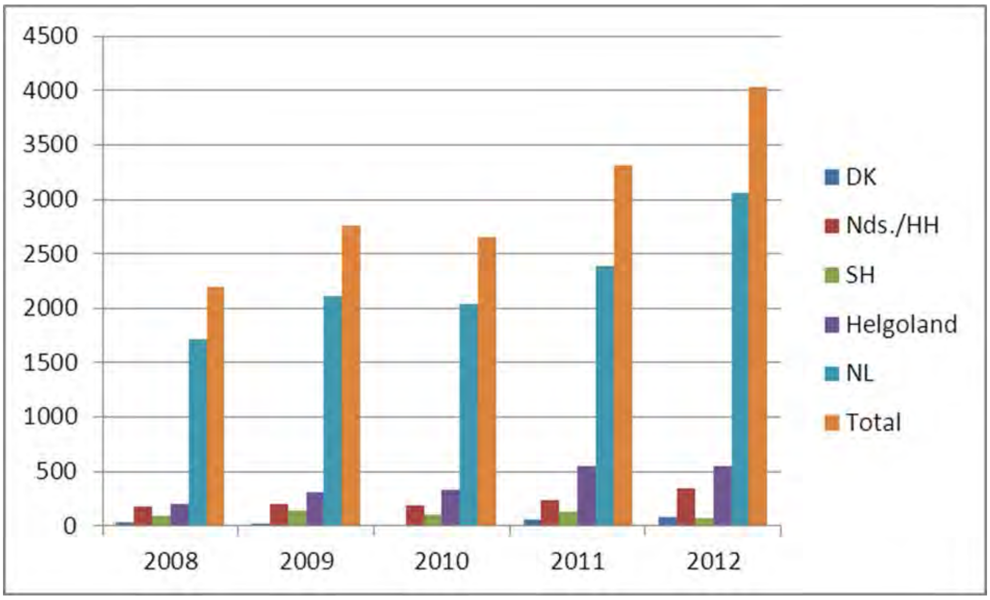


Figure 5-31: Grey seal population in Denmark (DK), Lower Saxony and Hamburg (Nds./HH), Schleswig-Holstein (SH), Helgoland, Netherlands (NL) and total during 2008 and 2012 (Common-Wadden-Sea-Secretariat, 2013c).

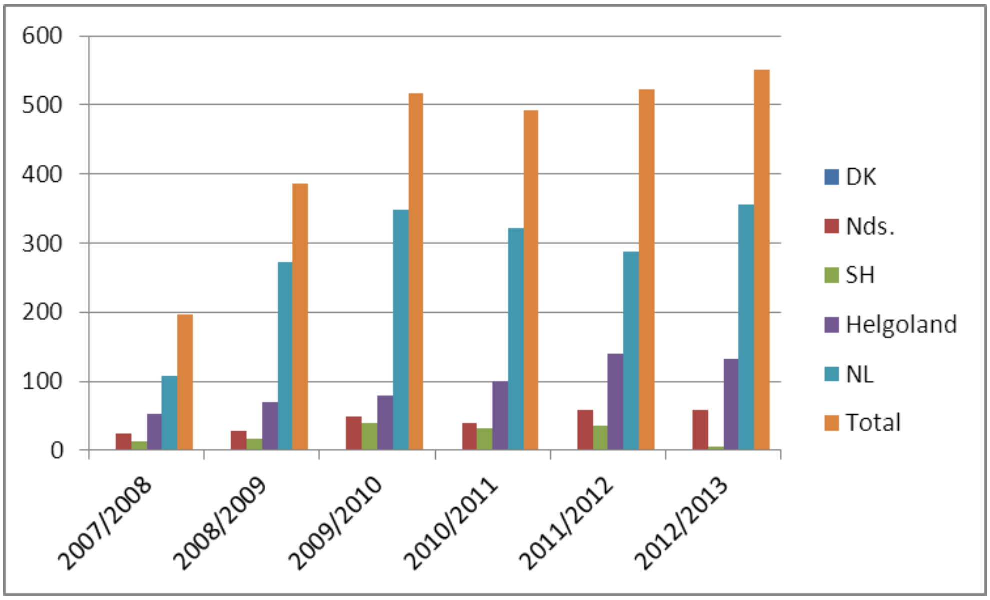


Figure 5-32: Grey seal pups in Lower Saxony and Hamburg (Nds./HH), Schleswig-Holstein (SH), Helgoland, Netherlands (NL) and total during the season 2007/8 and 2012/13 (Common-Wadden-Sea-Secretariat, 2013c).

5.4.2 Current knowledge of pollutant burdens in marine mammals from German waters

The following current knowledge results from a literature search which sometimes had to rely on older studies to obtain the information needed. Therefore, the concentration of harmful substances as well as the measurements and organs/materials examined differ tremendously. Also, the different units make the comparison and the evaluation of the process difficult. On the other hand, this result shows the variety of pollutant burdens in different samples. Contaminants were found for example in liver, kidney, muscle, lipid/blubber and blood.

Marine mammals are particularly likely to accumulate contaminates as they are top predators (Siebert et al., 2012a). Depending on the concentration of harmful substances they can lead to acute and chronical diseases or organ damages (Siebert et al., 2012a). The transfer of contaminants through placenta is known to occur in humans (Covaci et al., 2002a) and other mammal species, including marine mammals (Kakuschke et al., 2009; Habran et al., 2013). Even pups which feed on their mother milk have a contaminant intake during lactation (Debier et al., 2003; Habran et al. 2013).

A lot of research of pollutant burdens in marine mammals has been performed. In the following text some examples of measured values of different hazard substances in marine mammals are summarized. Moreover, reviews of hazards substances in marine mammals can be found (e.g. Aguillar and Borrell, 1995; Das et al., 2003; Kakuschke and Prange 2007; Szefer et al., 2002) and an overview of the current knowledge is given. Older studies (e.g. Drescher et al., 1977) can give evidence to the development of pollutant burdens but not all are listed in this text.

Right now, contaminant analyses are not performed in the course of the regular monitoring program on marine mammal carcasses from Germany.

5.4.2.1 Overview of research on pollutant burdens in marine mammals

Table 5-35: Literature about hazardous substances in **harbor porpoises**: hazardous substances: mercury (Hg), cadmium (Cd), lead (Pb), arsenic (As); samples: blood (Bl), blubber (B), brain (Br), heart (H), kidney (K), liver (Li), lung (Lu), muscle (Mu),melon (Me), spleen (S), urin (U); regions: Belgium (BE), Germany (DE) Denmark (DK), Spain (ES), Finland (FI), France (FR), Great Britain (GB), Greenland (GL), Ireland (IE), Iceland (IS), Netherland (NL), Norway (NO), Poland (PO), Sweden (SE), North (N) and Baltic (B) Sea; year of sampling (year), not reported (n.r.).

Hazardous substances	Samples	Region of sampling	Year	Author and Year
PBC	B, Li, Mu	DK	1972- 1973	Andersen & Rebsdorff 1976
PCB, DDT, Dioxin	B	DE (N)	1988- 1989	Beck et al. 1990
PCB	B	DK	1986- 1988	Granby and Kinze 1991
PCB, DDT, Dioxin	B	DE (N+B), GL	1993-1995	Bruhn et al. 1999
PCB	B	GB	1991- 2005	Law et al. 2010
PCB	B	DE (N)	1999- 2002	Siebert et al. 2002
PCB, DDT, PBDEs	B	DE (N+B), DK (N+B), NO, IS	1998- 2001	Das et al. 2006
PCB, DDT	Li	BE (N)	1997- 2000	Covaci et al. 2002b
PCB, PBDEs	B	BE, NL, DE (N)	1999- 2004	Weijs et al. 2009b
DDT	Bl	NL	2006- 2008	Weijs et al. 2009a
DDT	B	DK	1972- 1973	Andersen & Rebsdorff, 1976

Hazardous substances	Samples	Region of sampling	Year	Author and Year
DDT	B	DK, GL	1986- 1988	Granby and Kinze, 1991
PFOS	Li, Ki	BE, FR, NL	1995- 2000	Van de Vijver et al. 2003
PFOS	Li	IS, NO, DK, DE (N+B)	n.r	Van de Viejver et al., 2004
PFOS	Li	DE (N+B)	1991- 2008	Huber et al. 2012
PBDEs	B	DE (N+B), IS, NO	1997- 2001	Thron et al. 2004
PBDEs	B	IS, NO	1992- 1997	Rotander et al. 2012
TBT	Li	DK, GL	1998- 1999	Strand et al. 2005
PCB, DDT, Dioxin	B	SE, NO, DK	1978- 1993	Berggrena et al. 1999
TBT	Li	PO	1996- 2003	Ciesielski et al. 2004
Hg	Mu, Li, K	DE (N+B)	1991-1993	Siebert et al. 1999
Hg, Cd, Pb	Li	GB	1988-1989	Law et al. 1991
Hg, Cd, Pb	Li	GB	1990-1994	Bennet et a. 2001
Hg, Cd, Pb	Br, Li, K, S, H	GB	1974	Falconer et al. 1983
Hg, Cd, Pb	B, Mu, Li, Me	GB	1988	Morris et al. 1989
Hg	Li, K	IE, GB, NL, ES, FR	1997-2003	Lahaye et al. 2007
Hg	Mu, Li	DE (N+B)	n.r.	Harms et al. 1978
Hg	Li	DE (N+B), BE, FR, DK, NO, IS	1994-2001	Das et al. 2004
Hg	Li	FI, DK, NO	1994-2001	Fontaine et al. 2007
Cd, Pb	Mu, Li, K	DE (N+B)	n.r.	Harms et al. 1977
Cd	K	IE, GB, NL, ES, FR	1997-2003	Lahaye et al. 2007
Cd	Li, K, Mu	DE (N+B), BE, FR, DK, NO, IS	1994-2001	Das et al. 2004
Cd	K	FI, DK, NO	1994-2001	Fontaine et al. 2007
As	Li, U	DE (N+B)	n.r.	Künstl et al. 2009
Cd	Li, K, Mu	DK, GL, PO	1993-1996	Szefer et al. 2002
Pb	Li, K	DK, GL, PO	1993-1996	Szefer et al. 2002
Hg	Li	DK (N+B), GL	1998-1999	Strand et al. 2005
Hg, Pb	Li, Mu, B	DK	1972-1973	Andersen & Rebsdorff 1976
Hg, Cd, Pb, As	Li	PO	1999-2003	Ciesielski et al. 2006

Table 5-36: Literature about hazardous substances in **harbour seals**: hazardous substances: mercury (Hg), cadmium (Cd), lead (Pb), arsenic (As); samples: blood (Bl), blubber (B), brain (Br), gastric juice (G), hair (Ha), heart (H), kidney (K), liver (Li), milk (Mi), muscle (Mu), placenta (P), skin (Sk) spleen (S), urin (U); regions: Belgium (BE), Germany (DE), Danmark (DK), Greate Britain (GB), Netherland (NL), Norway (NO), Poland (PO) United States of America (USA), North (N) and Baltic (B) Sea; year of sampling (Year), not reported (n.r.).

Hazardous substances	Samples	Region of sampling	Year	Author and Year
PCB, DDT	B	DE (N)	1974- 1976	Drescher et al. 1977
PFOS	Li, Ki	BE, FR; NL	1995-2000	Van de Viejver et al. 2003
PCB, DDT	B, Li, K, Br	NL (N)	n.r.	Duinker et al. 1979
PCB, DDT	B	NO	1988	Skaare et al. 1990
PCB, DDT, Dioxon	B	DE (N)	1988-1989	Beck et al. 1990

Hazardous substances	Samples	Region of sampling	Year	Author and Year
PCB, DDT	Bl	DE, DK (N)	2006- 2008	Weijs et al. 2009a
PCB	B	USA	n.r.	Shaw et al. 2005
PFOS	Li	DE	2007	Ahrens et al. 2009
PFOS	Li	USA	2000- 2007	Shaw et al. 2009
PFOS	Li, K, Mu, S	NL	2002	Van de Viejver et al. 2005
PBDEs, PCB	B	BE, NL, DE (N)	1999- 2004	Weijs et al. 2009b
PCB, DDT	B, Li	DK, DE (N)	2001- 2002	Siebert et al. 2012b
Hg	Mu, Li	DE (N)	n.r.	Harms et al. 1978
Hg, Cd, Pb	Li	GB	1988-1989	Law et al. 1991
Hg	Bl	DE (N)	1997-2004	Das et al. 2008
Hg, Cd, Pb	Ha, Sk	DE (N)	1988	Wenzel et al. 1993
Hg, Cd, Pb	Li, Br, K	DE (N)	1974-1976	Drescher et al. 1977
Cd, Pb, As	Bl	DE (N), DK	2003-2004	Griesel et al. 2008, Kakuschke et al. 2005
Cd, Pb	Mu, Li, K	DE (N)	n.r.	Harms et al. 1978
Pb, Cd	B, Li, K, Br, S, H, P	NL	n.r.	Duinker et al. 1979
Hg	Li, K, Br	DE, DK, NL	1975-1976	Reijnders et al. 1980
As	Li, U, G, Bl	DE	n.r.	Künstl et al. 2009
Hg, Pb, As	Li	PO	2002	Ciesielski et al. 2006

Table 5-37: Literature about hazardous substances in **grey seals**: hazardous substances: mercury (Hg), cadmium (Cd), lead (Pb), arsenic (As); samples: blood (Bl), blubber (B), brain (Br), hair (Ha), heart (H), kidney (K), liver (Li), milk (Mi), muscle (Mu), spleen (S); regions: Belgium (BE), Germany (DE):, Finland (Fi), France (FR), Greate Britain (GB), Netherland (NL), Poland (PO) North (N) and Baltic (B) Sea; year of sampling (Year), not reported (n.r.).

Hazardous substances	Samples	Region of sampling	Year	Author and Year
PCB	Bl, B, Mi	GB	1998- 2000	Debier et al. 2003
DDT	B, K	GB	1988	Law et al. 1989
PFOS	Li, K	BE, FR, NL	1995- 2000	Van de Viejver et al. 2003
PFOS	Li	FI	n.r.	Kannan et al. 2002
TBT	Li	PO	1996- 2003	Ciesielski et al. 2004.
Hg, Cd, Pb	Li	GB	1988-1989	Law et al. 1991
Hg	Ha, Bl, Mi	GB	2008	Habran et al. 2013
Hg, Cd, Pb	Li	DE (N)	n.r.	Harms et al. 1978
Pb, As, Cd	Bl	DE (N), GB	2003	Kakuschke et al. 2006
Pb, Cd	Bl, Mi, B, Ha	GB	2008	Habran et al. 2013
Hg, Cd, Pb	B, Mu, Li, K	GB	1988	Morris et al. 1989
Hg, Cd, Pb, As	Li	PO	1996-2003	Ciesielski et al. 2006

5.4.2.2 German North Sea and adjacent waters

5.4.2.2.1 Organic substances

PCBs

Between 1972 and 1973, Andersen and Rebsdorff (1976) measured the PCB concentrations in different tissues of harbour porpoises from Danish coasts. The animals were trapped in pound nets (n=7) and most of them died shortly after. Blubber (n=4) concentrations of PCB ranged between 28 and 125 (median 78) parts per million (ppm) wet weight (ww). Liver (n=4) concentrations were 0.7-11 (median 5.4) ppm ww, muscle (n=4) concentrations ranged from 0.3-2.1 (median=1.3) ppm ww (Andersen and Rebsdorff, 1976).

After the mass mortality in 1988/89 due to PDV, one harbour porpoise was found dead and the blubber was taken for analyses of organochlorine compounds. The animal died because of a carcinoma. PCB 153 values were 20.3 ppm fat weight basis (fwb) (Beck et al., 1990).

Grandby and Kinze (1991) measured a median concentration of 11 mg/kg ww from blubber of harbour porpoises (age<one year) in the Danish North Sea. Harbour porpoises older than three years showed a blubber concentration of 23 mg/kg ww. Samples (n=27) were collected between 1986 and 1988 (Granby and Kinze, 1991).

Between 1999 and 2002 blubber samples of stranded and bycaught harbour porpoises from the German North Sea and Baltic Sea coast were taken. For the North Sea a median concentration of 17.01 µg/g lipid weight (lw) was measured. In comparison to that they found median concentrations of 14.91 µg/g lw in harbour porpoises of the Baltic Sea and 1.31 µg/g lipid in harbour porpoises from Arctic waters (Bruhn et al., 1999).

In the UK, blubber samples were taken from 440 stranded or bycaught harbour porpoises. PCB levels ranged between 13.34 mg/kg lw in 1991, 16.08 mg/kg lw in 1993 and 6.36 mg/kg in 2005. Adult male harbour porpoises showed the highest concentrations, as females transfer the lipophilic contaminants to their offspring (Law et al., 2010).

The Federal Agency of Environment funded a project on the influence of environmental chemicals on the endocrine and immune system of harbour porpoises. The research project took place between 1999 and 2002. In this period 148 harbour porpoises were examined. The median blubber concentration of PCB was measured at 2.5 µg/g lipid for the German North Sea (Siebert et al., 2002).

In another study, the thyroids of 57 harbour porpoises from German and Danish (North and Baltic Sea), Norwegian and Icelandic coasts were examined. Blubber was examined morphologically and tested for PCB, among other contaminants. The measured concentration of PCB was 7,664±5,075 ng/g lw for the German North Sea, 1,550±1,517 ng/g lw for the Icelandic coast, 4,710±2,861 ng/g for the Norwegian coast. It differs clearly from the values they got for the German and Danish costs of the Baltic Sea: 8,247±7,949 ng/g lw (Das et al., 2006).

The median PCB level of 21 harbour porpoise livers from the Belgian North Sea coast was measured 1997 to 2000. It was 33.8 µg/g lw (Covaci et al., 2002b).

A study from the Southern North Sea 2008 revealed PCB blubber concentrations of 12.4 µg/g lw in harbour porpoises (n=35), collected between 1999 and 2004 (Weijs et al., 2009b).

Early measurements of stranded or shot harbour seals from the German Wadden Sea were taken between 1974 and 1976. Blubber concentrations ranged between 27.3 to 564 mg/kg for PCB (Drescher et al., 1977).

Organochlorine concentrations in several tissues from harbour seals were measured (Duinker et al., 1979). The study took place in the Dutch Wadden Sea and included stranded animals. PCB-blubber concentrations ranged from 22-576 mg/kg ww (n=7). In comparison to that, liver concentrations ranged from 1.5-36 mg/kg ww (n=5), kidney from 1.6-31 mg/kg ww (n=2) and in the brain 1.4-46 mg/kg ww (n=3). The compound with the highest concentration in the investigated tissues was PCB and it showed the highest values in blubber, probably because PCB is a lipophilic compound (Duinker et al., 1979).

In Norway, the PCB blubber concentration was measured of harbour seals from summer 1988. For this study, 33 harbour seals under one year of age and 43 animals over one year were collected and considered. PCB levels were two to four times higher than DDT values. PCB in below one-year old animals (n=33) was at 7.1±3.8 mg/kg ww, in 17 females over one year old it was 8.2±3.6 mg/kg ww and in 26 males 14.5±2.1 mg/kg ww (Skaare et al., 1990).

After the mass mortality in 1988/89, caused by PDV, organochlorine compounds were suspected to weaken the immune system. For a study, five of these animals were chosen to perform a PCB analyses, beyond others. The mean concentration of PCB 153, found in blubber, was 7.85 ppm fwb (Beck et al., 1990).

Between 2006 and 2008 serum was sampled from 47 wild living harbour seals from the Danish and German North Sea and 21 harbour porpoises living in human care at Dolfinarium Harderwijk for PCB analysis (Weijs et al., 2009a). The samples were collected between 2006 and 2008. The PCB concentration in harbour seals ranged between 13.3 and 148.3 pg/ml (Weijs et al., 2009a).

A study from the Southern North Sea showed PCB blubber concentrations of 23.1 µg/g lw in harbour seals (n=28). Samples were taken between 1999 and 2004 (Weijs et al., 2009b). The contaminant levels of harbour porpoises and harbour seals have been compared and results showed that harbour seals are better able to metabolize PCBs and PBDEs (Weijs et al., 2009b).

In the blubber of 30 northwestern Atlantic harbour seals, PCB concentrations were measured in consideration of age (Shaw et al., 2005): In 2005, the PCB concentration in male blubber tissue was at 55,000±40,800 ng/g ww. Female adults had a median blubber PCB concentration of 11,100±3,680 ng/g ww, yearlings had a blubber concentration of 33,300±40,500 ng/g ww which is much higher than the concentration measured in female adults. Pups already had a concentration of 43,000±23,300 ng/g ww in their blubber tissue and fetus 5,660±1,610 ng/g ww (Shaw et al., 2005). This underlines the transfer of pollutants to the offspring by milk.

Blubber and liver samples were tested for PCB (discussed here: PCB_153) of 20 harbour seals from the German Wadden Sea to compare them with results from Greenland ringed seals. Blubber concentrations in harbour seals were 5,767.3 ng/g lw in comparison with 89.41 ng/g lw in ringed seals (Siebert et al., 2012b).

To find out more about the transmission of PCB from adult females to their offspring, a study conducted on Isle of May, Scotland, took serum, milk and blubber samples from mother and pup. Milk concentrations increased at the end of the lactation period: Beginning 0.31±0.17 µg/g milk, end 0.67±0.42 µg/g milk. The serum PCB concentration of the pup increased during lactation, as well as the serum concentration of the mother: Early lacation 6.69±3.45 ng/ml maternal, 11.86±7.00 ng/ml. Late lactation 12.18±7.15 ng/ml maternal and 27.89±18.13 ng/ml pup. Newborn grey seals had already higher PCB concentrations than their mother. Inner and outer blubber was taken through biopsis from the mother. Concentrations differed a lot between inner and outer blubber tissue: Early lactation 1.26±0.72 µg/g lipids inner blubber and 3.16±1.34 µg/g lipids outer blubber. In late lactation the results were similar: 3.24±2.60 µg/g lipids inner blubber and 3.59±1.46 µg/g lipids outer blubber. Samples were taken in 1998 and 2000 of 22 mother-pup pairs (Debier et al., 2003).

DDT

In 1972 to 1973 the DDT concentrations of seven harbour porpoises, trapped in pond nets at Danish coasts, were investigated. Results show the highest concentrations in the blubber (n=4): 14-81 (median 53) ppm ww, then liver (n=4): 0.49- 5.2 (median 2.5) ppm ww and then in muscle tissue (n=4): 0.16- 1.0 (median 0.47) ppm ww (Andersen and Rebsdorff, 1976).

After the mass mortality in 1988/89 due to seal distemper virus, one harbour por-poise was found dead and the blubber was taken for analyses of organochlorine compounds. The animal died because of a carcinoma. DDT values were 3.14 ppm fwb (Beck et al., 1990).

DDT concentrations in blubber of immature harbour porpoises from the North Sea were below detection limit in a study from 1999. Nevertheless, measured concentrations ranged from 0.19 to 0.38 µg/g lipid in harbour porpoises of Arctic waters (Bruhn et al., 1999).

For more mature animals, concentrations above the detection limit (1.5 ng/g lipid) were found only in harbour porpoises from the Arctic Sea between 1986 and 1988: 0.31 µg/g lipid. A concentration of 1.18 µg/g lipid in the harbour porpoise blubber from the Danish North Sea and Greenland Sea was measured (Granby and Kinze, 1991).

The DDT levels measured in 21 harbour porpoise livers, from 1997 to 2000 found at the Belgian North Sea Coast, showed a median of 2.6 µg/g lipid (Covaci et al., 2002b).

Serum was taken at Dolfinarium Harderwijk, Netherlands from 19 harbour porpoises between 2006 and 2008 and showed values of 197 pg/ml to 2,330 pg/ml for DDT (Weijs et al., 2009a).

Between 1998 and 2001, thyroids and blubber of 57 harbour porpoises were collected. The measured DDT concentration in blubber of harbour porpoises collected on the German coast of North Sea was 255±252 ng/g lipid (Das et al., 2006).

Early measurements were taken between 1974 and 1976 of stranded or shot harbour seals from the German Wadden Sea. Blubber concentrations ranged from 2.2 to 27.2 mg/kg for total DDT (Drescher et al., 1977).

In 1979, organochlorine concentrations in several tissues from harbour seals were measured. The animals originated from the Dutch Wadden Sea and were all stranded animals. PCB was found to be the predominant compound, followed by DDT. Blubber concentrations of DDT were 0.51-25.4 mg/kg ww (n=7), liver 0.15-1.3 mg/kg (n=5), kidney 0.05- 0.76 mg/kg (n=2) and brain 0.06- 3.10 mg/kg (n=3). The highest DDT concentrations were found in blubber (Duinker et al., 1979).

In Norway, the DDT blubber concentration was measured of harbour seals in summer 1988. For this study 33 harbour seals under one year of age and 43 animals over one year were collected. DDT levels were two to four times lower than PCB values. The DDT concentration in below one-year old animals (n=33) was 2.6±1.3 mg/kg ww, in 17 females over one year old it was 3.1±1.5 mg/ kg ww and in 26 males 3.9±2.1 mg/kg ww (Skaare et al., 1990).

After the mass mortality in 1988/89, caused by seal morbilli virus, organochlorine compounds were suspected to weaken the immune system. For a study, five of these animals were chosen to perform a DDT analyses, beyond others. The mean concentration of DDT, found in blubber, was 0.37 ppm fwb (Beck et al., 1990).

Weijs et al. measured serum concentrations of 47 harbour seals from Helgoland and Lorenzenplate in the North Sea. The concentration of DDT was 50 pg/ml to 678 pg/ml (Weijs et al., 2009a).

Blubber from grey seals found dead at Farne Islands in 1988 was taken and used for determination of organochlorine compounds, such as DDT. Median concentration was 1.5 µg/g ww (n=3) (Law et al., 1989).

Blubber and liver samples were tested for DDT from 20 harbour seals from the German Wadden Sea to compare them with results of Greenland ringed seals. Blubber concentrations in harbour seals were 43.51 ng/g lw in comparison with 74.17 ng/g lw in ringed seals (Siebert et al., 2012b).

PFOS

From 1995 till 2000 tissue samples were taken of several organs of nine different marine mammal species stranded along the North Sea costs of Belgium, France and Netherland for toxicological analysis. The PFOS concentration in 48 livers of harbour porpoises ranged between 12-395 ng/g ww, in 43 kidneys from below 10-821 ng/g ww (Van de Vijver et al., 2003).

Measurements on liver tissues from bycaught harbour porpoises from Iceland, Norway, Denmark and the German Baltic Sea were taken. A large geographical difference was remarked: Iceland (n=7) had the lowest PFOS concentrations with 38±14 ng/g tissue and Norway (n=19) had the highest concentration: 213±195 ng/g tissue. Denmark (n=7) was 270±171 ng/g tissue. In comparison with the measurements of the southern North Sea (Van de Vijver et al., 2004), the PFOS concentrations are higher in the German Baltic Sea and in Denmark.

PFOS was the predominant compound in a study from 1991 to 2008 when temporal trends and differences of perfluoroalkylated substances in harbour porpoises were measured. In the North Sea values ranged between 204 and 2404 ng/g ww (n= 60) in liver tissue, (Huber et al. 2012).

The PFOS concentration in 24 livers of stranded harbour seals ranged between < 10 – 532 ng/g ww, in 22 kidneys they found concentrations of below 10 – 489 ng/g ww, with samples taken along the North Sea costs of Belgium, France and Netherland. It has to be mentioned that in this study, the highest concentrations of PFOS was found in livers of harbour seals (Van de Vijver et al., 2003).

Perfluorooctane sulfonate (PFOS) was the predominant compound in livers of the examined harbour seals in 1996 (n=63), with a median concentration of 3,520 ng/g ww (Ahrens et al., 2009). High concentrations were only found in livers of harbour seals younger than seven months. This suggests that a transplacental transfer and a transfer through lactation is likely. Over the years, the concentration of PFOS decreased to 1,077 ng/g ww in 2007, and a not detectable concentration in 2008, concerning the seals younger than seven months. In livers of animals older than seven months a median concentration of 480 ng/g ww was found in 2008. The reason for the decreasing concentration could be due to the replacement of the PFCs by shorter-chained and less bioaccumulative compounds. The total number of examined livers for that study in the period from 1988 to 2008 was 63 (Ahrens et al., 2009).

In a study that examined the livers of 68 northwest Atlantic harbour seals, PFOS concentrations were 98±104 ng/g ww in adult males (n=8), 100±56 ng/g ww in female adults (n=10) and 258±312 ng/g ww in pups in the years 2000-2007 (Shaw et al., 2009).

Due to the PDV epidemics in 2002, over 10,000 dead harbour seals were collected and dissected. Aside, tissue samples of different organs were taken for further toxicological analysis. The following median results are from the Dutch Wadden Sea. Kidney (n=18): 319.30 ng/g ww, liver (n=24): 160.53 ng/g ww, blubber (n=17): 44.83 ng/g ww, muscle (n=18): 24.54 ng/g ww, trachea-branchial muscle (n=1): 2,724.58 ng/g ww and spleen (n=4): 319.58 ng/g ww. This shows that the concentration of perfluorinated sulfonates differs greatly between the organs (Van de Vijver et al., 2005)

From 1995 to 2000, livers and kidneys of grey seals from the French, Belgian and Dutch North Sea coast were sampled. The median liver-PFOS concentration (n=6) was 88 ng/g ww and the kidney-concentration of PFOS was 81 ng/g ww (Van de Vijver et al., 2003).

PBDEs

Thron et al. (2004) compared the influence of age, sex, body-condition and region on PBDE levels in harbour porpoises from European waters. Samples were taken from 69 animals collected between 1997 and 2001. The authors compared PBDE concentrations of harbour porpoises with a poor nutritional status (2,230 ng/g lipid) and animals with a good nutritional status (180 ng/g lipid). They also compared the North Sea with the Baltic Sea results in juvenile harbour porpoises with a good nutritional status: North Sea: 180 ng/g lipid, Baltic Sea: 160 ng/g lipid (Thron et al., 2004).

Concentrations between 1,081 and 1,526 µg/g lipid in blubber of harbour porpoises from the German North Sea were found (Das et al., 2006). Another study on contaminant levels in harbour porpoises showed a median blubber PBDE concentration of 138 ng/g lipid in 61 examined animals (Beineke et al., 2005).

Moreover, a study conducted between 1999 and 2004 revealed PBDE blubber concentrations of 0.22 to 5.93 µg/g lw in harbour porpoises, stranded or bycaught in the Southern North Sea (Weijs et al., 2009b).

Harbour porpoises from Iceland (sample collection in 1992 and 1997) and Norway (sampled in 2000) had 94-97 ng/g lw medians for PBDE in pooled blubber samples in Iceland and 161 ng/g lw as median in animals from Iceland (Rotander et al., 2012).

Harbour seals stranded or bycaught and collected between 1999 and 2004 displayed PBDE-blubber concentrations between 0.09 and 1.15 µg/g lw, n= 28 (Weijs et al., 2009b).

TBT

TBT was found in all harbour porpoise livers examined for a study conducted in Danish and West Greenland waters. For the Danish North Sea, a median concentration of 196 mg/kg ww ±145 mg/kg ww SD was established (Strand et al., 2005).

Dioxin

The blubber tissue of five harbour seals and one harbour porpoise from the North Sea were examined. The mean of 2,3,7,8-TCDD in seals was 3.3 ppt fwb and in the porpoise <0,5 ppt, fwb (Beck et al., 1990). The mean on 1,2,3,7,8-PeCDD was 4.2 ppt fwb in harbour seals and in the harbour porpoise <0,5 ppt fwb. 1,2,3,4,7,8-HxCDD was in both 0.8 ppt fwb. The mean of 1,2,3,6,7,8-HxCDD in seals was 6.1 ppt fwb (1.4 ppt, fwb in the porpoise). Moreover, the mean of 1,2,3,7,8,9-HxCDD was <0.5 ppt in seals fwb, and 0.5 in the harbour porpoise. 1,2,3,4,6,7,8-HpCDD was measures in seals as mean: 1.6 ppt fwb, and in porpoise 22 ppt fwb. In seals the mean of OCDD was 5.6 ppt fwb and in the harbour porpoise 97 ppt fwb (Beck et al., 1990).

Blubber samples from harbour porpoises were collected at the German North Sea in 1994-1995. The median value of ∑PCDD/F was 10.3 pg/g lipid (n=6) and of *p,p'*-DDD was 0.39 µg/g lipid (n=10) (Bruhn et al., 1999).

Others

In the German North Sea, the median HCB concentration of harbour porpoises was 0.19 µg/g lipid. α-HCH concentration ranged from undetectable to 0.02 µg/g lipid and β-HCH concentration from 0.11 to 1.34 µg/g lipid (Bruhn et al., 1999). 21 livers of harbour porpoises from the Belgian North Sea were examined between 1997 and 2000. The median HCB concentration was 0.6 µg/g lipid, while the sum of HCHs in the median was determined as 0.2 µg/g lipid (Covaci et al., 2002b). In harbour porpoises from Danish and West Greenland collected between 1986 and 1988, a range of HCB concentrations between 0.01 µg/g lipid and 3.20 µg/g lipid was measured (Granby and Kinze, 1991).

5.4.2.2.2Metals

Mercury (Hg)

In 1988 and 1989 liver tissues of 14 grey seals, 28 harbour seals and 20 harbour porpoises were collected around the British Isles (Law et al., 1991). Mercury values up to 430 µg/g ww in grey seals and 150 µg/g ww in porpoises were detected (Law et al., 1991). Overall, nine seals and one porpoise showed values of 100 µg/g ww or higher (Law et al., 1991).

In an older study one harbour porpoise, one grey seal and a number of harbour seals from the German North Sea were examined. Following levels of mercury were found: harbour porpoise 3.3 mg/kg [ppm] in muscle and 28.0 mg/kg [ppm] in liver; grey seal: 19.5 mg/kg [ppm] in the liver. 1.5-160 mg/kg [ppm] in the liver and 1.0-10.0 mg/kg [ppm] in muscle of harbour seals were found (data relate to fresh weight) (Harms et al., 1978).

Samples of harbour porpoises (n≤4) and two grey seals from Cardigan Bay in 1988 were collected. The mercury values of harbour porpoises ranged in the liver between 0.61-0.63 µg/g ww, in the blubber 0.01-0.18 µg/g ww, in the melon 0.04 µg/g ww and in the muscle 0.22 µg/g ww. The concentration of mercury in grey seals was in blubber: 0.05 µg/g ww, in muscle: 0.78 µg/g ww, in liver 1.7/2.5 µg/g ww and in kidney: 1.1-1.3 µg/g ww (Morris et al., 1989).

Muscle, liver and kidney samples of 57 harbour porpoises from German waters (30 from the North Sea, 27 from the Baltic Sea) were collected over a period of three years (1991-1993). Total mercury levels ranged between 0.6 and 398 µg/g dw (dry weight) in muscle. In liver 0.6-449 µg/g dw and 0.5-160 µg/g dw in kidney were measured. Methylmercury ranged between 0.2-18.3 µg/g dw in muscle, 0.2-26.0 µg/g dw in liver and 0.1-23.5 µg/g dw in kidney of harbour porpoises (Siebert et al., 1999). Concentrations neither differed between stranded or by-caught harbour porpoises nor between sexes. Adjusted means for mercury and methylmercury were higher in porpoises from the North Sea (Siebert et al., 1999).

In 1974 samples of brain, liver, kidney, heart and spleen from 26 harbour porpoises from the east coast of Scotland were collected. Mercury levels in the liver were higher than in the other organs. Following ranges were found in males/ females: brain 0.11-0.46/ 0.08-3.04 µg/g ww, liver 0.29-10.6/ 0.28-15.9 µg/g ww, kidney 0.47-2.82/ 0.23-1.79 µg/g ww, heart 0.20-1.20/ 0.44-1.08 µg/g ww, spleen 0.13-1.85/ 0.12-1.01 µg/g ww (Falconer et al., 1983).

Between 1997-2003 liver (n=102) and kidney (n=102) samples of harbour porpoises from Scotland, France, Ireland, Netherlands and Spain were collected. The mean level of mercury in the liver was 17.3 µg/g ww, median 6 µg/g ww (±27.0 SD) and in the kidney 1.57 µg/g ww (±1.28 SD). Moreover, samples of two mothers with foetus were sampled, the foetus had particularly lower concentrations compared with those of the mothers (Lahaye et al., 2007).

Between 1994 and 2001 liver samples of 14 harbour porpoises form the German North Sea were collected (Das et al., 2004). The mean concentration of Mercury was 14 µg/g dw (±18 SD). Moreover, sampels from Belgium, France, Denmark, Iceland and Norway were measured and compared, as well as the body condition and lesions of the respiratory tract. Animals from northern France, Belgium and German North Sea had higher mercury levels than animals from Norway and the Baltic Sea (Das et al., 2004).

Liver tissues of 30 harbour porpoises from Scandinavian waters were analized. Finnmark and Barents Sea had the lowest mean 0.8 µg/g dw (± 0.1 SD) and southwest coast of Norway (North Sea) the highest mean 19.7 µg/g dw (± 9.8 SD) (Fontaine et al., 2007).

Four harbour porpoises from Danish Water, collected between 1972 and 1973 had a mean of Mercury of 0.7 ppm ww in blubber, 22 ppm ww in the liver and 1.9 ppm ww in the muscle (Andersen and Rebsdorff, 1976).

In 1998 and 1999 harbour porpoises (n=15) from the Danish North Sea were examinated. The mean of Mercury in the liver was 8.5 mg/kg ww (±10.2 SD) (Strand et al., 2005). Furthermore, also levels from animals from Greenland were measured.

In the years 1990-1994, the liver tissues of 86 harbour porpoises from England and Wales were examined. Animals deceased due to infectious disease (37) had a mean value of 20 mg/kg ww, whereas healthy harbour porpoises killed by acute physical trauma (49) had a mean value of 12.3 mg/kg ww (Bennett et al., 2001).

Full blood samples of 22 free-ranging harbour seals caught along the German coast were taken between 1997 and 2004. Total Mercury (T-Hg) levels varied from 0.04 to 0.56 µg/g fresh weight (fw) (43 to 611 µg/L) (mean concentration: 0.16µg/g fw). T-Hg concentrations in males and females were similar. T-Hg concentration in blood is correlated to body mass and length (Das et al., 2008).

In 1988 fur samples (n=47) of harbour seals form the German west cost were collected. The mean of mercury in hair was 33.5±38.5 µg/g ww (male pups: 22.1±20.3 µg/g ww, female pups: 21.2±23.4 µg/g ww; male adults: 25±16.1 µg/g ww, female adults: 55.9±61.3 µg/g ww) and in skin 0.40±0.43 µg/g ww (male pups: 0.12±0.08 µg/g ww, female pups: 0.34±0.18 µg/g ww; male adults: 0.44±0.31 µg/g ww, female adults: 0.59±0.67 µg/g ww) (Wenzel et al., 1993).

During 1974 and 1976 tissue (brain, liver, kidney) of harbour seals (n=63) were examinated. Following levels of mercury were found: liver: 1.5-160 mg/kg, kidney: 1.6-12.5 mg/kg, brain: 0.11-1.4 mg/kg (Drescher et al., 1977).

Samples from harbour seals from German and Danish Wadden Sea as well as from the Dutch Wadden Sea were collected in 1975 and 1976. The total mercury in the kidney ranged between 1.6 and 17.9 µg/g ww in seals from Germany and Denmark. In animals from the Netherland the range was between 0.7 and 28.2 µg/g ww. Moreover, the range in the liver was 1.1-751 µg/g ww (Germany and Denmark) and 0.9-573 µg/g ww (Netherlands). In the brain the range was 0.5-3.8 µg/g ww (Germany and Denmark) and 0.4-17.5 µg/g ww (Netherlands). The author also makes a difference between pups, subadults and adults. Furthermore, methylmercury was also measured (Reijnders, 1980).

On the Isle of May, Scotland, 21 female grey seals with pups were caught in early and late lactation in 2008. Moreover, the pups were also caught in the early and middle post weaning fast (Habran et al., 2013). Following mean values of mercury were found in the adult females during early lactation: Blood: 0.083 mg/kg ww; milk: 0.012 mg/kg ww and late lacation: blood: 0.104 mg/kg ww, milk: 0.021 mg/kg ww, hair: 7.7 mg/kg dw (Habran et al., 2013). The results of pups for the different times of sampling are also given in this paper.

Cadmium (Cd)

Liver tissues of 14 grey seals, 28 harbour seals and 20 harbour porpoises were collected around the British Isles, in 1988 and 1989 (Law et al., 1991). Cadmium levels up to 2.9 µg/g ww in seals and 1.2 µg/g ww in porpoises were detected (Law et al., 1991).

In the past, one harbour porpoise, one grey seal and some harbour seals from the German North Sea were examined. Harbour porpoise cadmium levels of 0.006 mg/kg [ppm] in muscle, 0.19 mg/kg [ppm] in liver and

0.95 mg/kg [ppm] in kidney were found. The grey seal had 0.021 mg/kg [ppm] in the liver and following levels were found in harbour seals: 0.002-0.08 mg/kg [ppm] in muscle, 0.01-0.21 mg/kg [ppm] in liver and 0.06-1.0 mg/kg [ppm] in kidney (data relate to fresh weight) (Harms et al., 1978).

In 1988 samples of harbour porpoises (n≤4) and two grey seals from the Cardigan Bay were collected. The cadmium values of blubber, muscle, liver and melon of harbour porpoises ranged between <0.5-0.7 µg/g ww. In one grey seal all samples (blubber, muscle, liver and kidney) were <0.06 µg/g ww. The other grey seal had a liver value of <0.07µg/g ww and in the kidney 0.08 µg/g ww (Morris et al., 1989).

In 1974, harbour porpoises (n=26) from the east coast of Scotland had following cadmium ranges: Brain < 0.05 µg/g ww, liver <0.05-0.94 µg/g ww, kidney 0.17-2.91 µg/g ww (male) 0.24-7.42 µg/g ww (female), heart < 0.05-0.08 µg/g ww (male) <0.05 µg/g ww (female), spleen <0.05-0.24 µg/g ww (male) <0.05 µg/g ww (female) (Falconer et al., 1983). The concentration in kidney and liver seems to increase with body length (Falconer et al., 1983).

In 1990-1994 liver tissue of 86 harbour porpoises from England and Wales was examined. The animals were categorised in two groups (cause of death: infectious disease - physical trauma). The mean differed from 0.24 (infectious disease) to 0.19 mg/kg (physical trauma) (Bennett et al., 2001).

Samples of 102 livers and kidneys from harbour porpoises from Scotland, France, Ireland, Netherlands and Spain were collected between 1997 and 2003. The mean level of Cadmium in the kidney was 1.32 µg/g ww (±1.81 SD). Furthermore, two pegnant harbour porpoises were sampled, the foetus showed particularly lower concentrations of cadmium than the mothers (Lahaye et al., 2007).

Between 1994 and 2001 tissue samples from the liver (n=14), kidney (n=12) and muscle (n=13) of harbour porpoises from the German North Sea were collected. In the livers a mean of 0.7 µg/g dw cadmium were found. The mean in the kidneys was 4 µg/g dw and in muscle <0.05 µg/g dw. Furthermore, sampels from adjected waters as well as the body condition and lesions in the respiratory tract were compared. Animals from Iceland and Norway had high cadmium levels due to cadmium-contaminated prey (Das et al., 2004).

Harbour porpoises (n=30) from Scandinavian waters were analyzed. The range of cadmium in the kidney was 0.1 – 15.9 µg/g dw. A significant difference among sites was detected (Fontaine et al., 2007).

Thirteen harbour porpoises from Danish water stranded in 1996 were analyzed. Cadmium ranged in the liver from under the detection limit up to 0.43 µg/g dw. In the kidney cadmium ranged between 0.1 and 4.15 µg/g dw, in muscle cadmium was not detected (Szefer et al., 2002). Moreover, animals from Greenland were examinated.

During 2003 and 2004 blood samples of 28 free-ranging harbour seals from the German Wadden Sea at Lorenzenplate (n=16) and from the Danish Wadden Sea at Rømø were taken. Cadmium concentrations were found in the range <0.12-3.10 µg/L (coefficient of variation: 488%) (Griesel et al., 2008). The values for the seals from Lorenzenplate were between <0.12-1.06 µg/L, whereas the values for the Danish seals ranged between <0.12-3.10 µg/L (Griesel et al., 2008). Harbour seals (n=13) from the Danish and German Wadden Sea were caught in 2003 and 2004 and blood was collected before animals were released. Cadmium concentrations were measured up to 3.10 µg/L (median: 0.16 µg/L), but some samples were under the detection limit (Kakuschke et al., 2005).

Following ranges of cadmium were found in harbour seals from the Dutch Wadden Sea: Blubber (n=3) <0.01-0.02 mg/kg ww, liver (n=8) 0.03-0.21 mg/kg ww, kidney (n=2) 0.15-0.17 mg/kg ww, brain (n=7) <0.01-0.4mg/kg ww, spleen (n=2) 0.04-0.09 mg/kg ww, heart (n=2) 0.06-0.47 mg/kg ww. Moreover, one fetus and a placenta were analized (Duinker et al., 1979).

In 1988, hair samples (n=47) of harbour seals from the west coast of Germany were analized. The mean of cadmium in male pups was: 0.09±0.03 µg/g ww, in male adults: 0.17±0.12 µg/g ww and in female pups: 0.13±0.11 µg/g ww and in female adults: 0.1±0.09 µg/g ww. Skin values were mainly below the detection limits (Wenzel et al., 1993).

Tissue samples of harbour seals from the German North Sea were collected during 1974 and 1976. The levels of cadmium ranged in the liver: 0.010-0.200 mg/kg, in the brain: 0.002-0.024 mg/kg and kidney: 0.06-0.380 mg/kg (Drescher et al, 1977).

A case report about a 7-month-old grey seal in 2003 showed a mean value of cadmium of 0.28 µg/L in blood (Kakuschke et al., 2006).

During the early and late lactation female grey seals (n=21) with pups (n=20) were caught the Isle of May, Scotland in 2008. Furthermore, the pups were also caught in the early and middle post weaning fast (Habran et al., 2013). All values in blood, milk and blubber were under the limit of detection. The mean of cadmium in hairs of adult females during late lactation was 0.27 mg/kg dw and in pups 0.02 mg/kg dw (Habran et al., 2013).

Lead (Pb)

Lead concentrations measured in 62 (14 grey seals, 28 harbour seals, 20 harbour porpoises) liver tissue samples showed that only six (five grey seals, one harbour porpoise) had values between 0.6-1.8 µg/g ww in grey seals and 4.3 µg/g ww in a harbour porpoise originating from waters around the British Isles (Law et al., 1991). The others had concentrations between <0.6 and <0.8 µg/g ww (Law et al., 1991).

Harms et al. (1978) measured levels of lead in one harbour porpoise, one grey seal and some harbour seals from the German North Sea. The harbour porpoise had 0.05 mg/kg [ppm] in muscle, 0.35 mg/kg [ppm] in liver and 0.17 mg/kg [ppm] lead in the kidney. The grey seal showed 0.31 mg/kg [ppm] in the liver and following levels were found in harbour seals: 0.03-0.10 mg/kg [ppm] in muscle, 0.09-0.74 mg/kg [ppm] in liver and 0.08-0.60 mg/kg [ppm] in kidney (data relate to fresh weight).

In 1988, samples of harbour porpoises and grey seals from west Wales were collected. The lead values of blubber, muscle, liver and melon of harbour porpoises ranged between <0.5-<0.7 µg/g ww. In grey seals the range was between <0.6 and <0.7 µg/g ww in all tissues (blubber, muscle, liver and kidney) (Morris et al., 1989).

The lead concentration of tissue samples (brain, liver, kidney, heart, spleen) of harbour porpoises from Scotland were below the detection limit (Falconer et al., 1983).

86 liver tissue samples from harbour porpoises from English and Welsh waters collected in 1990-1994 were analysed (Bennett et al., 2001). The lead mean value was 0.13 mg/kg in one group (died of infectious disease) and 0.15 mg/kg in another (died of physical trauma) (Bennett et al., 2001).

During 1972 and 1973 tissue samples of harbour porpoises (n=4) from Danish Waters were collected. The mean of lead in blubber was 6.0 ppm ww, in the liver the mean was 3.5 ppm ww and in muscle 3.3 ppm ww (Andersen and Rebsdorff, 1976).

Lead concentrations in harbour porpoises collected in 1996 from Danish waters (n=13) ranged from "not detected" up to 0.32 µg/g dw in the kidney, in the liver lead was not detected (Szefer et al., 2002).

Fresh blood samples of 28 harbour seals from the German and Danish Wadden Sea were collected during 2003 and 2004. Lead concentration ranged from <0.02-4.52 µg/L. The median was 0.98 µg/L in German seals (around the Lorenzenplate) and 0.4µg/L in Danish seals. The coefficient of variation is 178% (Griesel et al., 2008). In another study, harbour seals (n=13) from the Danish and German Wadden Sea were caught in 2003 and 2004 and blood was collected before animals were released. Lead was measured up to 2.0 µg/L (median: 0.09 µg/L), but also one sample was under detection limit (Kakuschke et al., 2005).

Harbour seals from the Dutch Wadden Sea had following ranges of lead: blubber (n=3) <0.05-1.0 mg/kg ww, liver (n=8) <0.05-2.3 mg/kg ww, kidney (n=2) 0.16-0.23 mg/kg ww, brain (n=7) <0.05-2 mg/kg ww, spleen (n=2) 0.16-0.40 mg/kg ww, heart (n=2) 0.29-0.61 mg/kg ww. Morever, one fetus and a placenta were analysed (Duinker et al., 1979).

Hair samples (n=47) of harbour seals were collected in 1988 along the German west coast. The mean of lead in the hair of males was: 0.5±0.1 µg/g ww (pups) and 0.6± 0.3 µg/g ww (adults). In females the mean was: 1.1±0.8 µg/g ww (pups) and 0.6±0.3 µg/g ww (adults) (Wenzel et al., 1993). In the skin, the lead concentration was mainly below detection limit, just two pups showed higher lead concentrations (0.3 µg/g and 0.1 µg/g) (Wenzel et al., 1993).

In a study of Drescher et al. (1977) tissue samples (liver, kidney, brain) of harbour seals from the German Wadden Sea (1974-1976) were collected. The liver values of lead were 0.10-0.57 mg/kg, kidney: 0.14-0.55 mg/kg, brain: 0.04-0.20 mg/kg.

In 2003 the mean value of a young grey seal was 0.13 µg/L in blood (Kakuschke et al., 2006).

In 2008 female grey seals with pups were caught during early and late lactation at the Isle of May, Scotland. Pups were also caught in the early and middle post weaning fast (Habran et al., 2013). In females following mean values of lead were measured: early lactation: blood 0.009 mg/kg ww, milk 0.022 mg/kg ww, blubber 0.069 mg/kg ww and during late lacation: blood 0.007 mg/kg ww, milk 0.019 mg/kg ww, blubber 0.111 mg/kg ww and hair 2.2 mg/kg dw (Habran et al., 2013).

Arsenic (As)

Liver samples (n=14) of harbour porpoises from the North Sea were examined for arsenic. The range was 217-899 (median 421) µg/kg ww (Kuenstl et al., 2009). Moreover, blood of 81 free ranging harbour seals from the German Wadden Sea was analysed. The range was 46-780 µg/L. Furthermore, blood from seals in captivity during a special diet as well as blood of young pups was analysed (Kuenstl et al., 2009). The predominan species was arsenobetaine (in blood, urine, gastric juice of seals and urine in harbour porpoises) (Kuenstl et al., 2009).

In 2003 and 2004 blood was taken from 28 harbour seals from the Wadden Sea. Between 42 and 592 µg/L arsenic was measured in seals from the Lorenzenplate (Germany). Harbour seals from Rømø (Denmark) had values from 118 to 316 µg/L. The coefficient of variation is 62% (Griesel et al., 2008). In another study, 13 harbour seals from this region in the same year were analysed. Arsenic was measured at 69.3-235 µg/L (Kakuschke et al., 2005).

The mean arsenic level in blood of a young grey seal was 108 µg/L in 2003 (Kakuschke et al., 2006).

5.4.2.3 German Baltic Sea and adjacent waters

5.4.2.3.1 Organic substances

PCBs

In a study published in 1999 blubber samples of harbour porpoises from the German North Sea and Baltic Sea coast were taken. They found median concentrations of 14.91 ng/g lipid in harbour porpoises of the Baltic Sea (Bruhn et al., 1999).

Blubber PCB-concentrations in male harbour porpoises revealed concentrations of 160 ng/g lipid (standard deviation= s.d.: 80 ng/g) in 13 immature animals and 460 ng/g lipid (s.d.: 290 ng/g) in four mature animals from the Swedish Baltic Sea, the Kattegat-Skagerrak Seas and the west coast of Norway. Samples have been collected 1978-1993 (Berggrena et al., 1999).

Between 1998 and 2001, PCB concentrations in the blubber of harbour porpoises from the Baltic Sea were recorded. 17 animals had a median PCB concentration of 8247 ng/g lipid (s.d.: 7,949 ng/g) (Das et al., 2006).

DDT

In 1999, blubber concentrations of 150 ng/g lipid (s.d.: 18 ng/g) were measured in 11 immature harbour porpoises and 116 ng/g lipid (±134 ng/g SD) in four mature animals from the Baltic Sea (Berggrena et al., 1999).

Between 1998 and 2001 the DDT blubber-concentrations of 17 harbour porpoises from the Baltic Sea were investigated and amounted to 428 (±559 ng/g SD) ng/g lipid (Das et al., 2006).

PBDE

Thron et al. (2004) compared the influence of age, sex, body-condition and region on PBDE levels in harbour porpoises from European waters. Samples of 69 animals were taken between 1997 and 2001. The results of juvenile harbour porpoises from the Baltic Sea with a good nutritional status reached a concentration of 160 ng/g lipid (Thron et al., 2004).

PFOS

In 2004 a study was published about the trends of PFOS and related compounds in liver tissues of 41 bycaught harbour porpoises. The animals were caught by accident along the German Baltic coast (n=7). The measured PFOS concentration ranged between 534±357 ng/g (Van de Vijver et al., 2004).

PFOS was the predominant compound in a study from 1991 to 2008 when temporal trends and differences of perfluoroalkylated substances in harbour porpoises were measured. In the Baltic Sea values ranged between 159 and 2425 ng/g ww (n= 60). Values measured in animals from the Baltic Sea were higher than from the North Sea (Huber et al. 2012).

Livers of grey seals from the Bothnian Bay in the Baltic Sea have been collected by the Finnish Game and Fisheries Research Institute. 12 livers from male and 15 livers from female grey seals have been collected. The males had higher PFOS-values than the females. Males showed 148- 360 ng/g ww and femals 140- 290 ng/g ww (Kannan et al., 2002).

TBT

A study from Poland measured the TBT concentrations in livers of stranded and bycaught harbour porpoises. Samples were taken between 1996 and 2003. 14 livers were sampled, the TBT concentration ranged between 0.42 and 1.65 mg/g ww (Ciesielski et al., 2004).

The same study from Poland measured the TBT concentrations in livers of stranded and by-caught grey seals. Samples were taken between 1996 and 2003. Two livers were sampled, the TBT concentration ranged between 0.03 and 0.05 mg/g ww (Ciesielski et al., 2004).

Dioxin

The concentration of chlorinated contaminants in blubber seems to differ with respect to the age and sex of marine mammals.

During 1978-1993, 47 blubber samples of male harbour porpoises from the Swedish Baltic Sea, the Kattegat-Skagerrak Seas and the west coast of Norway were examined. The immature individuals from the Baltic had significant higher ∑PCDD/F and 1,2,3,7,8-PnCDD mean levels than those from Kattegat-Skagerrak, ANOVAs (Berggren et al., 1999). The mean concentrations of ∑PCDD/Fs in 1985-1993 of immature harbour porpoises from the Baltic Sea was 13 pg/g (±3.6 SD) lw, and in mature individuals between 1988 and 1989 the levels were 36 pg/g (±26 SD) lw. While in Kattgat and Skagerrak Seas the mean concentration and standard derivation was during 1989 and 1990 in immature 9.2 pg/g (±5.1 SD) lw, whereas in matures 16 pg/g (±3.7 SD) lw in 1988-1990 and 19 pg/g (±7.3 SD) lipid in 1978-1981. Mature harbour porpoises from the west coast of Norway had a mean concentration of 12 pg/g (±4.8 SD) lipid in 1988-1990 (Berggren et al., 1999).

The median value (∑PCDD/F) of four blubber samples of harbour porpoises from the German Baltic Sea was 6.2 pg/g lipid. The median of *p,p'*-DDD was 1.94 µg/g lipid (Bruhn et al., 1999).

5.4.2.3.2 Metals

Mercury (Hg)

Samples (muscle, liver, and kidney) of 57 harbour porpoises from German waters (27 Baltic Sea, 30 North Sea) were collected during 1991-1993. In harbour porpoises the total mercury levels ranged between 0.6 and 398 µg/g dw in muscle, 0.6 and 449 µg/g dw in liver and 0.5 and 160 µg/g dw in kidney. Methylmercury was measured between 0.2-18.3 µg/g dw in muscle, 0.2-26.0 µg/g dw in liver and 0.1-23.5 µg/g dw in kidney (Siebert et al., 1999). There were no differences in concentrations between stranded and by-caught harbour porpoises or between the sexes. Mercury as well as methylmercury adjusted means were lower in porpoises from the Baltic Sea (Siebert et al., 1999).

Liver samples of nine habour porpoises from the German Baltic Sea were collected between 1994 and 2001. The mean mercury concentration was 4.5 µg/g dw, median 4.1 µg/g dw (±3.6 SD). Moreover, sampels from Belgium, France, Denmark, Iceland and Norway were measured and compared, as well as the body condition and lesions of the respiratory tract. Animals from northern France, Belgium and German North Sea had highest mercury levels compared with animals from Norway and the Baltic Sea. Furthermore, mercury was linked to age. There was no significant variety between body conditions (Das et al., 2004).

Harms et al. (1978) measured mercury in two harbour porpoises in the German Baltic Sea (muscle: 0.15-0.92 mg/kg [ppm]; liver: 0.70-2.5 mg/kg [ppm], data relate to fresh weight).

Harbour porpoises (n=20) from inner Danish waters were collected in 1998 and 1999. The mean of mercury in the liver was 6.4 mg/kg ww (±20 SD) (Strand et al., 2005).

Mercury ranged in the liver of polish harbour porpoises (n=14) from 1.53-217 µg/g dw. In grey seals (n=5) the level of Mercury was 0.55-557 µg/g dw and of one harbour seal 75.4 µg/g dw. The samples were collected between 1996 and 2003 (Ciesielski et al., 2006).

Cadmium (Cd)

Tissue samples of liver (n=9), kidney (n=9) and muscle (n=9) of harbour porpoises from the German Baltic Sea were collected between 1994 and 2001. In the liver a mean of 0.2 µg/g dw were found. The mean in the kidney was 1.1 µg/g dw and in muscle <0.05 µg/g dw (Das et al., 2004).

In the past the level of cadmium was measured in two harbour porpoises from the German Baltic Sea (muscle: 0.002 mg/kg [ppm], liver: 0.023-0.025 mg/kg [ppm], kidney: 0.077 mg/kg [ppm], data relate to fresh weight) (Harms et al., 1978).

Cadmium levels in tissues of harbour porpoises (n=24) from the Polish Baltic Sea were measured (1993-1996). Cadmium levels were between 0.02 and 0.2 µg/g dw in the liver and 0.06-1.29 µg/g dw in the kidney, in the muscle Cadmium was not detected (Szefer et al., 2002).

In a study from Ciesielski et al. (2006) 14 harbour porpoises and two grey seals from the Polish Baltic Sea in 1996-2003 were analised. The range of cadmium in the liver of harbour porpoises was 0.15-0.34 µg/g dw and in the grey seals 0.12-0.19 µg/g dw.

Lead (Pb)

Lead was found in two harbour porpoises from the German Baltic Sea. Levels of 0.03/0.07/0.05 mg/kg [ppm] were found in muscle, 0.17/0.43/0.35 mg/kg [ppm] in the liver, and levels of 0.15/0.17 mg/kg [ppm] was found in the kidneys (data relate of fresh weight) (Harms et al., 1978).

Between 1993 and1996, 24 harbour porpoises from the Polish Baltic Sea were collected. Lead ranged from “not detected” up to 0.4 µg/g dw in the liver and up to 0.45 µg/g dw in the kidney (Szefer et al., 2002).

Lead in marine mammals from the Polish Baltic Sea could not be found in a study of Ciesielski et al. (2006), during 1996 and 2003.

Arsenic (As)

Liver tissue samples of eight harbour porpoises from the Baltic Sea as well as four from the River Elbe were analised. The range was 193-380 µg/kg ww in the Baltic Sea and 137-392 µg/kg ww in the River Elbe (Kuenstl et al., 2009).

In a study by Ciesielski et al. (2006) no arsenic could be found in the livers of marine mammals from the Polish Baltic Sea between 1996 and 2003.

5.4.2.4 Effects of chemical pollutants on marine mammals

5.4.2.5 Harbour porpoises

5.4.2.5.1 General health

During the necropsies between 1991 and 1996, most of the harbour porpoises from the North and Baltic Seas were in a good or moderate nutritional state, but 21 of 102 were emaciated. 46% of 130 harbour porpoises in this study were bycaught. Pneumonia was assumed to be the cause of death in 46% (Siebert et al., 2001). Furthermore, 94% of by-caught animals and 77% of stranded animals showed oedema. Moreover, a high burden of parasitic infection in different organs (lung, heart, liver, pancreas, ear, gastrointestinal system) was detected. For example, lungworms were detected in all harbour porpoises older than one year. Helminthic infestation in the gastrointestinal system and trematodes in the liver were also observed (Siebert et al., 2001).

In a comparative study including only bycaught harbour porpoises from Norwegian and Icelandic waters, none of the 22 animals were emaciated (Siebert et al., 2006b). Overall, parasitic infections in the lungs of harbour porpoises of Norwegian and Iceland were mild or moderate whereas some (up to 18%) animals from German waters had severe infection with almost complete obstruction of the airways (Jepson et al., 2000; Siebert et al., 2001; Jauniaux et al., 2002). High concentrations of hazardous substances may lead to increased susceptibility for severe parasitic and bacterial infections (Siebert et al., 1999; Siebert et al., 2001). In conclusion, harbour porpoises around Greenland, Iceland and Norway are healthier than porpoises from the North and Baltic Seas (Siebert et al., 2006b). Arctic porpoises have lower levels of chemical pollutants than porpoises from the North and Baltic Sea (Bruhn et al., 1999; Thron et al., 2004).

Furthermore, high input of noise pollution and other anthropogenic activities in German waters may permanently raise stress levels, which negatively affect the immune and hormonal systems of marine mammals.

Jepson et al. (2005) underlines the causal relationship between PCB exposure and infectious disease mortality with their research. They examined total PCB levels in porpoises from UK waters and showed that harbour porpoises that died from infectious diseases or parasitic infection had significantly higher concentrations of PCBs than those which died because of physical trauma.

Porpoises with lesions in the respiratory tract showed higher zinc and ferric levels, but lower copper concentrations (Das et al., 2004). Also, Bennett et al. (2001) showed that porpoises that had died due to infectious diseases had higher mercury concentrations in the liver and the Hg:Se ratio was significantly higher compared with porpoises that had died from physical trauma.

In a bacteriological study the bacterial flora of harbour porpoises from the German North Sea, Baltic Sea, Greenland, Iceland and Norwegian waters collected between 1988 and 2005 were compared (Siebert et al., 2009). Samples from Greenlandic and Icelandic porpoises had less bacterial growth and fewer associated pathological lesions than animals from the more polluted German North and Baltic Seas (Siebert et al., 2009). Comparison between harbour porpoises from Greenland and from the German North and Baltic Seas showed that animals from Greenland suffer from fewer inflammatory lesions and infectious agents (Siebert et al., 2001; 2009; Wunschmann et al., 2001). Porpoise morbilli virus antibodies were found in animals of the different regions, but no antigen was found in lung tissue (Müller et al., 2000; Wunschmann et al., 2001).

5.4.2.5.2Reproduction system

PCB concentrations of blubber of female harbour porpoises from the European Atlantic coast frequently exceeded the threshold at which consequences on fertility are probable (Pierce et al., 2008).

Murphy et al. (2010) assessed the effect of persistent organic pollutants (POP) on the reproduction of common dolphins and harbour porpoises. This resulted in an association between high POP burdens and a decreased ovarian scar number and hence it is assumed that high contaminant levels may hinder ovulation and may be responsible for a period of infertile ovulations prior to a successful pregnancy.

5.4.2.5.3Immune system

Bennett et al. (2001) carried out 86 post-mortem investigations of harbour porpoises of stranded individuals along British coasts. They identified a significantly higher concentration of mercury, selenium and zinc in the liver and increased Hg:Se molar ratio in porpoises that died because of infections in comparison to animals that died as a result of physical trauma.

A correlation between PCB and PBDE burden and thymic atrophy and splenic depletion was detected in harbour porpoises. These pathological findings are related to an impaired health status (Beineke et al., 2005).

5.4.2.5.4 Endocrinium

The thyroid glands of 57 harbour porpoises from German and Danish (North and Baltic Sea), Norwegian and Icelandic coasts were morphologically investigated and the blubber tested for PCB, among other compounds. Between 30% and 38% of the thyroids from the German (North Sea and Baltic Seas) and Norwegian coasts presented severe interfollicular fibrosis and a high number of large follicles (Das et al., 2006). In a study from 2005, Beineke et al. showed that thymic atrophy and splenic depletion were significantly correlated to increased PCB and PBDE levels (Beineke et al., 2005).

5.4.2.6 Harbour seals

5.4.2.6.1 General health

A stranding network for monitoring in Schleswig-Holstein was established in Schleswig-Holstein in 1990 to evaluate the health status of harbour seals (Siebert et al., 2007). The general health status of harbour porpoise can be measured by the results of life monitoring, pathological findings and the population size. Necropsy as well as histopathological, immunohistochemical, microbiological and parasitological examinations were performed on 355 carcasses of harbour seals from 1996 to 2005 (Siebert et al., 2007). The main findings were located in the respiratory and alimentary tracts. The most common cause of death was bronchopneumonia (Siebert et al., 2007).

PDV caused mass mortalities in 1988 and 2002. This disease killed about 30,000 individuals during the second outbreak (Härkönen et al., 2006). By means of life monitoring, a present antibody titer of PDV was detected. Currently, most harbour seals' antibody titer is negative (Wehrmeister et al., in prep.). The question of whether the PDV epidemic in 1988 targeted seals with the highest level of organochlorine was explored (Härkönen et al., 2006). Since the last outbreak of PDV in 2002, the population size has been increasing continuously (Common-Wadden-Sea-Secretariat, 2013). In a new study, serum of seals (mainly harbour seals, n=423) from Dutch coastal waters between 2002 and 2012 were examined for antibodies against PDV and CDV (Bodewes et al., 2013). The results suggest the majority of seals are not immune to PDV infection currently. A new outbreak of PDV may cause 82% infection of the population and a mass-mortaliy of >50% (Bodewes et al., 2013).

5.4.2.6.2 Reproduction system

Several studies suggest that increased PCB values are the cause of aggrieved reproduction in seals in the Wadden Sea and the Baltic Sea (Reijnders, 1980; Reijnders, 1986). Reijnders (1986) detect reproductive failure, especially during the implantation period, as a result of PCB and DDT exposure within a controlled feeding experiment on captured harbour seals. A declining reproduction ratio was determined for seals fed on contaminated fish from the Wadden Sea at an average total-PCB level of 25-27 µg/g lipid per day (Reijnders, 1986). Normal reproductive rates occurred in the control group at mean PCB levels of 5-11 µg/g lipid (Reijnders, 1986). Kannan et al. (2000) introduced A Σ-PCB level of 17 µg/g lipid in blubber as a threshold level for effects on reproduction in marine mammals.

PCB and dichlorodiphenylethane (DDE) methyl sulfone (MSF) metabolites occupy high appetence for binding a receptor protein in the uterus (uteroglobin, UG). Hence selective bioaccumulation of MSFs occurs in the uterus and can lead to implantation failure or abortion (Muckerjee et al., 1999; Troisi et al., 2001).

5.4.2.6.3Immune system

Pollutants are likely to promote mortality by compromising the immune system. Mercury, cadmium and lead, in particular, effect immune functions due to their high toxicity and accumulation characteristics (Wenzel et al., 1993).

Das et al. (2008) detected an immunosuppressive effect of Methyl-Hg (0.2 and 1 µM) on immune functions in harbour seals using an in vitro model with mitogen-stimulated peripheral blood mononuclear cells measuring the expression of Interleukin 2 (IL-2), IL-4 and TGF-β. Low concentrations (0.2 and 1 µM) of methyl-Hg are also capable of reducing the DNA and RNA synthesis and lymphocytes abundance, their viability and metabolic activity. The concentration of 1 µM methyl-Hg is termed to be the critical concentration for reduced lymphocyte activity, proliferation and survival.

While testing the T-lymphocyte response to metals in harbour seals 7 of 11 investigated indi-viduals showed metal hypersensitivities (molybdenum (Mo), titanium (Ti), nickel (Ni), chromium (Cr), aluminium (Al), lead (Pb), and tin (Sn)). Therefore, it is assumed that, as a result of a steady exposure to contaminants, seals (like humans) can evolve a metal hypersensitivity subject to individual immunity, age, term of exposure to pollutants and diet and nutrition status (Kakuschke et al., 2005).

Kakuschke et al. (2008a) examined the cellular immunity and the impact of several pollutants on newborn harbour seals. 12 of 20 tested metals, in particular beryllium (Be), cadmium (Cd), ethyl mercury (EtHg), methyl mercury (MeHg), and tin (Sn), were identified to inhibit the lymphocyte proliferation in newborn harbour seals.

In a feeding study with herrings from uncontaminated Atlantic Ocean or herrings from con-taminated Baltic Sea, seals fed by contaminated prey showed an immunotoxic risk (impairment of natural killer cell activity and of T-lymphocyte function, Ross et al., 1996; de Swart et al., 1996). Fasting (2 weeks) does not lead to a major additional immunotoxic risk of seals with high contaminate burdens nor an increased level in the blood (de Swart et al., 1966).

A study was performed, feeding herring of less contaminated waters (Atlantic Ocean) and more polluted waters (Baltic Sea) to harbour seals (Ross et al., 1996). The results were a contaminant-related suppression of delayed-type hypersensitivity and antibody responses (Ross et al., 1996).

Moreover mercury (Hg)-compound, aluminium (Al), berryllium (Be), cadmium (Cd) and in one test also silver () and cobalt (Co) have an immunosupressive/ cytotoxic effect on proliferation. But metals can also stimulate lymphocytes. The influence of metals may depend on the immune status of the animal (Kakuschke et al., 2008).

5.4.2.6.4Endocrinum

Brouwer et al. (1989) showed a decrease of total and free thyroxin (TT4 and FT4) and triiodothyronin (TT3) in common seals, fed with PCB-contaminated fish from the Wadden Sea as a sign of change in thyroid tissue.

5.4.2.6.5Other organ systems

A study using primary hepatocytes from harbour seals investigated the impact of environmentally relevant concentrations of PCBs (PFOS and an Aroclor mixture) on cell viability (Korff et al. in prep.). In a two-step biopsy perfusion method (Reese and Byard 1981) hepatocytes from fresh liver tissue were isolated from terminally ill harbour seal pups from the North Sea. During cultivation, the hepatocytes were exposed to PCBs with known (hepato-) toxic mode of action and in concentrations corresponding to those found in harbour seal tissue. Cell viability and maintenance was evaluating using three parameters, the activity of mitochondrial dehydrogenases (XTT assay), the membrane integrity (LDH release) and the maintenance of hepatospecific urea synthesis (Korff et al. in prep.). Seal hepatocytes were harvested for subsequent analysis of sublethal effects on the proteome and genome (Behr et al. 2008). Although main aspects of cell viability and specific metabolism of primary seal hepatocytes were not reduced by the used pollutant concentrations, urea synthesis decreased slightly during cultivation (Korff et al. in prep.). Primary hepatocytes may be an ideal tool to develop a cell culture model in which pollutant exposure can be monitored closely under controlled experimental conditions. In subsequent proteome analyses ten proteins from approx. l60/gel modified their protein expression levels (Behr et al. 2008); it was shown that protein expression patterns enable to discriminate between with pollutants incubated cells and negative controls. Some of the up-regulated proteins were putatively identified to belong to the group of cytochrome P450 enzymes (Behr et al. 2008) and may contribute to the optimization of an effect-oriented monitoring strategy concerning the influence of pollutants to marine mammals.

5.4.2.7 Grey seals

5.4.2.7.1 General health

Grey seal carcasses are collected through the stranding networks in Schleswig-Holstein and Mecklenburg-Western Pommerania and necropsies are conducted. Grey seals are hosts for several species of parasites, e.g. a parasitic roundworm called codworm (*Pseudoterranova decipiens*), other nematodes parasitizing in grey seals are contracaecum, anisakis, dioctophyme and otostrongylus (King, 1991; Bäcklin et al., 2003). Parasitic infestation might lead to severe bronchopneumonia or stomach and intestinal ulcers. Severe secondary systemic and bacterial infections are also possible consequences (Siebert et al., in prep.). In general, grey seals seem to be less succeptable to Morbillivirus infection than harbour seals. But they are suspected to be carriers of this disease (Härkönen et al., 2006).

During 1977 and 1983 grey seal females (n=61) were examined for their health status (Bergman and Olsson, 1985). Several changes in different organe systems were found (e.g. uterine stenosis and occlusions, benign uterine tumors, adrenocortical hyperplasia, hyperkeratosis, claw lesions and intestinal ulcers) and associated with contaminants such as organochlorines (Bergman and Olsson, 1985). The appearance of these diseases is summarized as: “Baltic seal (disease) complex”. In a following study (1977-1996) the number of claw lesions decreased over the study period. The adrenocortical hyperplasia and gynecological health showed this decreasing trend also. Contrary, colonic ulcers increased (Bergman, 1999; Bäcklin et al., 2003).

In 1960-1969 and 1971-1985 grey seals were collected and compared to seals found before 1950. An increase of skull bone lesion compared to the time before 1950 was observed. These changes belong to the disease complex of grey seals (Bergman et al., 1992). In light of this, bone mineral density of males was studied in three groups: 1850-1955 (very low organochlorid in the environment), 1965-1985 (high organochlorid) and 1986-1997 (decreasing organochlorid). The lowest trabecular bone mineral density was found in the years with high organochlorie burdens. The cortical bone mineral density decreased over the years (Lind et al., 2003).

5.4.2.7.2 Reproduction system

Six out of nine female grey seals from the Baltic and two harbour seals from the Swedish west coast showed uteri changes such as stenosis or occlusion. The resulting low reproductive rate was associated to PCB burdens (Helle et al., 1976). The Baltic seal complex may be influenced by the increased organochlorine levels, e.g. PCB Uterine stenosis and occlusions (30%) and benign uterine tumours have been observed in grey seals from the Gulf of Bothania and the Baltic proper (one animal was kept in an enclosure) (Bergman and Olsson, 1985). In another study 64% of female grey seals had uterine leiomyomas, mainly in the uterine corpus in the examination time 1975-1997 (Bäcklin et al., 2003). Moreover 65% of the animals with leiomyomas had no corpora contained in the ovaries. Leiomyomas seem to be associated directly or indirectly with organochlorines (Bäcklin et al., 2003).

A correlation between PCB and DDT and uterine stenosis and occlusions has been observed in grey seal populations in Liverpool Bay and were identified to eventuate in infertility (Baker, 1989).

5.4.2.7.3 Immune system

Mercury decreased the immune response (phagocytosis and lymphoblast transformation) in grey seal pups (Lalancette et al., 2003).

In a case report a young grey seal was hypersensitive against Nickel and Beryllium. It showed a helper T cell 1 (Th1)/Th2 imbalance and proliferation of lymphocytes (Kakuschke et al., 2006).

IL6 like activity was found in plasma of sick grey and harbour seals, but not in healthy seals. It seems to be that the IL6- like activity and the components of the leucocyte derived supernatants are similar to other mammals (King et al., 1993).

5.4.2.7.4 Endocrinum

In an *in vitro* study, the influence of different contaminants (PCB, Me Hg, As, Cd, Se) of grey seals steroid biosynthesis was tested. A correlation between contaminants and altered biosynthesis was found (Freeman et al., 1977).

Bergman and Olsson (1985) suggest a disease complex, caused by organochlorine interference with the endocrine system.

5.4.2.8 Other species

5.4.2.8.1 General health

The effect of different congeners of PCBs and DDTs on beluga (*Delphinapterus leucas*) splenocytes proliferation might be an allusion to the immunosuppressive consequence of organochlorines (De Guise et al., 1998).

During 1983-1990, beluga whales from the St. Lawrence population were examined. Non- neoplastic ulcera (31% gastric and oral 7%) as well as cancer was detected. At which the prevalence of cancers was higher than in other populations. Benzo[a]pyrene, which are in the sediment of St. Lawrence, are carcinogenic (Martineau et al., 1994).

5.4.2.8.2 Reproduction system

PCBs concentration in blubber of female common dolphins from Irish, UK, Spainish and French coasts were above the threshold, so that consequences on reproduction are likely (Pierce et al., 2008).

Higher levels of PCB and DDT were found in non-pregnant compared to pregnant ringed seals. These contaminants may lead to reproduction disturbances (Helle, 1976).

In ringed seals 70% had stenosis and occlusions in the uteri. An influence of pollutant burdens was assumed (Bergman and Olsson, 1985).

5.4.2.8.3Immune system

In a study from 1991 blood of bottlenose dolphins (*Tursiops truncatus*) from the west coast of Florida was taken. The results showed a reduced immune resonse is associated with increased concentration of contaminants, such as PCB and DDT (Lahvis et al., 1995)

Declining splenocyte and thymocyte proliferation in beluga whales was observed in vitro after exposure of the highly concentrated metals (mercury chloride, cadmium chloride (De Guise et al., 1996).

5.4.2.8.4Other organ systems

The high level of organochlorine pollutants burdening the St. Lawrence estuary is associated with changes in different organs in beluga. They set immunosuppression in relation to organochlorine bioaccumulation in beluga (De Guise et al., 1995).

There are just small differences in the concentration levels of metal from seals from different areas. So, the environmental changes seem to be just partly reflected (Duinker et al., 1979).

5.4.3 Lack of Data

5.4.3.1 On Pollutant levels

A large number of studies on chemical pollution in marine mammals have been conducted worldwide. As a result, several mean values and ranges for different hazardous substances have been published. However, the compatibility of this data is difficult. There is a lack of continuous data on chemical pollutant levels in marine mammals. This is also true for German waters. Additionally, recent levels of pollutant concentrations in all three marine mammals in the German waters are not available. This includes also lacking information on the most relevant chemical pollution for marine mammals in German waters, age and sex relation of chemical burden as well as regional differences. Furthermore, no analyses have been conducted on any pharmaceutical substances (antibiotics, contraception pills) in marine mammals from German waters.

5.4.3.2 For effects of pollutants

Harbour porpoise

Some tests for the investigation of the immune system of harbour porpoises have been developed in the ninethees. First studies on the influence of chemical pollutants on this organe system displayed that individuals with higher PCB and PBDE burdens showed an impairment of some components of the immune system (Beineke et al., 2005). No further investigations were conducted since the beginning of this century. This is also the case for the morphological changes in the thyroid indicating a dysfunction of this organe (Das et al., 2006). The health status of harbour porpoises continouse to be worse when compared to animals from less polluted waters (Siebert et al., 2006b; 2009; Lehnert et al., submitted). This is of special concern as harbour porpoises live in a polluted environment of the North and Baltic Seas with increasing anthropogenic activities.

Harbour seals

The decreasing antibody titer for the PDV, makes the population vulnerable to a new outbreak. Therefore, it is essential to know the current immune status of the population. The question whether the PDV affects rather animals with highest level of organochlorine in current studies rises (Härkönen et al., 2006). Current information the effects of chemical pollutants on the endocrinium and reproductive system are missing.

Grey seals

The Baltic seal complex with variegated lesions in different organ systems has been described previously (Bergman and Olsson, 1985). An influence of organochlorines on the Baltic seal complex is suggested (Bergman and Olsson, 1985). Moreover, the trabecular bone mineral density seams to decrease during the years of high organochlorie intake (Lind et al., 2003). Specific investigations on the effects of chemical pollutants on the endocrinium, immune, reproductive, skeletal and alimentary system are lacking.

5.4.4 Actual knowledge in context of the Marine Strategy Framework Directive (MSFD)

The aim of the MSFD is to maintain the biological variety and to protect the marine environment. Furthermore, a good environmental status should be aspired to. Therefore, attributes for a good environmental status should be established. Due to the fact that marine mammals obtain the end of the marine food web, they are important indicator for pollutant concentrations and their effects in German waters.

As an initial measure, the different levels of contaminants need to be taken into account. Obviously, marine mammals from less polluted environments are healthier than those from waters with a higher pollutant contamination (Jepson et al., 2000; Siebert et al., 2001; 1999; Jauniaux et al., 2002). Subsequently, the effects of different concentrations of hazardous compounds on marine mammals have to be examined. According to the current knowledge, contaminates seems to have both a negative influence on the reproduction system and the immune system. Influences on the immune system are certain, marine mammals become more vulnerable to other infectious diseases and parasites as a result of exposure to contaminants. Additionally, the general health status can be affected as well. To reduce the intake of contaminants in the environment, different methods need to be developed.

Marine mammals as top predators are a good indicator for accumulation of hazardous substances. Different studies show that marine mammals in European waters suffer from high pollution levels. However, information on relevant chemical and pharmaceutical pollutants in marine mammals from German waters is lacking for the last 10-15 years. These levels are needed to set in context their observed effects and create new critical limits. Furthermore, investigations on the effects of pollutant levels on different organ systems and the general health status are scarce. This data is essential and a main goal of the MSFD. Based on these limits, changes in the marine environment can be detected and actions for improving environmental status can be planned.

5.4.5 Research needs on

5.4.5.1 Chemical pollutants

The literature on pollutants, their concentrations and effects on marine mammals, shows that a lot of research has already been undertaken globally. Reffering to Germany specifically, however, just a few studies have been performed over the last 15 years. These results are mainly originating from adjacent waters and research facilities outside of Germany. The comparability of the collected data is complicated due to the fact that measurements were taken during different years, with different methods and different units. For the future protocols need to be established to secure a collection of homogenous data in relation to origin of animals, sex and age ratio, pollutants, tissue, methods, units. The protocol should be the basis for a future monitoring conducted on all marine mammal species and all areas of the German waters.

5.4.5.2 Effects of pollutants

The general health status needs systematically examined for all marine mammals from all areas of the German waters. This should allow to judge the development of the health status and to compare the situation with animals originating from areas with different pollutant levels. Investigations on target organe systems such as the immune, reproductive, digestive, auditory and endocrine systems need to be conducted. This observation is fundamental to evaluate of the overall burden caused by hazardous substances.

5.4.6 Recommendation of research concept for toxic effects in marine mammals

The general health status should be examined according to protocols described in Siebert et al., (2001; 2006; 2007). Sampling protocols should be established for the specific chemical and pharmaceutical pollutants and the matrices needed for effect analyses.

5.4.7 Recommendation of evaluation methods for toxic effects in marine mammals

One approach to assess pollutant-induced effects on health- and immune system of mammals are gene expression studies using molecular biomarkers. The genomic signal has been shown to be a new tool to monitor pollutant effects on the cellular level (Kim et al., 2002, 2005; Asakawa et al., 2008) and recently, molecular methods have been used successfully to assess health status of live marine mammals (Müller et al., 2013, Weirup et al., 2013). Gene expression studies have the potential to give early warning information on contaminant induced changes in top predators that are important indicator organisms in the marine ecosystem.

In this approach mRNA expression of several receptors and biomarkers associated to toxicology related genes is monitored using RT-qPCR from blood and tissue samples of marine mammals.

While studies investigating tissue samples can look at gene expression in liver or blubber, organs that are crucial in metabolizing and storing contaminants within the organism, the advantage of mRNA from blood samples is that live animals can be sampled minimally invasive during e.g. medicals and additional biomarkers for the immune system (e.g. cytokines) can be measured.

Contaminants associated with immuntoxicity in marine mammals can be mediated via the aryl hydrocarbon receptor (AHR), with "dioxin-like" compounds having the greatest immunotoxic potential (Luster, 1987; Safe, 1990). AHR upregulation is activated by environmental pollutants such as PCDDs, PCBs and PCDFs and aryl hydrocarbon receptor nuclear translocator (ARNT), its dimerization partner, is responsible for DNA binding and dimerisation (Beischlag et al., 2008; Fujii-Kuriyama and Kawajiri, 2010; Chopra and Schrenk, 2011). Peroxisome proliferator-activated receptor (PPARα) is a nuclear receptor involved in the induction of detoxifying enzymes, regulation of mRNA transcription and functions as a transcription factor (Van Raalte et al., 2004). PPAR alpha, AHR and ARNT as parts of the xenobiotic metabolism have been used to assess pollutant-induced changes and the ecological risk of dioxin-susceptibility in tissues of mammals (Kim et al., 2002, 2005; Asakawa et al., 2008). A new study established these markers for the first time in blood samples of harbor seals (Weirup et al., 2013).

The new and sensible method to assess early effects on the health and immune status of harbour seals using a spectrum of biomarkers is an important step in understanding wildlife populations in an anthropogenically exploited environment. Since baseline data from RNA expression levels in harbour seals are now becoming available for pollutant-induced biomarkers, the physiological mechanisms in the seals metabolism connected to contaminants and consequential health issues in the context of conservation purposes can now be better understood and monitored.

Although contaminant exposure has been discussed to negatively affect immune competence it has remained difficult to prove interrelationships between the two PDV epizootics in 1988 and 2002, previous animal exposure to pollutants and subsequent impairment of their immune responses (Hall et al. 1992a, b; Reijnders and Aguilar 2002; Härkönen et al., 2006). After a short lactation period of about 24 days pups are weaned abruptly and left to hunt on their own (Muelbert et al., 1993). Little is known about the development of the immune system in harbour seal pups after weaning and about the influence of contaminants, stress and infectious diseases upon their health status.

Cause-effect studies to reveal the correlation between pollutant levels and health effects on the molecular level are needed for early diagnoses of health risks in marine mammal populations. Biochemical markers which can be integrated in health monitoring programs are essential (Reijnders et al., 2007) and because the correlation between biochemical parameters and contaminant levels may vary from one species to another (e.g. in Baltic ringed and grey seals as shown by Nyman et al., 2003), species-specific biomarkers are valuable to monitor the health status and possible toxic effects of contaminants.

Samples for marine mammal studies from the wild are difficult to obtain and experimental work with captive animals is rarely performed (de Swart et al., 1994, 1996). The in vivo induction of cytochrome P450 1A (CYP 1A) in skin and liver biopsies of captive harbour seals after feeding-experiments with organochlorines was investigated to develop a biomarker for contaminant exposure (Miller et al., 2005; Assunacao et al., 2007) but the need for samples from free ranging seals was stressed. Nowadays, experimental studies on live animals are ethically doubtful. Cell culture models can replace in vivo experiments allowing for maximum control and minimum harm to study animals (Fresney, 2005). For identification of potential protein biomarkers in exposition studies primary hepatocytes from wild-ranging harbour seals were isolated and incubated with environmentally relevant contaminant concentrations (Korff et al. in prep.). In previous experiments testing pollutant exposure on marine mammal tissues mostly blood and epithelial cells were used, that can be obtained non-invasively and relatively easy (De Guise, 2005). In a recent study by Korff et al. (in prep.), liver tissue was obtained from freshly dead harbour seals in the German Wadden Sea, which died or had to be euthanized due to severe illness. Viable hepatocytes from fresh liver tissue were obtained using a two-step biopsy perfusion method (Reese and Byard, 1981). During cultivation, the hepatocytes were exposed to environmentally relevant concentrations of PFOS and an Aroclor mixture (PCBs), respectively. Cell viablilty and the cell culture test system as a model for the investigation of sublethal pollutant-induced effects was tested (XTT assay, LDH release, urea synthesis). Subsequent analysis of sublethal effects on the proteome/protein expression and genome (see Behr et al., 2008) was performed.

To investigate the connection between pollutant levels and health effects of marine top predators on the molecular level cell culture models may be an important tool. The identification of potential biomarkers for the diagnosis of disease and pollutant effects can complement laboratory work and help implement the results and to investigate effects of pollution in marine mammal populations in the wild.

In general, a broad spectrum of molecular markers can be used to allow for better assessment of the complex interactions between anthropogenic stressors, physiological response and immune status. Therefore, markers of the immune- and stress response have been shown to complement pollutant-induced biomarker data and help put results into perspective.

Cytokines are important cell mediators and responsible for the initiation, amplification and maintenance of an immune response (Mosmann and Sad, 1996). IL-2 and IL-10 are known to be correlated with inflammatory disease in marine mammals (Fonfara et al., 2008; Beineke et al., 2007; Weirup et al., 2013) and heat shock proteins (e.g. HSP70) are essential in the cellular response to viral, bacterial and parasitic pathogens (Lindquist, 1986; Chen and Cao, 2010) and frequently used to measure stress as well as immune reactions (Weirup et al., 2013; Müller et al., 2013). Acute phase proteins like haptoglobin (HP) influence several physiological functions of the immune system and have been used as markers for infection and stress (Petersen et al., 2004; Fonfara et al., 2007).

An array of markers of the xenobiotic metabolism (ARNT, AHR & PPARα), cytokines (IL-2 & IL-10), heat shock protein (HSP70) and acute phase protein (HP) may be powerful indicators to evaluate pollutant exposure and useful to serve as early warning indicators, monitoring and case-by-case tool for threats to marine mammal populations examined in the wild.

New markers should be established as additional indicators for environmental stress (e.g. c-reactive protein (CRP); CYP1) and other biotic (parasites) and abiotic stressors (temperature, UV radiation) have to be taken into account because pathogenic effects of several stressors can modify and enhance each other (Marcogliese et al., 2001).

Investigations on the endocrinium such as morphological and functional studies on the thyroid, adrenal gland and hypophysis need to be conducted. Effects on the reproductive system need to be studied. This should include e.g. the reproduction rate, sperm activity, pathological changes of reproductive tract. Relationship between chemical and pharmaceutical substances and the impairment of hearing needs to be assessed.

5.4.8 Conclusion

It can be concluded that recent data on chemical and pharmaceutical burdens of marine mammals from German waters are lacking as well studies on effects of pollutants on marine mammals. Research and monitoring programs for the assessement of pollutant levels and their effects need to be established as described above. It is not possible to fulfil the requirements by the MSFD before the relevant data are collected. As marine mammals are sentinel species in their environment and showed and impaired health status it is essential to understand and decrease effects of pollutants on marine mammals to reach a good environmental status.

6 Work package 5: Marine Litter (Descriptor 10)

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[b)] Chapter 6.6

[c)] Chapter 6.7

6.1 Literature Study

Amongst others within work package 5, the Institute of Environmental Systems Research (USF) of the University of Osnabrück performed a literature study focused on literature on marine litter abundance in the North and Baltic Sea and on methods to measure its abundance. Therefore, articles in scientific journals, PhD and Master Theses, and reports of various institutions and organizations have been reviewed.

More than 500 publications – journal articles, books, theses and reports – regarding marine litter have been collected in a literature database. Table 6-1 gives a detailed overview on the numbers of publications recorded until November 4, 2013.

Table 6-1: Number of publications identified by the literature study categorized by publication type until November 4, 2013. Where sensible, a sub-categorization is defined. In the category "reports", not only reports from research projects but also reports from NGOs, international and national governmental agencies are listed. Those reports contain survey data, survey protocols or techniques, consequences of marine litter, and political instruments against input of litter.

Publication Type	Number of publications	Subcategory Publication	Number in subcategory
Journal Article	432	Peer-reviewed	424
		Non-peer-reviewed	8
Books	22	Books	13
		Book chapters	9
Theses	5	PhD Theses	3
		Master Theses	2
Conference Proceedings	9		
Reports	109		
Sum	577		

The database has been made available to all partners involved in work package 5 and will be updated in regular intervals.

The focus of the literature study lies on macro-litter in the North Sea and the Baltic Sea. Hence, the set of scientific journal articles, which focus on regions outside of Europe and especially those on micro-plastics, is incomplete. However, the number of articles regarding the North Sea region is 20. These comprise twelve articles in peer-reviewed journals (Dixon and Dixon, 1981, 1983; Dubaish and Liebezeit, 2013; Franeker et al., 1985, 2011; Galgani et al., 2000; Liebezeit and Dubaish, 2012; Sommerville et al., 2003; Storrier et al., 2007; Thiel et al., 2011; Vauk and Schrey, 1987; Vauk and Vauk-Hentzelt, 1991) and eight in non-peer-reviewed journals (Clemens, 1992; Clemens et al., 2002, 2004, 2011; Hartwig, 2000, 2001; Niedernostheide and Hartwig, 1998a, 1998b). Most research on marine litter has been carried out in the USA with a major emphasis on the Pacific Ocean. Recent articles focus on the detection and removal of marine litter, especially derelict fishing gear (Mace, 2012; McElwee et al., 2012; Morishige and McElwee, 2012; Pichel et al., 2012; Veenstra and Churnside, 2012). In 2012, a literature study on methods used for the identification and quantification of micro-plastics was published (Hidalgo-Ruz et al., 2012) and has been merged with our database.

Many reports on marine litter written by NGOs (such as Greenpeace, WWF and Ocean Conservancy), international and national governmental agencies (such as UNEP, UBA, KIMO, EPA and OSPAR[63]) and researchers, such as research project reports, have been listed in the database. 53 of the collected reports are useful, meaning that they provide useful data, describe methods for measuring the amount of marine litter in certain compartments, or describe politic instruments aiming at the reduction of marine litter.

Thirteen books, nine book chapters, and three PhD theses have been included in the database.

Additional to scientific literature, several articles on marine litter have been published in Newspapers and Magazines. Internationally most recognized is a L.A. Times article (Weiss et al. 2006). In Germany, articles published in the SPIEGEL raised awareness of marine pollution with litter. Aside, one chapter in the UNEP Yearbook 2011 deals with marine litter and its consequences.

6.2 Trends of abundances in litter deposited on beaches and/or discarded in coastal waters, as well as micro-plastics

6.2.1 Methods

6.2.1.1 Identification of input variables

Derivation of a beach classification demands univariate and multivariate statistical analyses of the monitoring data mentioned above. Primarily, the OSPAR database (OSPAR, 2009) was taken for canonical correlation analyses to figure out significant increasing or decreasing temporal trends and for further analyses, such as clustering of beaches applying hierarchical cluster analyses. For this purpose, input variables (items and general categories) useful for analyses and subsequent classifications were identified from the set of OSPAR beach litter items. These input variables were selected according to the following criteria:

- sufficient abundances

[63] UNEP = United Nations Environmental Programme; UBA = Umweltbundesamt (German Environment Agency); KIMO = Kommunenes Internasjonale Miljøorganisasjon (Local Authorities International Environmental Organization); EPA = Environmental Protection Agency (USA); OSPAR = Oslo and Paris Convention

- obvious temporal trends according to scatter plots of absolute abundances of items against time
- regionally differentiating
- risk potential for animals by entanglement

Table 6-2 gives an overview of selected input variables.

Table 6-2: Selected input variables for statistical analyses and their weighting factors in the evaluation system proposed.

OSPAR ID	Input variables (items and source variables)	Weighting factor
33	Tangled nets/cord	1.5
35	Fishing line (angling)	1.5
200	Rope/cord/nets < 50 cm	1.5
201	Rope/cord/nets > 50 cm	1.5
15	Caps/lids	1.0
98	Cotton bud sticks	1.0
39	Strapping bands	1.5
4	Plastic drink bottles (Drinks)	1.0
19	Crisp/sweet packets and lolly sticks	1.0
49	Balloons	1.5
204	Cartons/Tetrapacks	1.0
43	Shotgun cartridges	1.0
-	Sum of all objects originating from fishing	3.0
-	Sum of all objects originating from shipping	3.0
-	Sum of all objects originating from tourism	3.0
-	Sum of all plastic objects	3.0
-	Sum of all packaging material objects	3.0

6.2.1.2 Descriptive statistics

All statistical analyses were carried out with the statistical software package SPSS 20.0 (IBM Corp., USA, www.ibm.com/software/analytics/spss/).

OSPAR data from beach litter monitoring were aggregated for OSPAR sub-regions and statistically described by boxplots of general categories for the period 2008 to 2012. Based on recent beach litter monitoring data from 2012, general compositions of beach litter were described for three OSPAR sub-regions.

6.2.1.3 Canonical correlation analyses

For time series correlation analyses, data from OSPAR beaches of 100 m length were taken, because litter categorization of 100-m-beaches is far more detailed than categorization of 1000-m-beaches. First, datasets of the OSPAR database were separated into three subsets:

1) entire annual cycle
2) winter period (surveys in December/January and March/April)
3) summer period (surveys in June/July and September/October)

Subsequent correlation analyses were applied to each of these three subsets, each input variable, and each OSPAR beach of 100 m length.

For analyses of temporal trends of beach litter monitoring data, bivariate canonical correlation analyses were performed between absolute abundances of input variables and time. Prior to correlation analyses, input variables should have been tested for normal distribution applying Kolmogorov-Smirnov-tests for normality. However, we assumed that at least one time series of one input variable was non-normally distributed. Therefore, tests for normality were skipped and for correlation analyses, the canonical correlation coefficient Spearman-r was used, which is applicable to both, normally and non-normally distributed data. Spearman-r is defined as follows:

$$r_s = 1 - \frac{6 \cdot \sum_{i=1}^{n} d_i^{\,2}}{n \cdot (n-1) \cdot (n+1)} \tag{1}$$

where r_s is the correlation coefficient Spearman-r, d is the distance of rankings between two paired values, and n is the number of replicates.

Results comprised Spearman-r, levels of significance (p-levels), and numbers of replicates. For reasons of statistical reliability, correlation analyses were only calculated if the number of replicates was $n > 9$.

6.2.1.4 Linear regression analyses

A screening of single linear regression analyses with general categories (sum of items originating from fishing, shipping, and tourism, as well as total plastic and total packaging material) as dependent variables against time was performed, in order to check whether the magnitudes of temporal decreases and increases, indicated by significant rank correlations, were substantial.

F-statistics were calculated in order to test regression models for overall significance. For reasons of statistical reliability, regression analyses were performed for all beaches and all general categories with a minimum number of 10 replicates and with significant rank correlations. Thus, there were a total of 47 regression analyses. By linear regression models, temporal trends were quantified, calculating percentages of increases or decreases within three years and related to the start value (intercept of the regression line in 2001).

6.2.1.5 Non-parametrical analyses of variance

OSPAR Beach litter monitoring data were tested for significant seasonal differences, applying Kruskal-Wallis-H tests (non-parametrical one-way analyses of variance) for each beach and each input variable when the number of replicates within each season (group) amounted to at least four. Time series of 21 beaches were eligible for analyses of variance.

Subsequently for time series with significant seasonal differences, Games-Howell post-hoc tests were carried out, in order to test single groups for significant differences to other groups.

6.2.1.6 Hierarchical cluster analyses

For the envisaged classification of beaches according to absolute abundances of beach litter, mean abundances were calculated for each 100-m-beach (n = 78) and each input variable. This data aggregation was necessary to obtain a complete matrix of input data. The matrix consisted of input variables ordered in columns and beaches as objects to be clustered (rows).

Ward method was chosen as algorithm for clustering of beaches, because Ward delivers few clusters of similar size. Euclidean distances were taken as measures of proximity.

Prior to cluster analyses, raw data were standardized calculating z-values, to account for large differences in the order of magnitude between input variables. z-values are defined as follows:

$$z = \frac{x - \mu}{\sigma} \tag{2}$$

where μ is the mean of a variable, σ is the standard deviation of the same variable, and x is a replicate of the same variable.

In order to avoid data redundancy, six cluster analyses were calculated. For the first analyses, single items given in Table 6-2 were used as input variables. In each of the other five analyses, one general variable was used.

Dendrograms of these analyses showed three outlier beaches with extraordinary high abundances of beach litter. These three beaches were excluded in subsequent and final cluster analyses applying the same procedures as described above. Results of cluster analyses comprised dendrograms and assignments of beaches to clusters in forms of tables.

6.2.1.7 Classification of beaches

For each cluster, mean values of input variables were calculated. Subsequently, mean values of cluster means were calculated to derive the limits of three different environmental status classes. We assumed that in the OSPAR regions, the Good Environmental Status (GES) was not achieved within the survey period 2001-2012. Therefore, environmental status classes according to detected pollution with marine litter were defined as ‘mediocre’, ‘unsatisfactory’, and ‘bad’. Manual adaptations of the limits of the three status classes were necessary, in order to obtain intervals of similar size. Thereby, limits were always adapted to lower values.

The upper limit of the GES was defined as 10 % of the upper limit of the mediocre status. Limits were rounded to integers. Definitions of GES as 10%-percentiles ensured that intervals were of similar size. In addition, by defining very low limits, GES would be expected to reflect a low risk of harm for biota and the ecosystem as a whole.

6.2.1.8 Development of an evaluation system of beach pollution with marine litter

We propose a two-part evaluation system, where the first part relies on the classification described above and on mean abundances of input variables within at least three years of monitoring. The second part is based on significant increasing and decreasing trends implied by results of canonical correlation analyses within the same minimum period of monitoring. For statistical reliability, three years of monitoring was set as a necessary minimum value, in order to obtain time series of at least twelve replicates. We abstained from including results of additional regression analyses, because the vast majority of regression models hint at substantial changes for significant correlations between input variables and time within three years.

In both parts, classification variables are differently weighted for each single item variable or sum variable (Table 6-2).

Single item variables with a risk potential for animals by entanglement are weighted by the factor 1.5. In OSPAR beach monitoring, micro- and mesoplastics, which are prone to ingestion by vertebrates, are not defined as separate categories. Therefore, the proposed evaluation system lacks direct consideration of ingestion of plastics by sea birds or marine organisms. However, breakdown of monitored macroscopic plastic items leads to fragmentation to smaller plastic particles, which in near future might be ingestable for vertebrates, such as the Northern Fulmar. Nonetheless, single items with no direct risk potential via entanglement or ingestion are weighted by the factor 1. Sum variables, such as shipping or plastics, are weighted by the factor 3, because these variables include a variety of single item variables.

Each beach will be evaluated according to the classification system given in Table 6.6 and mean abundances of each input variable within at least three years of future monitoring. Classes of the environmental status (1, 2, 3, and 4 corresponding to good, mediocre, unsatisfactory, and bad, respectively) will be assigned and subsequently weighted by the factors given above. Finally, for each beach, a weighted average value of the environmental status will be calculated. Rounded weighted averages will be assigned to beaches as overall environmental status.

The second part of the evaluation system relies on significant rank correlations (Spearman-r) within at least three years. For a significant negative rank correlation, a beach obtains the assignment -1 multiplied by the weight of the classification variable. For a significant positive rank correlation, a beach obtains the assignment +1 multiplied by the weight of the classification variable. Subsequently, weighted assignments will be summed up for each beach.

A beach with a sum < -4 will get the attribute 'improving' or '+'. A beach with a sum value between – 4 and +4 will get the attribute 'stable'. A beach with a sum > 4 will get the attribute 'worsening' or '-'.

The MSFD demands evaluation systems and recommendations for entire water bodies. Therefore, aggregation of beach evaluations to evaluations of sub-regions, such as the North-East Atlantic, is necessary. In accordance with the OSPAR Convention, we propose averaging of evaluation results of single beaches for three of the five OSPAR (and MSFD) sub-regions, namely a) greater North Sea, b) Celtic Seas, and c) Bay of Biscay and Iberian Coast. Comprehensive data sets of beach litter monitoring are only available for these three sub-regions, while there are no or scarce monitoring data for the Arctic Region and the Wider Atlantic.

The proposed evaluation system was applied to twelve OSPAR beaches with sufficiently long and complete time series between 2007 and 2011.

6.2.1.9 Multidimensional scaling

Two-dimensional scaling was applied, in order to group a) the OSPAR input variables selected for beach evaluation according to their source or transport behavior, and b) OSPAR beaches to regions of similar pollution with marine litter. In the first analyses, the 17 OSPAR input variables selected for beach evaluation were used. In the latter analyses from these input variables, only twelve single categories were used, in order to avoid redundancy of input data.

Prior to analyses, time series of input variables from 2001 to 2012 were aggregated to mean values, in order to obtain a complete matrix of input data. This matrix was standardized calculating z-values, in order to compensate for different variability and orders of magnitudes of input variables. Euclidean distances were used as measures of proximity.

6.2.1.10 Factor analyses

Principal axis analyses were carried out, in order to group single categories of marine litter according to their source or their transport behavior differently affected by wind drift. For this purpose, time series of beach litter monitoring data from NGOs were used as input data, because the number of single categories of these datasets amounted to 28 and therefore contrary to the OSPAR beach litter monitoring data (112 single categories) was ideal for factor analyses. Three datasets were chosen for principal axis analyses, namely time series each from 1992 to 2002 of the beaches Sylt, Mellum Nord, and Mellum Süd. The beaches on the Island Mellum are situated in a sheltered estuary, while the beach on the Island Sylt is strongly exposed to westerly winds and currents.

Prior to statistical analyses, raw data were aggregated to seasonal mean values, so that there was substantial reduction of the number of replicates from n = 650 to n = 34. For factor analyses, both data types, raw data and seasonal means, were used separately.

All six data matrices were tested for necessary correlation applying the Kaiser-Meyer-Olkin Criterion. We calculated non-rotated principal axis analyses, as well as rotated analyses applying the algorithm Varimax. Rotation serves to unambiguously assign input variables to factors via factor loadings. The elbow criterion (scree plot) and the Criterion of Kaiser (eigenvalue > 1) were used to identify factors worth to be considered for interpretation. Input variables with high factor loadings > 0.7 or < -0.7 were assigned to these factors.

6.2.1.11 Artificial neural networks

In artificial neural networks, few representative litter categories were used as predictors of general categories, such as all objects originating from fishing, made of plastic, or used as packaging material. In addition, neural networks were applied to definitely assign single litter categories to litter sources, such as fishing, shipping, and tourism, which until present has relied on expert knowledge and consensus rather than on statistical criteria.

For these purposes, seven exemplary beaches of 100 m length each, located at the southern coast of the North Sea, were selected from the OSPAR beach litter monitoring database, namely Sylt, Minsener Oog, Juist, Bergen, Noordwijk, Veere, and Terschelling.

Prior to creating a prototype of neural networks, three single litter variables were selected as input variables, namely tangled nets, strapping bands, and crisp and sweet packaging, which are attributable to the source variables sum of fishing items, sum of shipping items, and sum of tourism items, respectively. These three single categories served as predictors, because they are among the most abundant single beach litter categories and show considerable variation over time. Season coded by numerical integers from 1 to 4 was chosen as additional predictor, because in the North Sea irrespective of the kind of beach litter, almost all significant seasonal patterns of litter abundances peak in spring, while they are usually low during the other seasons.

The sum variables items originating from fishing, shipping, and tourism, as well as total plastic, and total packaging material were chosen as dependent variables within the neural networks, because by these variables, the vast majority of beach litter is represented. In addition, the chosen single variables are also attributable to plastic and partly to packaging material.

Monitoring datasets of the seven beaches selected were subdivided into two portions each, where the first portions comprised monitoring data from 2001 to 2005, and the second portions included monitoring data from 2006 to 2012. The first subsets were used to train seven neural networks, corresponding to the seven selected beaches and identical to the prototype of neural networks. The second subsets served to verify the models.

In order to test simulations for bias due to temporal trends in observations, data of the seven beaches were randomly subdivided into two portions each, where the first portion comprised 40% and the second portion 60% of monitoring data. Again, the first subsets were used to train seven neural networks, corresponding to the seven selected beaches. The second subsets served to verify the models.

Subsequently, a prototype of a feed-forward neural network consisting of three layers was developed. The input layer comprised four units corresponding to the four predictors, and the output layer consisted of five units corresponding to the five chosen dependent general variables. Optimization of the network structure according to the criterion of the least total standard error required a single monitoring dataset of the beach Minsener Oog and resulted in inclusion of a hidden layer consisting of five units, as well as two additional bias units. Bias units had two functions: Together with logistic activation functions, they served to introduce variable thresholds of activation in the hidden layer. In addition, bias units served to keep minimum activations of units in the output layer. Figure 6-1 illustrates the structure of the prototype. The prototype was used for all seven beaches so that there was a total of seven neural networks identical in structure.

Prior to the training of networks, all input data and output data were normalized to values between 0 and 1, in order to account for different variations and degrees of magnitude of predictors and dependent variables. We defined logistic activation functions for all units of the hidden layer, while for the output layer, identity functions were chosen as activation functions. All seven neural networks were trained applying gradient descent learning with back propagation, where the training was supervised by the above-mentioned training datasets. Optimization of weights of network connections relied on the criterion of the least total standard error.

The statistical software package SPSS 20.0 (IBM Corp., USA, www.ibm.com/software /analytics/spss/) was used as platform to install and to run neural networks.

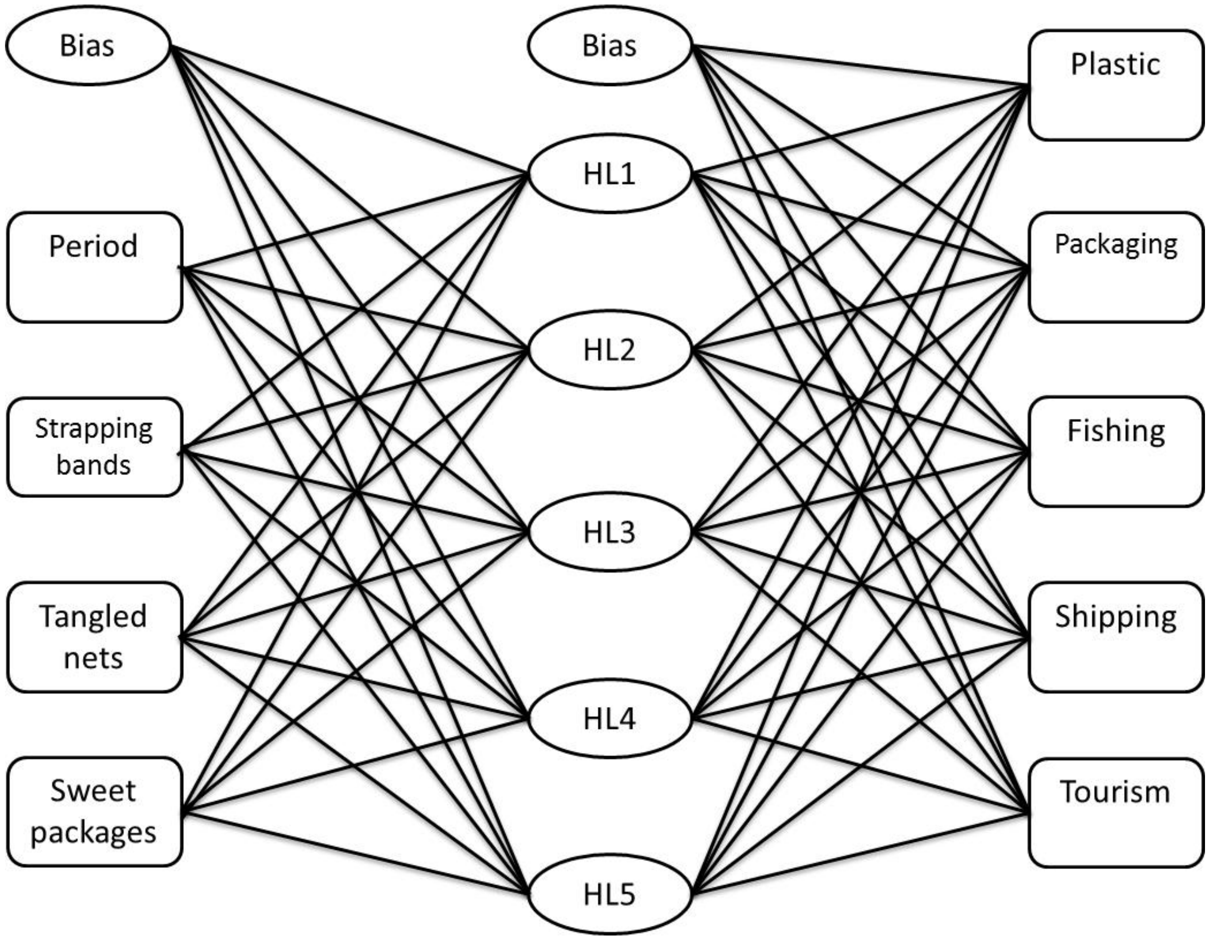


Figure 6-1: Structural scheme of the prototype of neural networks applied to model time series of beach litter on seven selected beaches. On the left-hand side, input units are revealed, while on the right-hand side, output units are shown. Elliptic units in the center of the scheme are units of a single hidden layer (HL). Additionally, two bias units are introduced. Lines represent information links between units.

6.2.1.12 Spatio-temporal trends of beach litter in the Weser Estuary and the North Sea between Bremen and the island Minsener Oog

Rivers are assumed to be major transport pathways of litter to the marine environment. However, quantification of input of litter from river mouths to adjacent marine regions has seldom been quantified. Therefore, amongst other monitoring of litter in the Weser Estuary, Bioconsult performed beach litter monitoring from May 2012 to April 2013 at four locations along the Weser Estuary between Bremen and Bremerhaven, applying the respective OSPAR protocol. Data of this one-year beach litter monitoring were provided to USF and subsequently related to contemporaneous beach litter data in the North Sea, where OSPAR beach litter monitoring has been carried out on the beach of the island Minsener Oog since 2002. Spatio-temporal trends were highlighted by multivariate statistical and geo-statistical analyses.

6.2.1.12.1 Data preparation

Data of beach litter monitoring in the Weser Estuary were paralleled to contemporaneous data of beach litter monitoring on the OSPAR beach of the island Minsener Oog.

Euclidean distance [km] was taken as distance downstream from the monitoring location in Bremerhaven to Minsener Oog, whereas distances between monitoring locations along the thalweg of River Weser were provided by Bioconsult.

6.2.1.12.2 Analyses of variance

Analyses of variance (ANOVA) were chosen as method to identify significant spatial and temporal trends of beach litter in the Weser Estuary and the North Sea. Prior to linear two-way ANOVAs, season coded by integers and distance downstream between Bremen and Minsener Oog were selected as two random factors. Six dependent variables were chosen from an evaluation system of marine beach litter, namely the source variables fishing, shipping, and tourism, as well as total plastic, total packaging material, and the single category crisp/sweet packages.

In the following, we use OSPAR terms for source variables, which might be misleading, because there are scarce fishing activities in the Weser Estuary, and tourism mainly comprises recreational activities.

All six dependent variables showed sufficient abundances and temporal variation, while all other categories revealed abundances close to zero and small variation. Levene-tests were carried out in order to check dependent variables for homogeneity of variance. For each dependent variable, a linear ANOVA was calculated, so that there was a total of six ANOVAs.

6.2.1.12.3 Geo-statistical analyses

Geo-statistical analyses served to illustrate spatio-temporal trends of beach litter in the Weser Estuary and the North Sea and thus to identify potential sources of litter for the North Sea. Orthogonal Kriging was applied with time between May 2012 and April 2013 and distance downstream between Bremen and Minsener Oog as x- and y-axes variables, while the six dependent variables mentioned above were chosen as interpolated z-variables. Spatio-temporal resolutions of six contour plots amounted to 1 km x 3 days each.

6.2.1.13 Micro-plastics in beach samples

6.2.1.13.1 Extraction method

The conventional standard method for separation of micro-plastics (diameter less than 1 mm) from natural material in sand samples had been optimized. Micro-plastics can be extracted from samples as large as 1 kg sand. The method consists of the following steps:

1) Pre-extraction with saturated sodium chloride solution (density ≈ 1.2 g mL^{-1}) via fluidization with an additional airflow that increases the buoyancy of particles.
2) Density separation by flotation in saturated sodium iodide solution (density ≈ 1.8 g mL^{-1}), where mineral material with higher density is collected at the bottom of the separation vessel.
3) Decantation of the supernatant and collection of the micro-plastics particles by filtration.
4) Visual analysis under a stereomicroscope counting the number of particles per sample.
5) Identification of selected particles by pyrolysis-GC/MS utilizing the fact that most plastics materials have typical pyrolysis products, which can be identified by their mass spectra after gas chromatographic separation (*fingerprint*).

6.2.1.13.2 Method blanks

During analysis of the first samples, a large number of fibers of different size had been collected on the filters. Closer inspection revealed that it could be textile fibers from clothing. Visual analysis of filters that were solely exposed to the laboratory air (blank samples) showed the same kind of fibers as on the sample filters. This led to the conclusion that the majority of fibers – if not the complete material – stemmed from filter contamination by other sources (lab air). This was corroborated by analyses of samples where this contamination pathway was excluded by keeping the samples in closed systems all the time.

Nevertheless, it is not yet clear, whether fiber material from textile may enter the aqueous environment: Washing machine effluent most likely includes such fiber material that is then transported to the sewage treatment plant via household wastewater. In how far treated effluent from sewage treatment plants is still contaminated with textile fibers has to be investigated to explain findings of such fibers in environmental samples from rivers, beaches or ocean water.

6.2.2 Results and Discussion

6.2.2.1 Descriptive statistics

In all three regions considered in 2012, plastic was the most abundant material, while packaging material contributed between 19% and 34% to total beach litter. In the following, we provide compositions of beach litter in 2012 in forms of Table 6-3 and Table 6-4 as well as in Figure 6-2 and Figure 6-3. Tables and figures show averages of four sampling campaigns on 19 beaches in the southern and eight in the northern North Sea, respectively, and on five beaches in the Celtic Sea.

Table 6-3: Material composition of beach litter in three OSPAR sub-regions in 2012. Southern North Sea comprises France, Belgium, the Netherlands, Germany, and part of UK. Northern North Sea comprises Denmark, Sweden, Norway, and part of UK. Celtic Sea comprises Ireland and part of UK.

Material	Southern North Sea			Northern North Sea			Celtic Sea		
	Mean [-]	Standard deviation [-]	Percentage [%]	Mean [-]	Standard deviation [-]	Percentage [%]	Mean [-]	Standard deviation [-]	Percentage [%]
Plastic	547	715	85	632	724	75	556	569	84
Rubber	6	7	2	30	47	3	9	7	2
Cloth	2	3	1	167	396	8	4	3	1
Paper	19	67	2	15	20	2	9	12	2
Wood	19	24	4	11	13	2	3	2	0
Metal	8	15	3	19	39	1	6	4	2
Glass	5	7	2	16	35	2	5	7	1
Pottery	1	2	0	1	3	0	1	2	0
Sanitary	10	29	1	103	157	7	85	119	8
Medicals	0	1	0	1	3	0	1	1	0

Table 6-4: Composition of beach litter according to purpose in three sub-regions of the OSPAR-regions in 2012. Southern North Sea comprises France, Belgium, the Netherlands, Germany, and part of UK. Northern North Sea comprises Denmark, Sweden, Norway, and part of UK. Celtic Sea comprises Ireland and part of UK.

Pur-pose	Southern North Sea			Northern North Sea			Celtic Sea		
	Mean [-]	Standard devia-tion [-]	Percent-age [%]	Mean [-]	Standard devia-tion [-]	Percent-age [%]	Mean [-]	Standard devia-tion [-]	Percent-age [%]
Packag-ing	118	135	19	258	414	26	228	251	34
User item	91	166	15	355	495	35	171	182	25
Other	413	594	66	392	470	39	280	288	41

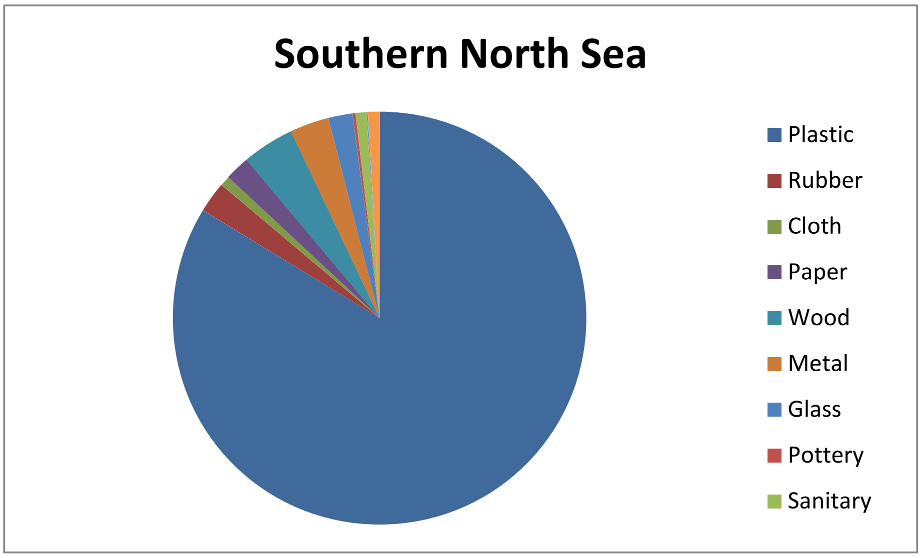


Figure 6-2: Material composition of beach litter in the southern North Sea in 2012.

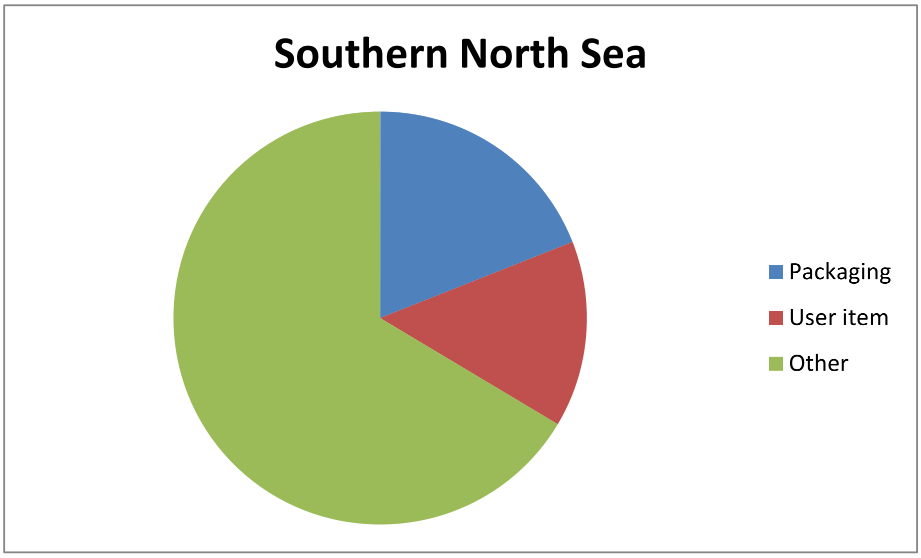


Figure 6-3: Composition of beach litter according to purpose in the southern North Sea in 2012.

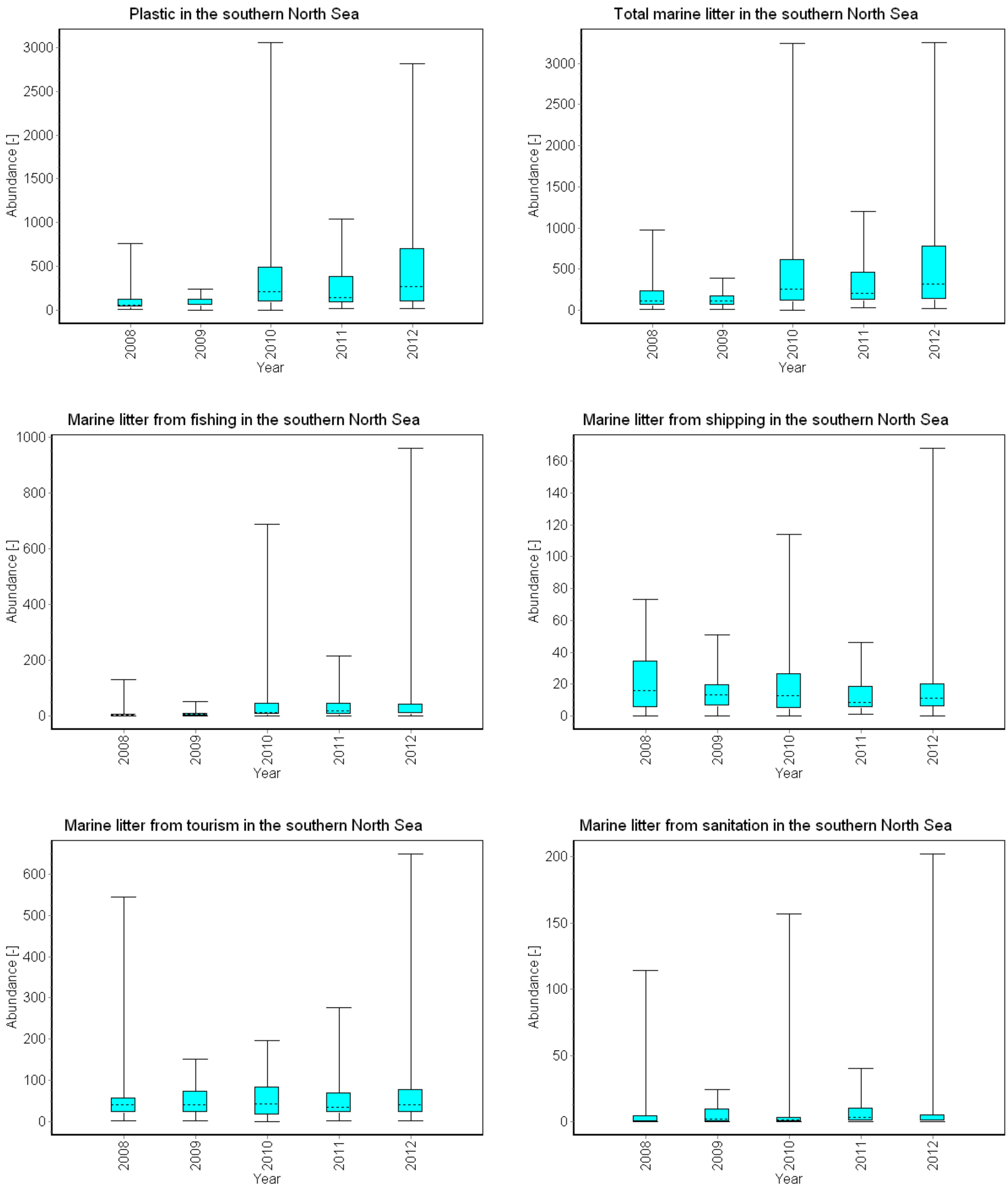


Figure 6-4: Exemplary boxplots of general beach litter categories in the southern North Sea from 2008 to 2012.

Overall in boxplots (Figure 6-4), general categories showed large scatter within each year of monitoring and sub-region. Therefore, temporal trends were scarcely detectable. For time series analyses, we recommend to remove outliers and to regard single beaches, in order to detect any significant temporal trends.

In Figure 6.4, exemplary boxplots of the southern North Sea are given as self-explanatory descriptive statistics of beach litter abundances in four OSPAR sub-regions between 2008 and 2012. Within the plots, quartile and median values are displayed by horizontal lines in the boxes, while error bars give the range of monitoring data.

6.2.2.2 Canonical correlation analyses

Table 6-5 provides an extract of results of correlation analyses showing results of German OSPAR beaches for the entire annual cycle. Only few correlations between input variables and time were significant. Mainly on German OSPAR beaches, negative correlations corresponding to decreasing trends can be observed when trends are significant. Other regions, such as Dutch beaches mainly revealed positive correlations corresponding to increasing trends when correlations are significant. Differences between regions probably originate from different flow regimes, different proximity to source regions, and different pollution with marine litter in the region itself (Galgani et al., 2000; Tudor et al., 2002). Detailed spatial analyses of sources and sinks of marine litter are necessary to explain differences in temporal trends in beach litter pollution and will be done, as soon as sufficient information on sources of marine litter, such as fishing, shipping, and tourism, is available.

Table 6-5: Extract from results of correlation analyses showing results of German beaches and the entire annual cycle. Spearman-r, levels of significance, and numbers of replicates are given for each correlation. Based on a significance level of p < 0.05, significant correlations are marked with an asterisk.

Input variable	Correlation results	Sylt	Scharhörn	Minsener Oog	Juist
Sum of items from fishing	Spearman-r	-.499*	-.307	-.521*	-.401*
	P-level	.003	.127	.002	.023
	N	34	26	34	32
Sum of items from shipping	Spearman-r	-.266	-.103	-.158	-.140
	P-level	.128	.615	.371	.445
	N	34	26	34	32
Sum of items from tourism	Spearman-r	-.235	-.020	-.383*	-.055
	P-level	.181	.922	.025	.766
	N	34	26	34	32
Plastic	Spearman-r	-.033	.339	-.195	.061
	P-level	.854	.090	.269	.741
	N	34	26	34	32
Packaging material	Spearman-r	-.156	-.188	-.068	-.209
	P-level	.420	.389	.720	.277
	N	29	23	30	29
Tangled nets/cord	Spearman-r	-.290	.093	-.381*	.058
	P-level	.096	.652	.026	.752
	N	34	26	34	32
Fishing line (angling)	Spearman-r	-.073	-.110	-.097	-.043

Input variable	Correlation results	Sylt	Scharhörn	Minsener Oog	Juist
	P-level	.681	.592	.587	.813
	N	34	26	34	32
Rope/cord/nets < 50 cm	Spearman-r	-.044	.196	.284	.061
	P-level	.829	.408	.160	.779
	N	26	20	26	24
Rope/cord/nets > 50 cm	Spearman-r	.081	.117	-.229	.200
	P-level	.694	.623	.261	.349
	N	26	20	26	24
Caps/lids	Spearman-r	-.095	.026	-.118	-.160
	P-level	.595	.901	.505	.382
	N	34	26	34	32
Cotton bud sticks	Spearman-r	.065		-.257	-.165
	P-level	.714		.142	.366
	N	34		34	32
Strapping bands	Spearman-r	-.228	.156	-.089	-.086
	P-level	.195	.445	.616	.638
	N	34	26	34	32
Plastic drink bottles (Drinks)	Spearman-r	.229	-.090	-.044	-.047
	P-level	.193	.662	.805	.797
	N	34	26	34	32
Crisp/sweet packets and lolly sticks	Spearman-r	-.237	.187	-.203	-.033
	P-level	.178	.361	.251	.860
	N	34	26	34	32
Balloons	Spearman-r	.046	.378	-.192	.178
	P-level	.796	.057	.276	.330
	N	34	26	34	32
Cartons/Tetra-packs	Spearman-r	-.286	-.012	0.000	.015
	P-level	.157	.961	1.000	.946
	N	26	20	26	24
Shotgun cartridges	Spearman-r	.208	-.389*	.133	.045
	P-level	.239	.049	.453	.805
	N	34	26	34	32

6.2.2.3 Linear Regression analyses

Thirty-five of 47 linear regression models between input variables and time were significant with $p < 0.05$. Except for one significant regression model within a period of three years, all significant models gave substantial linear increases or decreases of more than 20% of the start value. Our results exhibited good agreement between significances of correlation analyses and regression analyses, supporting the implicit evaluation method proposed.

Ribic et al. (2010; 2012) derived several nonlinear regression models to describe the development of pollution of coastal areas with marine litter. However, the purposes of their studies were to identify significant drivers of temporal development rather than to create a simple method to evaluate the environmental quality of beaches. Therefore, Ribic et al. (2010; 2012) had to test a variety of linear and nonlinear models for best fit applying Akaike's information criterion (Akaike, 1974), which is probably a too complex method for non-statistician users of an evaluation system.

6.2.2.4 Non-parametrical analyses of variance

Seasonal differences within time series of selected input variables were partly significant with $p < 0.05$. Spatial distribution of significant seasonal differences was heterogeneous. One Spanish (ES1) and one Swedish (SE3) beach revealed eleven and nine significant seasonal differences, respectively, while an additional twelve beaches exhibited between one and three significant seasonal differences each.

Results of Games-Howell post-hoc tests indicated two water bodies with different significant seasonal pattern. These two seasonal patterns were independent from input variables and could be assigned to a) the North-East Atlantic and b) the North Sea. The typical North-East Atlantic pattern peaked in autumn and winter, while the typical North Sea pattern had its maximum in spring (Figure 6-5).

Morishige et al. (2007) found no significant seasonal differences in litter pollution on beaches of Hawaii, but significant inter-annual variation due to El Nino events. In tropical regions, do Sul and Costa (2007) attributed seasonal variation of beach litter to tourism and rainfall. In the OSPAR region, similar spatio-temporal patterns independent from items and sources hint at hydrodynamics, wind direction and speed as driving factors of seasonal variation, because in the North-East Atlantic and the North Sea, climatic factors are not considerably different. Nevertheless, for the evaluation system proposed, distinct spatio-temporal patterns provide support for a differentiation between at least sub-regions.

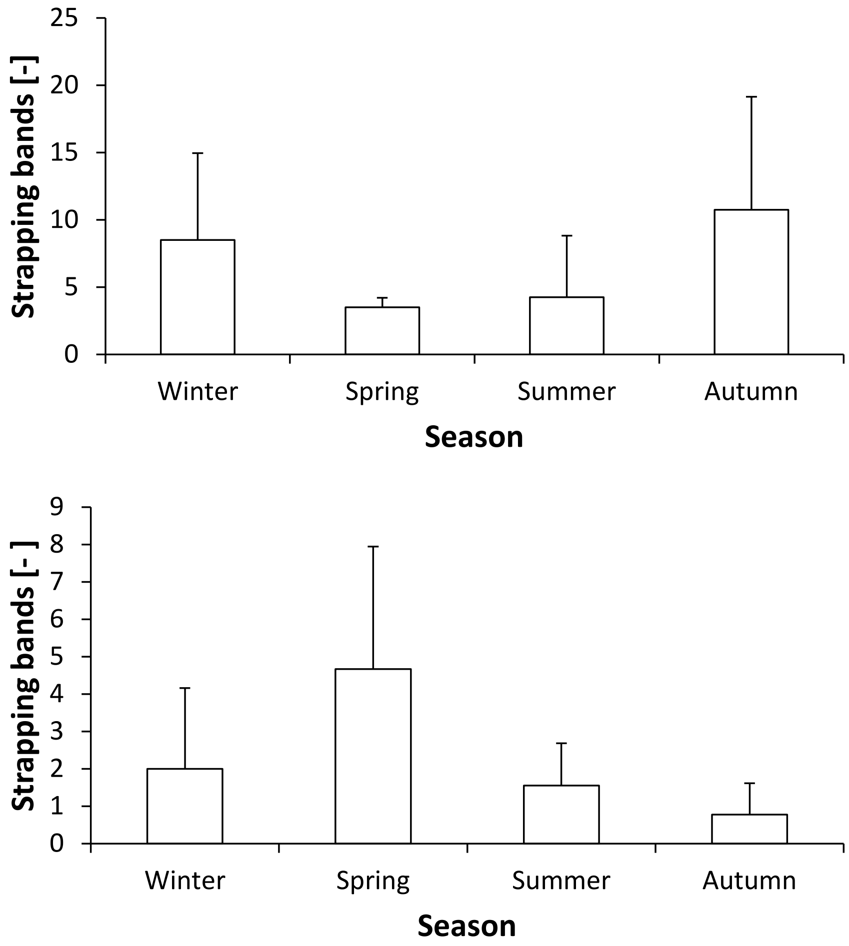


Figure 6-5: Exemplary seasonal patterns of beach litter in the North-East Atlantic (upper) and in the North Sea (lower).

6.2.2.5 Hierarchical cluster analyses

Figure 6-6 gives an exemplary dendrogram of cluster analyses and shows grouping in three major clusters, as it was intended by the procedures applied. These three resulting clusters were assigned to the three environmental status classes given above.

The temporal averaging of monitoring data, carried out in order to extract a data matrix applicable for cluster analyses, means substantial reduction of information. This step was necessary, because overlapping of time series of beaches was insufficient to use raw data as input for cluster analyses. Additional consideration of significant temporal trends in the evaluation system proposed might compensate for loss of information.

Williams et al. (2003) applied the Ward method to cluster beaches according to pollution with marine litter, as was done in a similar way in this study. Concordantly, outlier beaches were identified, and clustering of beaches was consistent with their spatial distribution. However, clusters of beaches do not necessarily mirror regions, but additionally and irrespective of location, they represent groups of beaches with similar environmental quality. Therefore, in this study, classification of beach quality primarily relied on three major clusters derived from cluster analyses.

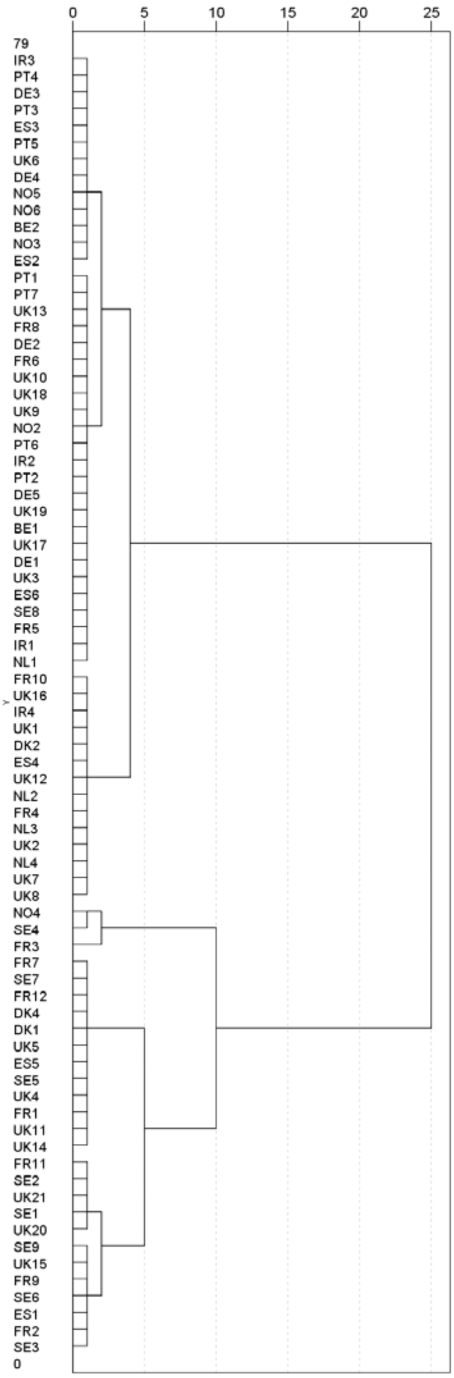


Figure 6-6: Exemplary dendrogram resulting from cluster analyses based on packaging material as input variable.

6.2.2.6 Classification of beaches

Table 6-6 gives the resulting classification of beaches based on absolute abundances of marine litter items and general categories. Expert knowledge and feedback from the OSPAR IC Group 'Marine Litter' acknowledged the scheme proposed.

Table 6-6: Classification of OSPAR 100m-beaches into four environmental states.

OSPAR ID/ general category	good	mediocre	unsatisfactory	bad	Remark
Plastic drink bottles (Drinks)	< 2	2 - 20	21 - 34	> 34	
Caps/lids	< 7	7 - 65	66 - 156	> 156	
Crisp/sweet packets and lolly sticks	< 2	2 - 16	17 - 61	> 61	
Tangled nets/cord	< 1	1 - 6	7 - 12	> 12	
Fishing line (angling)	< 1	1 - 10	11 - 32	> 32	manually adapted limits
Strapping bands	< 1	1 - 11	12 - 25	> 25	manually adapted limits
Shotgun cartridges	< 1	1 - 6	7 - 17	> 17	
Balloons	< 1	1 - 8	9 - 18	> 18	manually adapted limits
Cotton bud sticks	< 3	3 - 26	27 - 56	> 56	
Rope/cord/nets < 50 cm	< 6	6 - 59	60 - 121	> 121	manually adapted limits
Rope/cord/nets > 50 cm	< 2	2 - 16	17 - 34	> 34	manually adapted limits
Cartons/ Tetrapacks	< 1	1 - 4	5 - 10	> 10	manually adapted limits
Fishing	< 13	13 - 119	120 - 385	> 385	
Shipping	< 3	3 -22	23 - 88	> 88	
Tourism	< 6	6 - 55	56 - 164	> 164	
Packaging	< 14	14 - 143	144 - 336	> 336	
Plastic	< 50	50 - 502	503 - 1533	> 1533	

6.2.2.7 Evaluation system of beach pollution with marine litter

All beaches considered for evaluation of beach litter pollution were in the mediocre or unsatisfactory state, except for the German beach Minsener Oog, which nearly reached the good environmental status. Within the period 2007-2011, only two beaches from the Netherlands received additional attributes, such as improving or worsening. Table 6-7 gives an overview of the resulting evaluation for the period 2007 - 2011.

Our evaluation system based on a unique comprehensive database is easily applicable and requires little knowledge of applied statistics. Similar and reliable multi-criteria evaluation systems for coastal waters were created within the Water Framework Directive of the European Union (Muxika et al., 2007). Alakaly et al. (2007) derived an evaluation system for marine litter pollution for the coast of Israel, which solely relied on abundances of plastic debris. In their study, it remains unclear how the definitions of limits of a proposed pollution index were justified, and contrary to our approach, their evaluation system is applicable to a small region only.

Table 6-7: Evaluation of OSPAR beaches based on beach litter monitoring data between 2007 and 2011. Integers stand for environmental states (1 = good; 2 = mediocre; 3 = unsatisfactory; 4 = bad). '+' and '-'are attributes of the overall environmental state and stand for 'improving' and 'worsening', respectively.

Input variable/OSPAR ID	OSPAR beach ID											
	DE1	DE2	DE3	DE5	ES1	ES2	NL1	NL2	NL3	NL4	UK2	UK20
Plastic drink bottles (Drinks)	2	2	1	2	2	2	2	2	2	2	2	2
Caps/lids	2	2	1	1	4	2	2	2	2	2	2	3
Crisp/sweet packets and lolly sticks	2	2	1	2	3	1	2	2	3	2	3	4
Tangled nets/cord	2	4	2	1	4	2	3	3	4	4	1	1
Fishing line (angling)	2	2	1	1	2	2	2	2	2	2	2	3
Strapping bands	2	2	2	2	4	2	2	2	2	2	2	2
Shotgun cartridges	1	1	1	1	3	2	2	2	2	2	2	3
Balloons	2	3	1	2	1	1	3	3	4	3	2	2
Cotton bud sticks	1	1	1	1	4	4	2	2	3	2	1	4
Rope/cord/nets < 50 cm	2	3	2	2	4	2	2	2	4	3	2	2
Rope/cord/nets > 50 cm	2	3	2	2	2	2	2	2	4	2	2	2
Cartons/ Tetrapacks	2	2	1	2	2	2	2	1	3	2	1	1
Fishing	2	3	2	2	4	2	2	2	4	3	2	2
Shipping	2	3	2	2	3	2	2	2	3	3	2	3
Tourism	2	2	1	2	3	2	2	2	2	2	2	4
Packaging	2	2	1	2	4	2	2	2	3	2	2	4
Plastic	2	2	2	2	3	1	2	2	3	2	2	3
Rounded weighted mean and assignment of an environmental status	2	2	2	2	3	2	2	2-	3+	2	2	3

However, extrapolation of the proposed evaluation system to the Baltic Sea requires collection of additional information on beach pollution with marine litter, because in this marine area, marine litter partly originates from other sources than in the North-East Atlantic and the North Sea. Until present, little is known of abundances of marine litter in the Baltic Sea, where fishing strategies differ significantly from those in the North Sea. For spatial extrapolation of the proposed evaluation system and for common strategies against pollution with marine litter, extensive and systematic monitoring of pollution with marine litter is still necessary.

6.2.2.8 Multidimensional scaling

Kruskal's first measures of stress describing the goodness of configuration were moderate to high ranging from 0.17 to 0.26. Accordingly, in the Shephard diagrams, the criterion of monotony was scarcely fulfilled.

Configuration of the 17 selected input variables was sensible only in part: Caps and lids were placed in the vicinity of the source variable tourism, and nets were placed near the source variable fishing. However, grouping of single categories according to their origin was not consistent (Figure 6-7).

Configurations of the 78 OSPAR beaches did scarcely reproduce regional patterns of similar degree of pollution with marine litter (Figure 6-8).

Overall, multidimensional scaling did not succeed in sensible grouping of beaches and input variables. Therefore, multidimensional scaling gives little support for classification und subsequent evaluation of beaches according to pollution with marine litter.

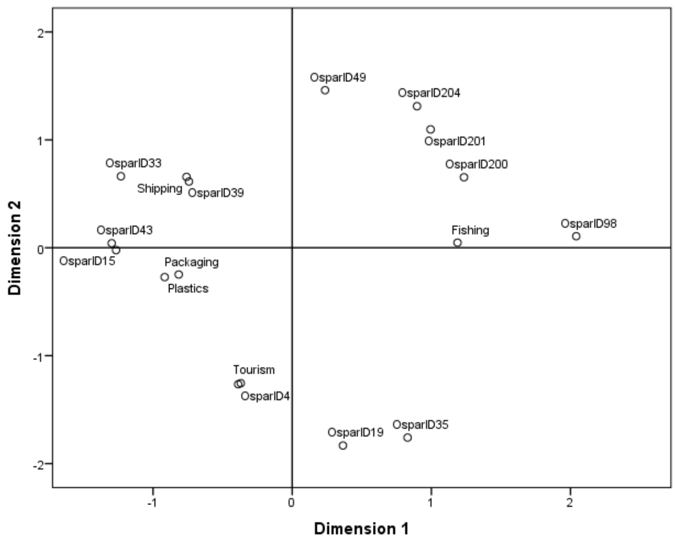


Figure 6-7: Two-dimensional configuration of the selected 17 input variables. Distances are given as Euclidean distances between z-values [-].

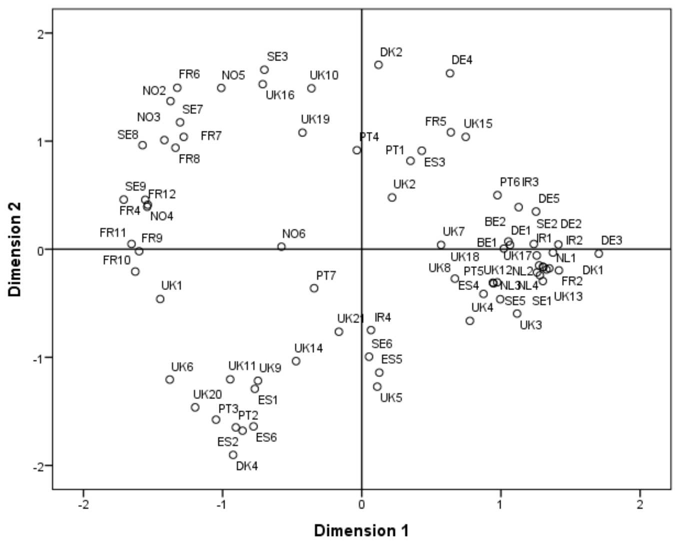


Figure 6-8: Two-dimensional configuration of 78 OSPAR beaches based on twelve single categories. Distances are given as Euclidean distances between z-values [-]. Abbreviations give OSPAR beach IDs.

6.2.2.9 Factor analyses

According to the Criterion of Kaiser-Meyer-Olkin (Cureton and D'Agostino, 1983), seasonal means were not acceptable for factor analyses, because for each beach their scores amounted to < 0.5. In contrary, raw data matrices were sufficiently correlated with scores > 0.7 for each of the three beaches. Therefore, seasonal means were not considered for factor analyses.

In the following, results of the beach on Sylt are given and discussed exemplarily. Communalities were low and very variable ranging between 0.4 % and 75 %. According to the Criterion of Kaiser (Kaiser and Dickman, 1959), nine factors would have been extractable. According to the elbow criterion, three factors would have been extractable (Figure 6-9). We applied the elbow criterion, because we intended to group input variables to few factors representing three sources of marine litter (fishing, shipping, tourism) or three wind drift factors.

The non-rotated solution gave only three variables loading highly on the first factor with the highest eigenvalue, while factor loadings were < 0.7 and > -0.7 for all other factors. The rotated solution gave only two variables loading highly on the first factor, while one further input variable was loading highly on the second factor.

Overall, results of factor analyses were poor and not eligible to group single categories according to their source or transport behavior, because explanation of variance (communalities) was low and input variables could not be grouped sensibly on the basis of factor loadings. This finding is in good agreement with previous studies (Tudor et al., 2002), which failed to assign single categories of marine litter to general categories applying factor analyses. Therefore, expert knowledge is necessary to define sources of single categories, as it was used for the OSPAR beach litter monitoring database.

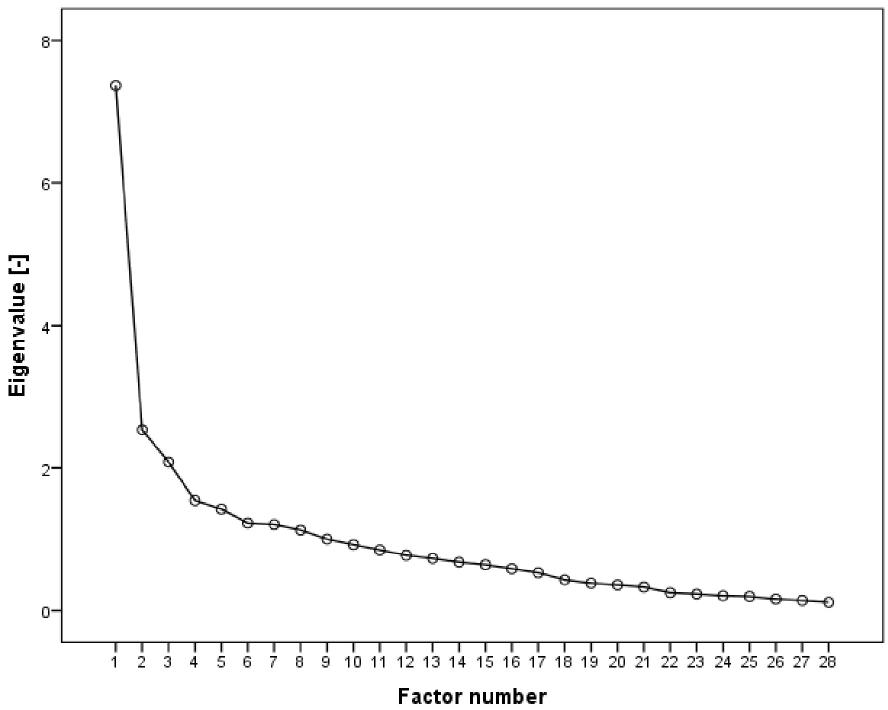


Figure 6-9: Exemplary scree plot of a principal axis analysis using raw data of the beach Sylt.

6.2.2.10 Artificial neural networks

Measured and modeled time series of output variables were in good agreement. Almost all rank correlations coefficients were > 0.6 and highly significant with levels of significance (p) < 0.01, both indicating significantly good correlations between measured and modeled time series. In addition, measured and modeled data were in the same order of magnitude, and minima and maxima overlapped well (Figure 6-10). There was no considerable difference between simulation results based on random selection of training and test data and simulation results with assignment of consistent time series to training and test data, respectively. This lack of bias shows that the neural networks were robust and rather insensitive towards any temporal trends in input data. Therefore, the neural networks employed are eligible to forecast general categories by a small selection of single categories as input variables. The good agreement between measured and modeled data evidences that the selected single variables are well representative of general sources and therefore may serve as indicators of these general categories.

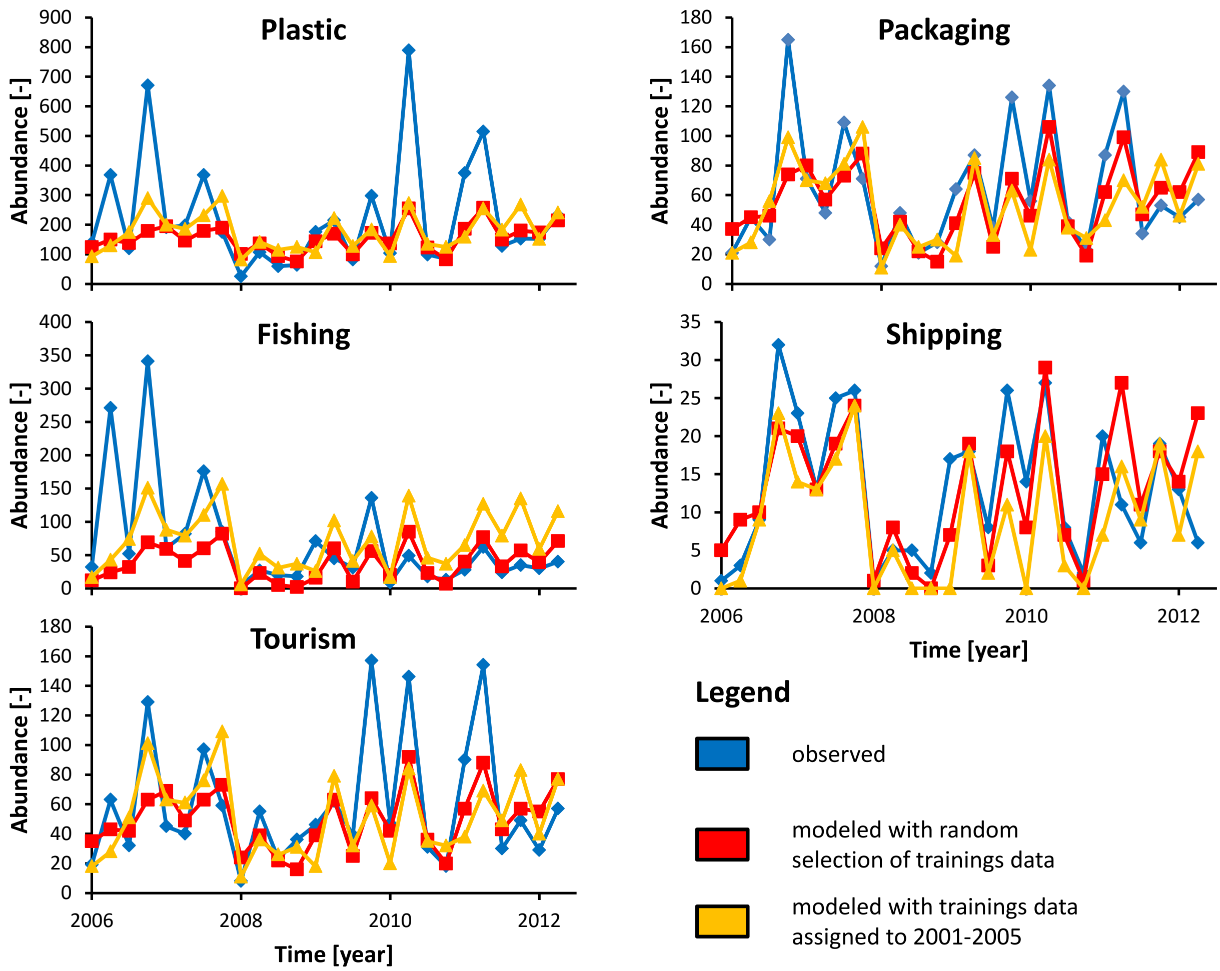


Figure 6-10: Exemplary plots of time series of observed (blue lines) and modeled (red and orange lines) output variables of the beach Bergen (The Netherlands).

However, in this study, neural networks underestimated measured maxima of general categories, when peak values were extremely high. This notable smoothing of maxima could not be attributed to normalizations of input data or to the number of units in the hidden layers. Smoothing of time series is inherent to neural networks with sigmoidal activation functions and can be helpful to generate datasets with less outlying data, which in turn might be useful for further time series analyses (Moon and Janowski, 1995). Moreover, definitions of linear terms as activation functions in the hidden layer would have led to faulty partly negative predictions of abundances of general categories.

There are very few previous studies that dealt with predictions of litter in the marine environment applying neural networks (Balas et al., 2004; Balas and Tur, 2006). These authors used neural networks in order to model single categories of beach litter by a variety of general categories as input units, which is the opposite way to our procedures. Balas and Tur (2006) could evidence that complex neural networks in combination with fuzzy logics are capable to predict the occurrence and composition of some part of beach litter. Our study demonstrates that also simple neural networks are reliable models not only to simulate selected beach litter categories, but also to predict long-term time series of general categories, which are representative of the entity of beach litter. Thus, few input data are necessary to generate comprehensive datasets of beach litter.

Future monitoring of beach litter could require less detailed categorization, focus on few single item variables and thus benefit from the application of the neural networks presented. Thereby, less human and economic resources could be required to quantify abundances of beached litter.

6.2.2.11 Spatio-temporal trends of beach litter in the Weser Estuary and the North Sea between Bremen and the island Minsener Oog

All six datasets showed homogeneity of variance and were therefore eligible for linear two-way ANOVAs.

Table 6.8 summarizes the results of all ANOVAs. Accordingly, except for shipping, downstream distance had significant effects of abundances of beach litter, while there were no significant effects of season. However, in two cases, significant interaction effects could be identified (total plastic and tourism).

Table 6-8: Results of linear two-way analyses of variance (ANOVA): levels of significance (p). All effects with $p < 0.05$ were considered as significant and are marked with asterisks ($p < 0.05$: *, $p < 0.01$: **, $p < 0.001$: ***).

Factor	Dependent variable					
	Plastic	Packaging	Fishing	Shipping	Tourism	Crisp/Sweet packages
Constant term	.283	.295	.263	.051	.247	.299
Distance downstream	.003**	.001**	.000***	.144	.003**	.006**
Season	.567	.513	.445	.642	.687	.513
Distance downstream * Season	.023*	.160	.198	.330	.047*	.053

Contour plots (Figures 6.11 to 6.16) were made for visualization purposes and illustrate temporal trends combined with spatial gradients. Contour plots evidence downstream decreasing trends for tourism, total plastic, total packaging material, and crisp/sweet packages, while abundances of fishing increased in downstream direction. In agreement with results of ANOVAs, shipping did not show any consistent spatial trend. Downstream decreasing gradients of litter give hints on potential sources and sinks, where hot spots are supposed to be potential source areas.

In addition, there were consistent temporal patterns of tourism, total packaging material, total plastic, and crisp/sweet packages, all of which peaked in early spring.

Spatial trends highlight tourism in the vicinity of Bremen as potential source of litter for the North Sea, also because plastic and packaging material mainly consist of items related to tourism. Temporal trends in the Weser Estuary are in good agreement with temporal trends of beach litter observed in the North Sea, giving further evidence for recreation-related littering in Bremen and its surroundings as major source of litter in downstream regions.

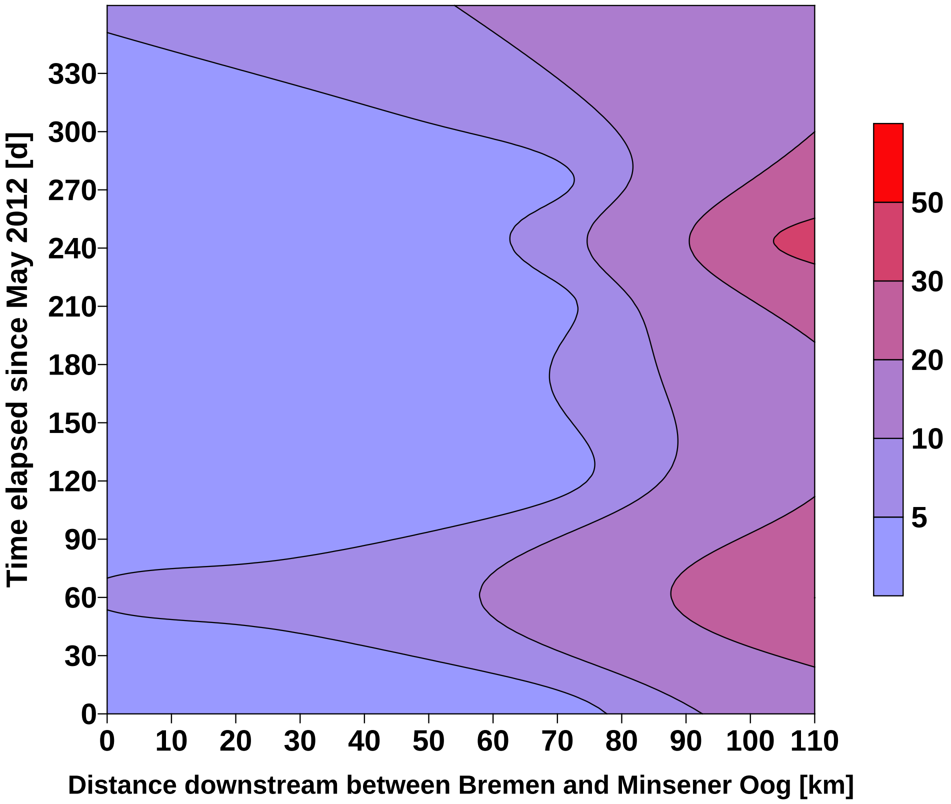


Figure 6-11: Contour plot of abundances of Fishing [-] based on orthogonal Kriging.

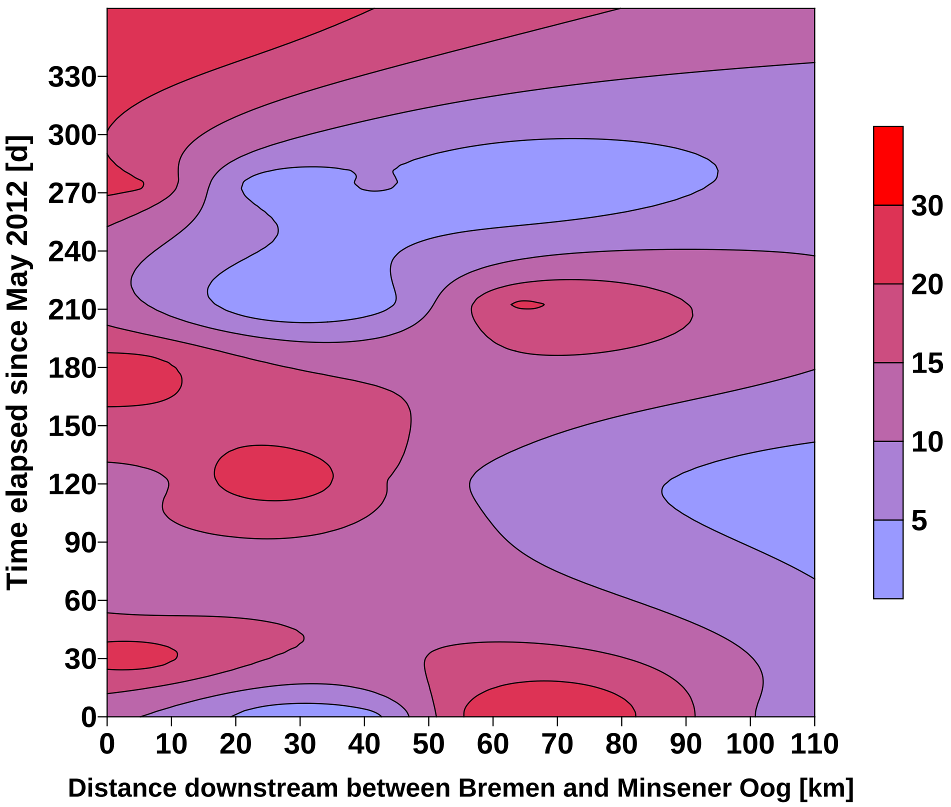


Figure 6-12: Contour plot of abundances of Shipping [-] based on orthogonal Kriging.

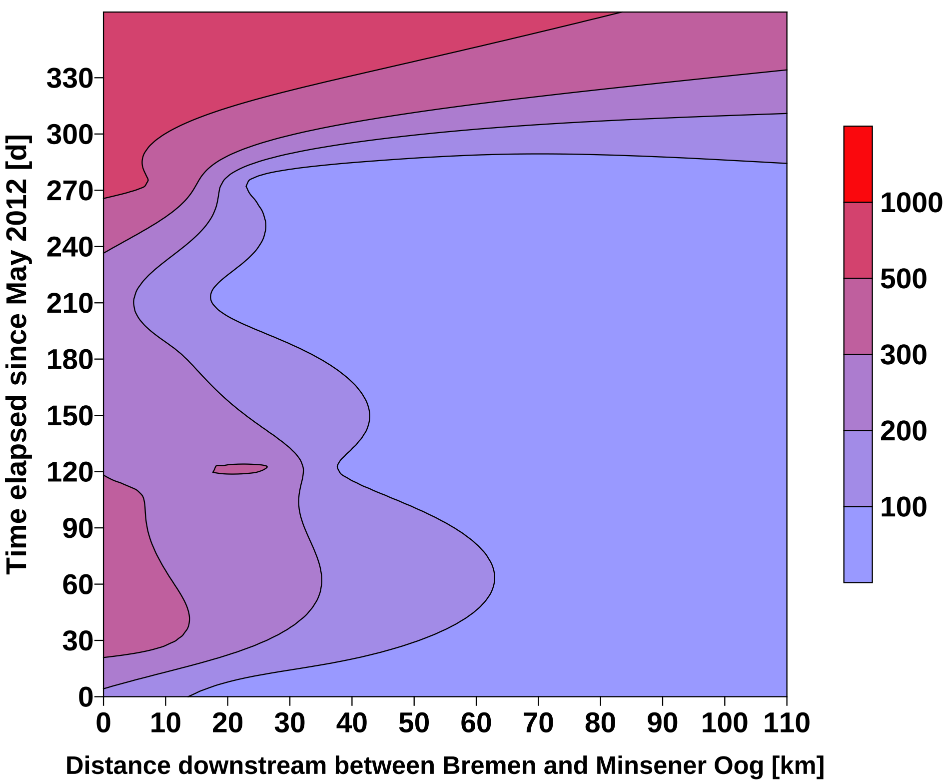


Figure 6-13: Contour plot of abundances of Tourism [-] based on orthogonal Kriging.

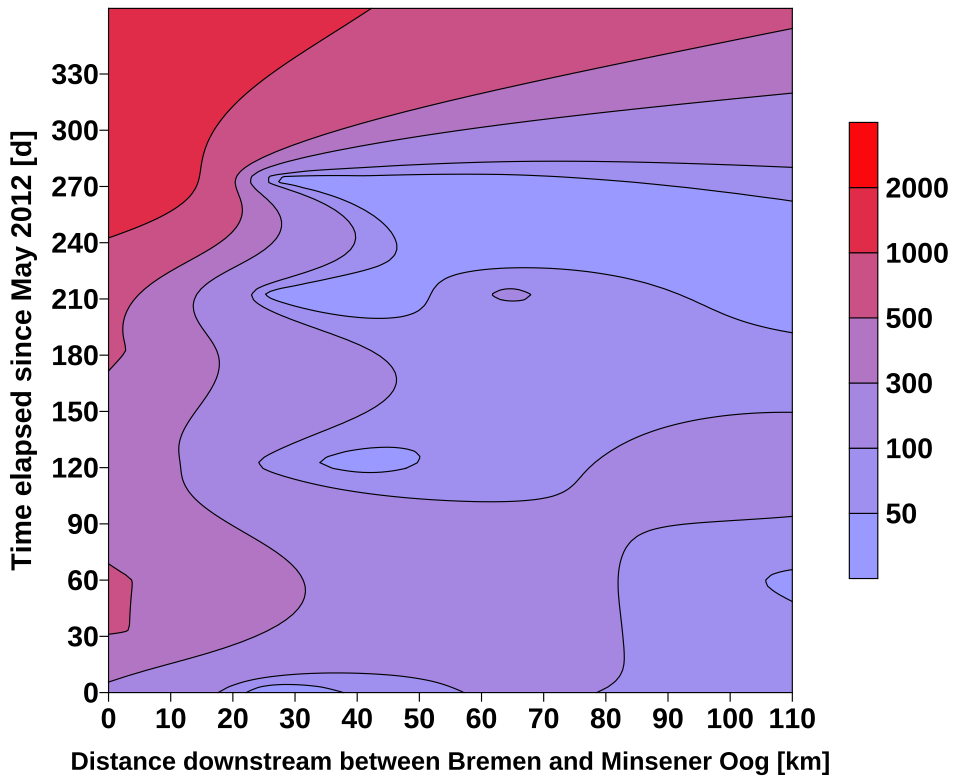


Figure 6-14: Contour plot of abundances of Total Plastic [-] based on orthogonal Kriging.

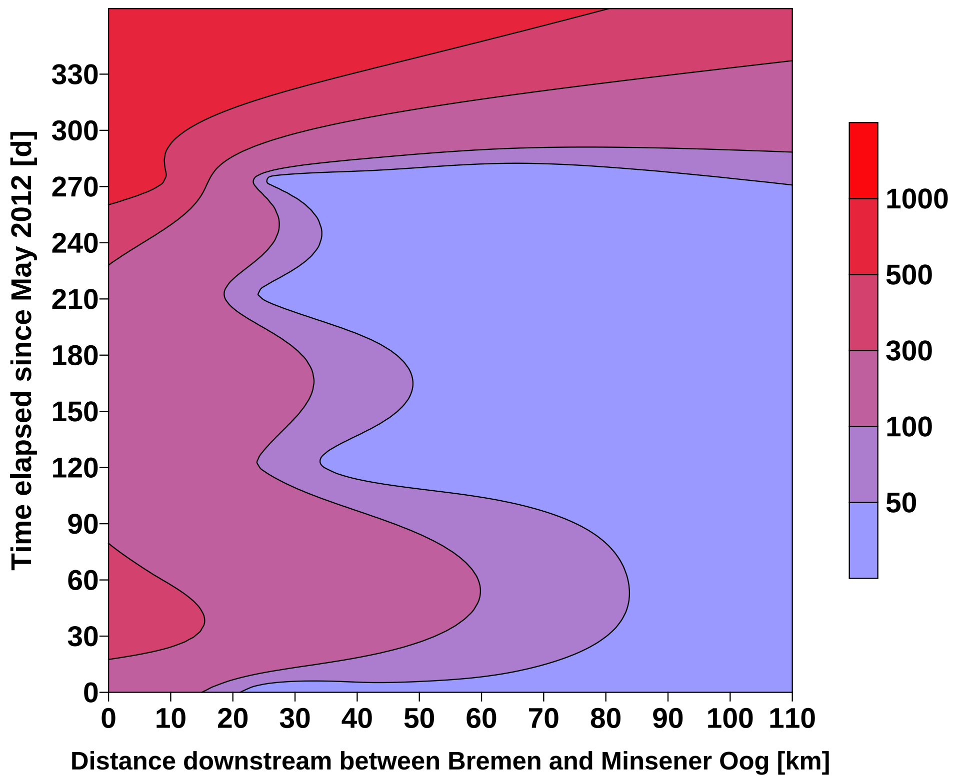


Figure 6-15: Contour plot of abundances of Total Packaging Material [-] based on orthogonal Kriging.

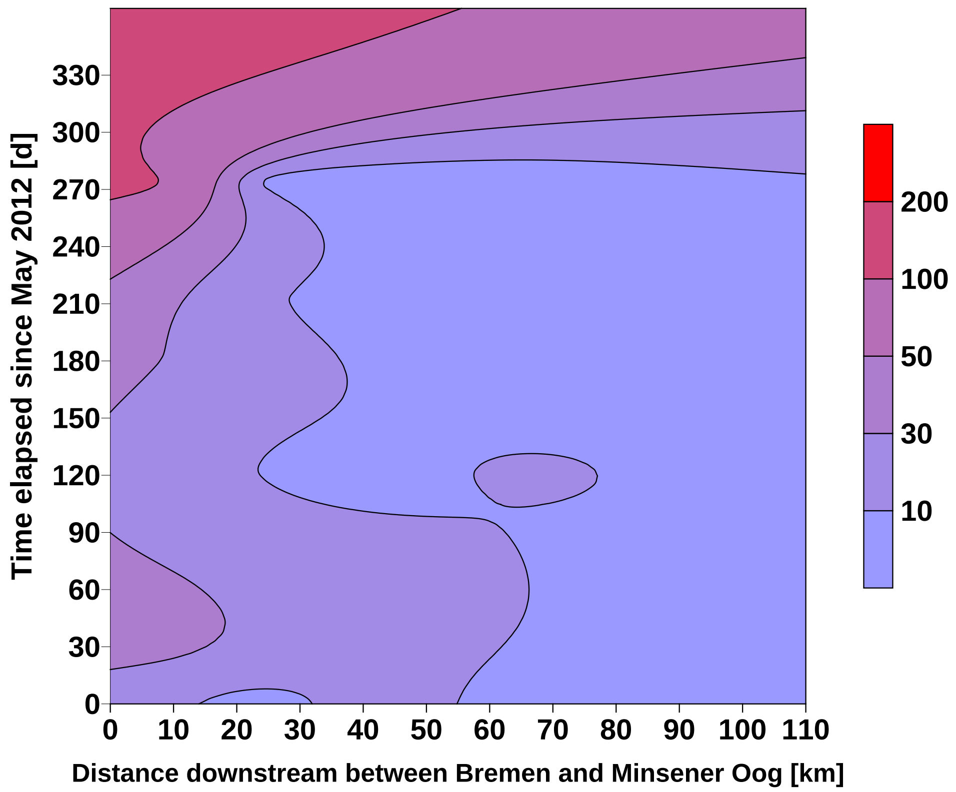


Figure 6-16: Contour plot of abundances of crisp/sweet packages [-] based on orthogonal Kriging.

6.2.2.12 Micro-plastics in beach samples – first results

Sand samples from German beaches off the coast of the islands Norderney (North Sea) and Fehmarn (Baltic Sea) as well as from a number of beaches in the South of France and the island of Corse (Ligurian Sea) have been investigated. First results have been presented at the Workshop "Mikroplastik! Quo Vadis?", which was organized by the Alfred-Wegener-Institute from 5.-6. September 2012 in Bremerhaven (J. Klasmeier: „Analyse von Mikroplastik in Sandproben verschiedener europäischer Strände").

6.2.2.12.1 Norderney samples

A total of six sites have been sampled at the northern coast of the island. Six aliquots of each sample were analyzed for micro-plastics (< 1 mm). Site N3 M and O were chosen to represent samples with (M) and without (O) visible macro-plastics burden on the beach. The results in terms of micro-plastics particles per kg of sand are summarized in Figure 6.17. Only particles identified as plastics by pyrolysis-GC/MS were counted.

Total number of particles ranged from zero to four in the 1 kg samples, which is very low compared to other literature data. However, all particles counted here have been identified as plastic material, mainly polyethylene (PE) or polypropylene (PP), while literature data often solely rely on visual inspection without analytical confirmation. Shape and form of the particles indicate that they are a matter of fragments from larger objects.

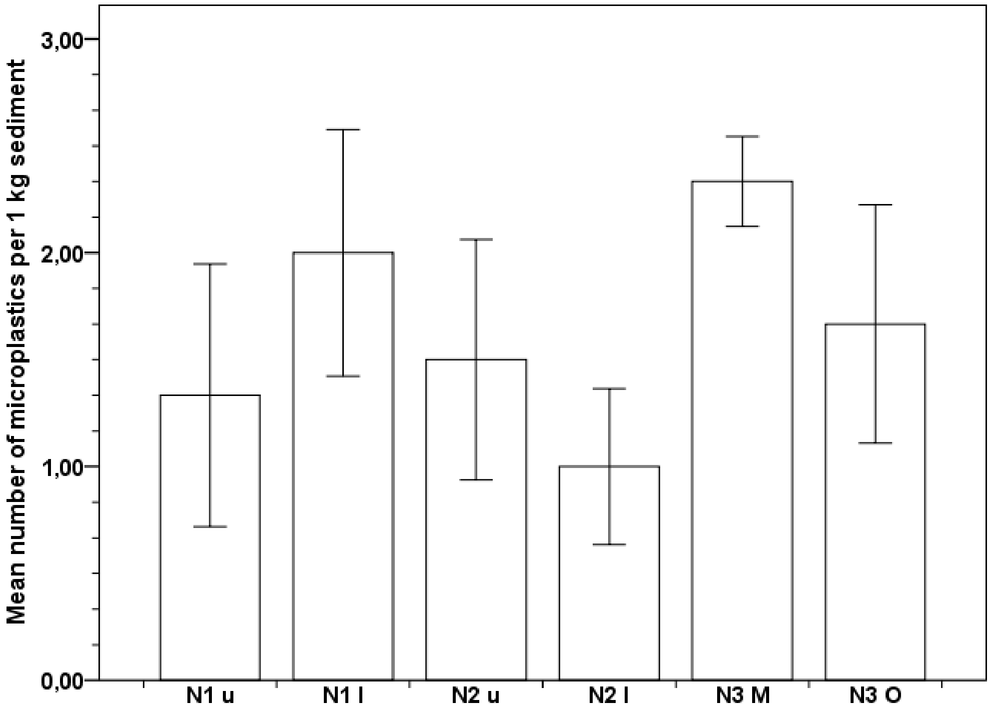


Figure 6-17: Number of identified micro-plastic particles (mean of 6 replicates) in 1 kg samples from Norderney beach (error bars indicate standard deviations).

A clear difference in micro-plastics counts at sites N3 with and without visible macroplastics contamination could not be identified (see overlap of the respective intervals mean ±standard deviation in Figure 6-17).

6.2.2.12.2 Fehmarn samples

A total of six sites from three different beaches in the North and in the West of the island have been sampled and analyzed. Again, six aliquots (replicates) of each sample were analyzed. None of the samples was free of micro-plastics (< 1 mm) with counts ranging from 3 – 10 per kg of sediment sample. For one sample, an exceptionally large number of particles (47 per kg) was detected by visual inspection, but pyrolytic identification was not successful for more than 50% of the material. Therefore, it is likely that false positive counts are partly responsible for this outlier. Investigations are ongoing to verify the number counts of micro-plastics in the various samples.

6.2.2.12.3 Mediterranean samples

In total, 58 samples from 12 beaches (French Mediterranean coast: seven, Corse: five) were analyzed for micro-plastics. Total number of particles counted under the microscope ranged from 35 – 240 per kg sample. However, further analysis of a subset of these particles revealed that less than 20% are likely to be micro-plastics, while the rest is of different origin. Nevertheless, results indicate that micro-plastics burden on the investigated Mediterranean beaches is somewhat higher (by a factor 2-4) than at Norderney and Fehmarn.

6.3 Trends of marine litter in the water column

6.3.1 Materials and methods

6.3.1.1 Data on the litter abundance and cooperation

Regarding data on marine litter in the water column of German marine waters, two long time series (> 10 years) are available. However up till now, they are not completely processed and analyzed for their use.

One series contains data on marine litter at the sea surface, obtained as by-product of aerial habour porpoise[64] surveys. Presently, these surveys are performed by the Institut für Terrestrische und Aquatische Wildtierforschung (ITAW) of the TiHo Hanover. Four times a year, the aerial habour porpoise monitoring records the abundance of habour porpoises and buoyant items at the sea surface of the German North and Baltic Sea areas. Currently, these waters are surveyed alternating: in one year the North Sea and in the other the Baltic Sea. For the period from 2002 to 2006 the data were processed and analyzed in Herr (2009). Uncertainty in the litter data increases with increasing number of habour porpoise sightings.

The other times series contains data on litter at the sea floor. These data are a by-product of the International Bottom Trawl Surveys (IBTS) which are performed on behalf of the International Council for the Exploration of the Seas (ICES) in the North Sea since 1990. The German exclusive economic zone is surveyed by the Alfred-Wegener-Institute (AWI). Till 1997, surveys were performed quarterly and since then twice a year. The aim of them is to estimate the abundance of fish and other marine species which are relevant for the fishing industry. The caught marine litter is noted in a database. Galgani et al. (2000) processed and published data of one of the 1998 IBTSs. Close cooperation between the AWI and the USF for utilizing the IBTS-data is installed in the frame of this project. On December 3, 2012 a meeting between AWI and USF took place in order to evaluate the cooperation in processing that data, and comprehensive beam trawl and otter trawl data were provided to USF.

6.3.1.2 Spatial and temporal analyses of floating litter and litter at the sea-floor

Observation data of floating litter in the North Sea from 2006 to 2008 and time series of beam trawl data were provided by AWI.

Data of floating litter were prepared and interpolated above defined areas with a spatial resolution of 0.01 decimal degrees, applying the interpolation algorithm Kriging. Beam trawl data of the same area and period (2006-2008) were interpolated applying the same procedures. Both sets of the so-derived raster data were spatially identical and therefore could be subdued to rank correlation analysis (Spearman-r).

Beam trawl data from 2001 to 2008 of two spatial clusters (Cluster 100, pelagic region of the North Sea; Langeoog, Wadden Sea) were spatially interpolated, applying the interpolation algorithm Kriging. Subsequently, interpolated data were integrated above defined areas. Data from each sampling period were treated separately, so that two time series of integrated beam trawl data were generated. Temporal trends of these time series were analyzed performing Spearman rank correlation analyses between integrated data and time.

[64] habour porpoise (engl.) = Schweinswal (dt.)

All geostatistical analyses were carried with the geostatistical software Surfer 8.0 (Golden Software, USA, http://www.goldensoftware.com/products/surfer).

6.3.2 Results and Discussion

6.3.2.1 Spatial and temporal trends of floating litter and litter at the sea-floor

There was poor correlation between temporal trends and compositions of pelagic litter and beach litter. However, results of rank correlation analyses of litter at the seabed (Cluster Langeoog) with time partly indicate a significant decreasing temporal trend (r = 0.73, p = 0.003, n = 14).

Rank correlations between floating litter and beam trawl data were poor, and areas of high litter density of both datasets did not overlap, perhaps reflecting different sources or floating/sinking behavior of litter pollution (Figure 6-18).

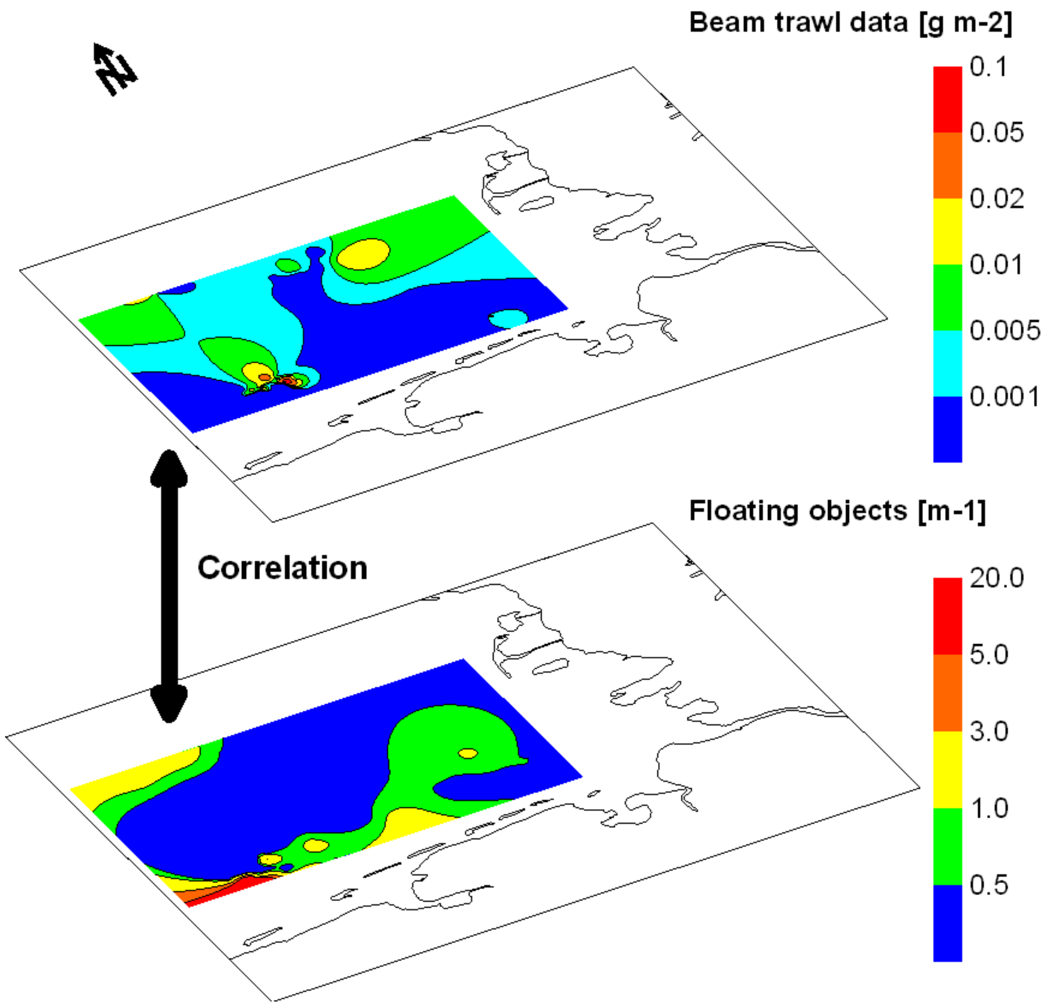


Figure 6-18: Contour plots of the German Bight based on interpolated data from beam trawl surveys and ship surveys of floating litter.

Hinojosa et al. (2011) also found scarce evidence for a relation between floating litter and beach litter. Their correlation analyses between the abundances of floating objects and the distance to the nearest sources were generally non-significant, suggesting that post-supply processes affected the distribution of the floating objects in the study region. Thiel et al. (2011) explained spatial variation of floating litter with differences in spatial distribution of sources, temporal supply, persistence at sea surface, and transport by winds and currents.

6.3.3 Recent literature on marine litter at the sea surface and in the water column

In a recent publication (Kukulka, 2012), the vertical distribution of marine litter in the water column and especially the mixed layer has been modeled. Depending on weather conditions the depth of the mixed layer varied, leading to a variable litter item density and thus to a variable absolute litter amount during surveys catches. During stormy weather, the amount of litter items was underestimated by up to the factor 27 when only the sea surface was surveyed. During calm weather, overestimation was considerably lower. A fixed factor for extrapolating surface data (e.g. a factor 5: 20% items at the surface and 80% below) led to underestimation of the entire litter amount during stormy weather and to overestimation during calm weather. Therefore, during surveys, weather is an important parameter. Additionally, it is questionable, whether sea surface surveys are sufficient to measure the abundance of litter at sea. However, the depth of the North Sea is comparatively low which implies a smaller variability in the depth of the mixed layer and thus less variable absolute litter numbers during surveys.

In 2008, a workshop on the detection of derelict fishing gear (DFG) was held in Honolulu, Hawaii, (McElwee and Morishige, 2010) and was followed by a special issue “At-sea Detection of Derelict Fishing Gear” of the Marine Pollution Bulletin in 2012 (Volume 65, Issues 1–3). The workshop succeeded the Ghost Net project, which was initialized in 2001 and had the aim to develop tools to detect and remove large derelict fishing-nets from the oceans. These efforts do not focus on estimating the quantity of marine litter for scientific purposes or on identifying sources.

McElwee et al. (2012) gave an overview of the effect of biofouling on derelict fishing gear (DFG) and of modeling the aggregation and drift of DFG and applying direct and indirect methods of detection. Mace (2012) suggested a multi-step approach to detect DFG and other marine litter items. In a first step, probable accumulation regions of marine litter should be identified by hydrodynamic transport models. Secondly on the base of satellite data, a spatially extended survey should be performed in order to confirm or reject the model results and to obtain more detailed information. Third in lower altitudes, airborne reconnaissance could provide more detailed information on the abundance of marine litter. McElwee et al. (2012) developed the idea to employ unmanned aircraft systems (UAS) with automated marine litter detection instruments instead of manned planes. For the automated detection of litter, several active and passive sensor systems are under consideration (Mace, 2012; Veenstra and Churnside, 2012). However, until present, no spectral libraries for the purpose of remote sensing focused on marine litter are available, which implies high efforts for determining and validating those spectra. In a final step, litter items could be removed from the marine system.

Lebreton and Borrero (2013) presented a global ocean circulation model coupled to a particle-tracking model to simulate the transport of floating debris washed into the North Pacific Ocean by the Tohoku tsunami. This model assumes that debris particles are mostly submerged in the water and extra forcing on potentially emerged parts of the debris is neglected. This approach led to underestimations of the transport of debris particles that sit high above the water line and are subject to additional wind stress. Further spatial discrepancies between simulation results and litter observations arose from the neglect of wind drift.

Eriksen et al. (2013) collected neuston samples from 48 sites in the South Pacific subtropical gyre. Their results showed an increase in surface abundance of plastic pollution as they neared the center of the gyre and decrease as they moved away, verifying the presence of a garbage patch.

Ryan (2013) applied a simple technique for counting marine debris at sea that revealed steep litter gradients between the Straits of Malacca and the Bay of Bengal. The density of debris in the Straits was correlated with terrestrial vegetation, and peaked close to urban centers, indicating the predominance of land-based sources. Ryan highlighted biofouling-induced sinking and wind-driven export of debris items to explain discrepancies between observations and simulation results of a transport model.

Thiel et al. (2013) investigated anthropogenic marine debris in the coastal environment. High densities of marine litter were found in coastal waters and on local shores of a large bay system in northern-central Chile. No seasonal pattern in marine litter densities was found, but there was a trend of increasing densities over the entire study period. While plastics and Styrofoam were the most common types of marine litter both on shores and in coastal waters, marine litter composition differed slightly between the two environments. The authors observed no clear seasonal variation in floating litter, which probably is due to the fact that no major climatic events (e.g. strong El Niño with heavy rainfalls) occurred during the study period.

6.4 Proposals for future monitoring of marine litter

The following suggestions for a future monitoring of marine litter in the German North and Baltic Seas is mainly based on proposals made by the Technical Subgroup 'Marine Litter' (TSG ML) of the MSRL. Amendments and changes rely on results of the expert workshop on "Umsetzung der Meeresstrategie-Rahmenrichtlinie (MSRL) – Monitoring von Abfällen im Meer" in Dessau on October 8-9, 2013. Here, we present the essence of recommendations of the above-mentioned workshop.

6.4.1 Monitoring of beach litter

We recommend application of the OSPAR protocol of beach litter monitoring (OSPAR, 2010) with modifications, which mainly have been proposed by the TSG ML.

6.4.1.1 Categorization

The TSG ML recommends applying the application of an extended master list of items with additional levels, which relate marine litter to its sources. We agree to this proposal, but require additional information on the master list, such as a fotoguide. New items, such as tape, have to be included in the master list. For ease of beach surveys, creation of an app for smartphones and tablet PCs is necessary.

Completely hidden objects and fragments should not be counted and removed from survey sites, because depths of intrusion are variable. A lower size limit of 2.5 cm in the longest dimension is proposed for non-identifiable objects. Objects, which can be attributed to items, shall also be counted, when their sizes are smaller than 2.5 cm.

6.4.1.2 Spatial and temporal resolutions

The TSG ML recommends to survey at least two replicates of OSPAR 100m-sections of beaches and to cancel monitoring of the OSPAR 1000m-sections. In an initial one-year-long pilot phase, the number of replicates should amount to at least five. Subsequently based on results of statistical analyses of these data, an optimal number of replicates can be figured out. Surveys should cover the entire beach of 100m-sections and not only the strandline.

In the above-mentioned pilot phase in the Baltic Sea, a high number of beaches (n = 20) shall initially be monitored, in order to identify representative survey sites based on statistical analyses of results of the pilot phase. Subsequently the number of survey sites can be reduced to those beaches, which are representative.

Monitoring data of the NGOs Mellumrat e.V., Jordsand e.V., and Schutzstation Wattenmeer e.V shall exemplarily analyzed for the influence of high frequencies on survey results. In addition, OSPAR monitoring data (beaches on Sylt and Minsener Oog) shall be tested and compared with these data. In the pilot phase, intervals of four weeks shall be applied. Thus, effects of seasonal trends potentially masking long-term trends shall be described. Long-term trend analyses can benefit from analyses of seasonal trends considering seasonal variations in raw data treatment. Furthermore, it shall be investigated whether there is a steady state of stranding and erosion of beach litter after four weeks. In addition, event-based monitoring shall be carried out, in order to quantify effects of meteorological and touristic events.

6.4.1.3 Spatial extension

Surveys of accumulation zones (hot spots) are barely practicable because of associated high amounts of marine litter. However, such surveys are sensible in order to relate beach litter to spatial sources and to source categories, such as shipping, tourism, and fishing. Thereby in addition, spatial variation of the entire beach litter database would considerably be increased. Survey results of hot spots can be interpreted by means of Lagrangian transport simulations. Recording of labels of beach litter, scanning bar codes, detection of the age of beach litter and categorization of fishing nets can give further information on sources of beach litter.

Surveys of beaches at estuaries and coastal waters of the Baltic Sea give additional information on sources. For the same reason, limnetic waters could be investigated for pollution with litter also if this is not part of the MSRL. In this context, retention zones, such as flood barriers and embankments, are preferred sites of investigation, although permeability of flood barriers is highly variable decreasing their representativity.

For reasons of spatial representativity, it is important to survey inner coastal areas of barrier islands, such as dyke areas and sandy beaches. In the past, surveys of salt marshes gave very low abundances of beach litter, and accessibility of salt marshes is low, which are severe arguments against surveys of salt marshes. Surveys of the inner coast of barrier islands should be combined with the existing FFH-monitoring, in order to avoid disturbances of biota and ecosystems.

6.4.1.4 Miscellaneous

The influence of consideration of fragmentary items on trends of abundances of general categories, such as total plastic, can be quantified by comparisons of trend analyses with and without inclusion of fragmentary items. Survey results from the North and Baltic Seas should be compared separately. Subsequently, the master list should be reevaluated based on results of these comparisons. Weighing beach litter could quantify the portion of fragments of general categories. However, weighing of beach litter is not practicable at least at the coast of the North Sea, because here abundances are considerably higher than in the Baltic Sea. The weight of beach litter is dependent on its porosity and associated water content, as well as on contamination with beach sand. Each category should be weighed separately if weighing is carried out nonetheless, because the gain of information by detecting the weight of total beach litter of a 100m-section would be negligible.

Collection activities of beach litter, as they are performed by the BUND at the coast of the Northern Sea, can alter results of surveys. Therefore, it is recommendable to survey beaches in close vicinity to each other, with and without litter collections, in order to quantify their effects on survey results. Analyses of the content of litter boxes can give additional information on the composition of beach litter. By comparison of survey results from touristic and non-touristic beaches in close vicinity to each other, the influence of local tourism on beach litter abundances can be quantified.

Beaches with high abundances of marine litter (> 1000) are difficult to survey. Therefore, we recommend reducing lengths of heavily polluted beach sections to 50 m or 25 m. Alternatively, the monitoring method by Bravo et al. (2009) could be applied, who recommend investigating beach litter composition in randomly selected squares. However, results of the OSPAR method and the method by Bravo et al. (2009) are not comparable, which would lead to inconsistencies of monitoring results.

Samplings of mesolitter should be carried out at OSPAR beaches, in order to enable correlation analyses between time series of macrolitter and mesolitter.

6.4.2 Monitoring of marine litter at the sea surface

Both opportunistic and concerted surveys of floating litter should only be carried out under defined conditions. The following definitions are recommended:

1. Oceanographic and meteorological conditions should be comparable (wave action, tides, light conditions, wind forces).
2. The observation height should range between 3 m and 10 m. In maximum, the width of observation transects should be 5 m to the left and to the right, respectively.
3. In maximum, the velocity of the ship should amount to 5 bends. The ship should move in a direction rectangular to the wind direction.

Overall, defined conditions are necessary to reduce variability and to increase comparability of results of surveys.

Results of surveys made from FS Heincke (AWI) showed a spatial gradient between inshore and offshore areas. Survey sites should be representative, but hot spots should be surveyed as well. Potential survey sites are the East and North Frisian coasts of the North Sea, the Elbe Estuary, and the region around the island Helgoland. In the Baltic Sea, appropriate survey areas could not yet be identified. In the Baltic Sea, surveys of the Thünen Institute could not detect any floating litter.

Opportunistic use of regular shipping lanes is barely possible if defined conditions have to be followed. On research vessels, personal capacities are usually low. Therefore, it is barely practicable to deploy two observers for extra investigations, such as observations of floating litter. For the same reason, opportunistic applications of manta trawls are not practicable.

6.4.2.1 Categorization and counting of floating litter

Even at slow movement of a ship at high abundances, it remains difficult to survey the entity of floating litter. Thereby, collected data become incomplete. Generally, a detailed categorization, such as that made in the OSPAR protocol (OSPAR, 2010), is not sensible. Based on the master list, categorization according to size can be made. Three size classes should be distinguished, namely small, medium, and large. Additional photos are helpful to amend incomplete observations. Video recordings and subsequent comparisons of both methods can be used for quality control and teaching purposes.

6.4.2.2 Needs for future research

Visual surveys should be verified by means of trawl surveys, because probably visual surveys underestimate actual abundances. This verification could be used to derive a conversion factor.

Automatized surveys with cameras are yet not ready for a routine monitoring. Therefore, visual observations remain the method of choice. However, two steps of development of camera surveys are suggested:

1. For quality assurance (storing) and for a comparison of methods, deployments of cameras and visual observations shall be carried out in parallel. Survey methods shall be tested from fixed positions. Thereby, costs can also be reduced.
2. Automatized image evaluation procedures have to be developed.

Sampling of marine litter from the sea surface probably gives more precise results than camera-based techniques, but until now it remains unclear which method is most efficient.

The wind drift of litter fractions floating at the sea surface shall be investigated experimentally, in order to optimize surveys and to link survey results to litter sources by means of Lagrangian transport simulations. Stationary platforms, as they are already installed on River Thames in London, could be deployed in estuaries as potential source regions for sampling litter. It is recommendable to calibrate survey methods by means of an experimental approach with defined amounts and kinds of marine litter. Investigations of seasonal variations of abundances of floating litter are necessary to optimize survey periods and frequencies.

6.4.3 Monitoring of marine litter at the seafloor

Monitoring programs of marine fish populations (Thünen Institute, IBTS) can be used for parallel monitoring of marine litter at the seafloor. The spatial resolutions of existing monitoring programs are sufficiently high for a large-scale balancing of marine litter at the seafloor of the North Sea. However, for a sufficient number of replicates, bottom trawls of several years should be merged. A target value could be > 10 replicates per sampling.

Existing datasets should be analyzed for seasonal differences if the number of replicates per season is high enough. Data of the AWI were merged over periods of several years. Here, seasonal differentiation was not possible. In the North Sea, identification of hot spots is apparently possible when data by the Thünen Institute (n = 72/year) are evaluated for periods of several years.

For sampling of nearshore areas, we propose the deployment of chartered fishing boats. We suggest continuing the sampling of the Polaris cluster near the island Langeoog.

Investigations on fish populations by the NLWKN (3m-beam trawl) could be used for a sampling of marine litter. In the Baltic Sea, the bottom trawl equipment is more robust than in the North Sea, because of the prevailing hard substrates at the seafloor in the Baltic Sea. Therefore here, the method applied in the North Sea has to be adapted. In coastal regions, the initiative Fishing for Litter by NABU could deliver amending data on marine litter at the seafloor.

It is necessary to use a highly resolved categorization, such as that given in the OSPAR protocol for beach litter monitoring (OSPAR, 2010), in order to relate survey results to sources of marine litter. Assessments by experts of fishing and shipping should be made for the same reason. For example, within the initiative Fishing for Litter, nets are related to different trawling techniques in various countries. In sensitive areas, such as the national park Wadden Sea, in the region of Helgoland, and in accumulation zones, which are difficult to survey, divers could be deployed in subtidal shallow waters. Video-based investigations are only applicable above homogeneous substrates.

6.4.3.1 Needs for future research

Seasonal aspects of marine litter at the seafloor should be examined. Here, land-based extreme input should be considered. Systematic comparison of efficiencies of trawl equipment and mesh sizes is necessary, in order to enable comparisons with existing datasets of AWI. Video-based surveys within the framework of the deployment of divers should be tested for diverse substrates. Developments of techniques for sensitive areas (Gopro cameras, ROVs) are desirable. New data on ghost nets by BSH should be further evaluated. Until present, only a small portion of ghost net data could be evaluated. In the North and Baltic Seas, ghost nets should be examined for catch efficiency and diversity of nets at least exemplarily by divers. Systematic comparisons of marine litter in different compartments of estuaries (water surface, water column, water bottom) are desirable for quantifying input to the North Sea via different pathways and for concerting subsequent reduction measures.

6.4.3.2 Ghost nets

In the North Sea until present, AWI investigated 65 wrecks for ghost nets. The recovery of ghost nets is difficult and cannot be carried out together with scientific investigations. In the North Sea, a Dutch initiative investigates wrecks and recovers ghost nets. Here, photographs evidence the efficiency of ghost nets concerning ghost fishing. Information on ghost nets can be gathered at http://www.ghostfishing.org and http://www.ghostnets.com.au.

At BSH, data on ghost nets are available. In the Baltic Sea, fishers use maps, on which wrecks with ghost nets are documented. Until present, the German Marine Museum has investigated twelve wrecks and their ghost nets for their future recovery. In Poland, trawl equipment with hooks is commonly used to scour the seafloor for lost fishing gear.

6.4.4 Monitoring of microlitter

6.4.4.1 Subdivision into size classes

The proposal of the TSG ML concerning subdivision of marine litter into four size classes (> 2.5 cm = macrolitter, 0.5 – 2.5 cm = mesolitter, 1 – 5 mm = large microplastic particles (L-MPP), and < 1 mm = small microplastic particles (S-MPP)) should be accepted as a standard to ensure comparability of future monitoring programs.

For the same reason, a lower size limit has to be defined, also in order to exclude nanoparticles (< 100 nm) from analyses. At present, particles can be analyzed for material composition down to a size of 1 µm, although the Technical Subgroup on Marine Litter proposes to consider only particles down to a size of 20 µm. The definition of the lower size limit could rely on relevance concerning accumulation of microparticles in biota and within the food chain.

6.4.4.2 Monitoring of biota

Under realistic environmental conditions, there is no clear evidence of ingestion and accumulation of microplastics by biota. Probably, the major part of micro- and mesoparticles is released after passage through the digestive tract of biota. Accumulation may only occur when for physiological reasons microparticles cannot be released. The respective pathway of ingestion should then be considered for the planning of future monitoring. Very small particles might penetrate into tissue and cells, which might cause harm of biota. Effects of transient accumulation in biota should be investigated in pilot studies. At sufficiently long transport time within the digestive tract and high rates of ingestion, microparticles might be disseminated to the food chain.

There is a strong need for research concerning potential leaching of organic pollutants, which are added to plastic. Under conditions altered in the stomach to low pH values, the release of such pollutants might be reinforced. However, in biota, residence times of small particles are usually considerably lower than in sea water, so that the major portion of additives and relicts of plastic monomers is probably leached out in the water phase. Microplastic has been considered as transport vector for the uptake by biota via sorption. This process is negligible, because of the low surface area of microplastics compared to the volume of sea water and competition between sorption and ingestion as food.

Investigations on faeces of mussels (Mytilus edilus) are promising, because thus an integral parameter describing the pollution of filtered water can be defined. Irrespective of the fact that the volume of filtered water cannot be quantified because of highly variable filtration rates in turn depending on environmental conditions, future research should elucidate whether faeces of mussels should be monitored as indicator of pollution with microplastics.

6.4.4.3 Analytics

Specific identification of all particles potentially being microplastic particles is a prerequisite for subsequent analyses. Single particles < 1 mm can be identified with small effort (5-10 minutes/particle) by means of ATR-FT-IR or Raman spectroscopy. Small particles down to a size of 1 µm can be analyzed after sample preparation on a filter by means of a scanning method. The temporal effort of this method is considerably higher than that of analyses of single particles. For analyses of single particles (> 1 mm) and their additives, pyrolytic GC/MS with upstream thermo-sorption is an appropriate method.

6.4.4.4 Monitoring in the water phase

Sampling of the upper water column down to a depth of 10 to 20 cm should be preferred. In analogy to size classes, we suggest using nets with mesh sizes of 1 mm or 5 mm for a routine sampling. At small mesh sizes of 333 µm, problems arise from clogging of filters and nets. At any rate thereby, the volume of the sample may be considerably confined. The filtered volume serves as reference parameter and is measured using a flow meter. Sampling of mesolitter and L-MPP can be integrated into a routine monitoring, because little sample preparation is needed. Definition of standard procedures and equipment (nets, sampling depth, sampling volume) prior to starting the monitoring is important. Blank values due to usage of nets made of synthetics are assumed to be negligible.

Alternatively, pump system can be used, which filter sea water on ship or in laboratory. Online filtration is preferred, where the filter should be installed upstream of the pump, in order to avoid mechanical damage of the pump by suspended particles. Such pump systems have been found to be reliable for water depths of 6 m, but not for sampling at the water surface. Efficiency and cost efficiency of this method have to be tested, and a standard protocol of its application has to be developed. For reasons of cost efficiency, sampling frequencies of both methods, applications of nets and pumps, depend on availabilities of windows of opportunity, such as ship excursions of BSH.

For S-MPP there is considerable need for research concerning the present pollution and sources. Emissions from households via water treatment plants have not yet been clearly evidenced. There is no reliable information on concentrations and amounts of S-MPP in treated and untreated sewage water. Emissions from water treatment plants are probable, but not proven. Fibers of textile are emitted from washing machines into sewage water. In samples of treated sewage water, such fibers have been detected. In the DELTARES project, fibers were also detected in effluent sludge, which was yielded in small amounts as fertilizer onto arable land. A pilot study should be initiated, in which input and output of microplastics in water treatment plants are balanced. Similarly, balancing of microplastics in river systems from crenal regions to river mouths is regarded as sensible. In analogy to investigations of marine waters, samplings at river mouths could be performed by means of drift nets. Sediment samples should be taken as well. Within an F&E project, UBA balances total production of microplastic by cosmetic industries and its release into the environment.

6.4.4.5 Monitoring of beach sediments

The proposal of the TSG ML to reduce volumes of sediment samples by sieving (mesh sizes 1 mm and 5 mm) was objected, because sieving leads to additional fragmentation of microplastic. In the Wadden Sea, sieving was applied successfully. Therefore, we recommend sieving the mesolitter fraction and identifying mesolitter particles in laboratory. Microlitter should not be sieved. However, prior to a routine monitoring, the informative value of such analyses has to be figured out. According to the state of the art, high temporal effort for sample preparation by flotation (1 day/sample) is necessary. Both fractions of sediment samples, S-MPP and L-MPP, should be analyzed in parallel, in order to save analytical time.

Sampling (location, sample size) is critical concerning the informative value of results. Principally, the upper 10 cm of beach sediment should be cored. Until now, no recommendation can be given in which beach section sediment samples should be taken. Therefore, random sampling is proposed. Two sample locations per beach are suggested: one within the recent strandline and another between the recent strandline and the upper edge of the beach. In the Baltic Sea, the strandline is more stable than in the North Sea. Therefore, in the Baltic Sea, sampling along transect perpendicular to the strandline could be used to quantify the stranding of microplastic.

However, at first, it should be clarified if there are any spatial gradients decreasing towards the upper beach. Therefore, examinations of spatial patterns are recommendable, before sampling is carried out along transects perpendicular to the strandline. The proposal of the TSG ML to use metal square frames of 50 cm x 50 cm for random sampling was rejected.

In conclusion, existing basic information on pollution of beach sediments with litter of different size is not sufficient for planning an informative and cost-efficient monitoring of beaches for micro- and mesolitter.

6.4.4.6 Monitoring of subtidal sediments

Microlitter has been detected at the seafloor of subtidal areas. The density of microparticles can be increased by biofilms so that the particles settle to the seafloor instead of floating on the water surface. Sedimentary records could elucidate historical developments of pollution with microplastic. However, there is a strong need for research on this topic.

So far, grab samples have been routinely taken for purposes other than investigations on microplastic. Such windows of opportunity (ship excursions of the BSH and regional agencies) could be used for samplings of microplastic. In estuaries, such as the Elbe estuary, sampling along transects from upstream regions to the North Sea could deliver valuable information on emissions from terrestrial areas.

6.4.5 Monitoring of ecological effects of marine litter

6.4.5.1 Ingestion by fish

There is high need for more research on potential biotic indicators. The following fish species are proposed for the monitoring of ingestion of marine litter:

- Herring (pelagic in the North and Baltic Seas),
- Mackerel (pelagic in the North Sea, perhaps also in the Baltic Sea). Presently, Mackerels are spatially spreading and therefore can be sampled in large regions. Around the island Helgoland during late summer, high ingestion rates of net fragments by mackerels have been observed, coinciding with the presence of pipe fish, a natural prey of mackerels. Perhaps, this phenomenon can be correlated with the presence and breeding of northern gannets on Helgoland.

- Codfish (demersal in the Baltic Sea), feeding non-selectively and eating also large objects,
- Cod (demersal in the North Sea), has become very rare,
- Whiting (demersal in the North and Baltic Seas),
- Dab (benthic in the North Sea),
- Fluke (benthic in the Baltic Sea),
- Eelpout (regionally important in coastal waters),
- Sea scorpion (regionally important in coastal waters).

The subdivision of fish should follow size classes rather than age classes (e. g. dab: 20-25 cm = two years old, > 25 cm = three years old and older). Information on age is still available after examination of fish in laboratory. It should be differentiated between planktonivorous fish, mollusk eaters (flatfish), and piscivorous fish. In addition, feeding behavior (selective, non-selective) should be used as distinctive feature. There are some hints on increasing pollution of fish with marine litter with increasing size and age.

In the North and Baltic Seas, Thünen Institute regularly conducts investigations on fish diseases, thus providing sufficient sample material for a monitoring of plastic ingestion. However, for routine analyses, additional personal capacities are required.

The frequency of sampling should be twice a year, preferably in late summer and in winter. Samplings over complete annual cycles are enabled by existing investigations on fish biology. For sampling, migrations of fish populations should be considered. Investigations on fish biology are advantageous, because thereby a comprehensive health feature of fish is generated by detections of visible diseases, changes of organs, especially of the liver, liver weights, and blood samples. The standard ICES protocol for investigations of fish diseases should be modified, to allow for investigations on ingested particles of marine litter.

Preferably, fish should be examined directly after their catch. Only the stomach should be stored frozen for later analyses. Effects on defrosted fish are difficult to observe due to self-digestion after death, especially during freezing and defrosting. Fish individuals should be frozen separately if immediate examinations are not possible.

The faeces of fish held alive could be used for analyses for microlitter, because fish have short residence times in their digestive tracts. The sample size should amount to 40-50 individuals. Analyses should be performed at faeces of the entire sample. There is still no information whether mackerels excrete plastic particles.

It is recommendable to regard the abundance of litter particles and not their mass. The protocol proposed by the TSG ML has to be adapted insofar particles > 1 mm length should be regarded and not only particles > 5 mm length.

A pilot study funded by UBA investigates pelagic fish and fluke for incidence of plastic particles and potential transfer of pollutants from plastic to tissue. The environmental database by UBA includes samples of eelpouts, breams, and blue mussels. Linking of activities of Thünen Institute, UBA, and its environmental database are desirable.

6.4.5.2 Ingestion by birds

Pellets and faeces of large sea gulls and terns should be examined for microlitter. Black-headed gull, herring gull, and lesser black-backed gull are further potential indicator species. However, sea gulls feed in both terrestrial and marine environments. The Arctic tern is highly appropriate for investigations on ingestion of microlitter, because this species solely feeds on schooling fish and sand eel. Probably, microlitter is ingested via prey fish. In the Baltic Sea, breeding colonies can easily be accessed in bird sanctuaries.

6.4.5.3 Ingestion by marine mammals

Amounts of macro- and mesolitter, as well as their incidence in stomachs of seals and habor porpoise, are not sufficient for a routine monitoring. Investigations of faeces of seals (common seal, gray seal) for microplastic are practicable (1 day/sample analyses) and sensible. Microplastic is ingested by marine mammals via feeding on prey fish. At the coast of the North Sea, the Kachelotplate, Norderney, and Helgoland are potential sites for sampling. In pilot studies on guts and stomachs of habor porpoise, considerable amounts of microlitter have been found. Results of monitoring of microparticles in mammals could be correlated with results of investigations on prey fish. It is recommendable to contact the seal station in Norden and to ask for the contents of dissected seals.

6.4.5.4 Ingestion by mollusks and crustaceans

In the North and Baltic Seas, a monitoring of blue mussels is established. In the context of monitoring pollutants, tissues of blue mussels could be annually sampled and analyzed for microparticles. Sampling should be carried out annually in late autumn, because then the mussels are less stressed. Results of a pilot study of AWI on crustaceans (brown shrimp) should be requested.

6.4.5.5 Investigation of toxicity

Investigations of toxicity should focus on metabolites (detection by antibodies) rather than on additives, because living fish metabolize certain additives. In marine waters, the concentrations of POPs are low. In Biota and sediments, POPs are detectable, but probably microplastic is of low importance as potential vector of POPs, especially when compared with other suspended matter.

6.4.5.6 Entanglement

Potential indicator species of entanglement have to be identified among northern gannet, kittiwake, large gull, cormorant, European spoonbill, and bald coot. Appropriate breeding locations, preferably on islands, have to be identified. Nests of cormorants are partly difficult to observe, because cormorants also breed on sea marks.

NGOs supervising breeding areas and ornithologists should be embedded within the monitoring of entanglement, focusing on litter in nests and identifying appropriate breeding colonies and alternative indicator species. Visual surveys of nests of cormorants in trees with spotting scopes are recommended. Perhaps after the breeding season, nests of cormorants could be removed for subsequent analyses for litter.

Surveys of dead birds along the coast of the North Sea are sensible only in part when the existing protocol of NLWKN is modified. At the Baltic Sea, surveys of dead birds are not recommendable, because here scavengers remove dead birds.

6.4.5.7 Quality objectives

Reference regions with low pollution have to be identified, as they already exist for the northern fulmar. For fish, such regions could be the Atlantic around Iceland, which has been investigated for organic and inorganic pollutants by Thünen Institute. Alternatively, significant decreases can be defined as quality objectives. For identification of stagnating or decreasing trends, long-term trend analyses have to be carried out. These trend analyses should use single categories and serve for testing the effectiveness of countermeasures against pollution.

6.4.5.8 Miscellaneous

The fish grub database of Thünen Institute can be used for investigations on microlitter, because within the database long time series are available. Sampling is performed with neuston nets. Samples are stored in formalin.

PE-film collectors for monitoring of pollutants could be used for investigations on biofilms and incidence of invasive species. However contrary to sampling of floating litter probably, film collectors are not the equipment of choice for these purposes.

6.5 Meetings, Conferences and other Activities

A first meeting between researchers from the TiHo Hannover (Prof. Ursula Siebert, Dr. Helena Herr, Sebastian Müller) and the USF (Prof. Michael Matthies, Dr. Marcus Schulz, Daniel Neumann) regarding potential use of the litter data from habour porpoise monitoring data took place in Hanover in October 2012.

A meeting between the USF and the project partner Bioconsult GmbH took place in Bremen on November 9, 2012. Information on marine litter in estuaries and on beaches was presented and exchanged, and further collaboration was concerted.

On January 21, 2013 in Osnabrück, USF arranged a project meeting with all partners of work package 5. USF presented statistical and model results, as well as the evaluation system proposed, which was acknowledged by all attendees. A journal of the meeting has been approved by all participants and was sent to the overall project manager (Dr. Claus-Dieter Dürselen, AquaEcology GmbH).

Cooperation with Dr. Ulrich Callies from the Helmholtz-Center in Geesthacht on transport modeling and simulation of marine litter in the North Seas was continued. Main research activities have been focused on better understanding of the relation between sources and receptors (beaches) and its seasonal and spatial variability.

Results of the particle transport simulation with PELETS-2D have been presented at the Workshop 'Mikroplastik! Quo Vadis?', which was organized by the Alfred-Wegener-Institute from 5.-6. September 2012 in Bremerhaven (M. Matthies, D. Neumann, S. Lotter, U. Callies: „Transportsimulationen von Plastikpartikeln in der Nordsee“). J. Klasmeier presented results of microplastic analyses (“Analyse von Mikroplastik in Sandproben verschiedener europäischer Strände“).

M. Schulz, D. Neumann, and M. Matthies participated in OSPAR beach surveys on the OSPAR reach on Juist (Germany) on September 29, 2012 and April 22, 2013, respectively. Participation in beach monitoring served to collect knowledge of strategies and problems of beach litter surveys.

From April 10 to April 12, 2013, Prof. Michael Matthies and Dr. Marcus Schulz (USF) attended the 'International Conference on Prevention and Management of Marine Litter in European Seas' in Berlin. USF presented two posters and acquired data on beach litter pollution in the Baltic Sea from the non-profit environmental organization NABU, as well as new OSPAR beach litter monitoring data of 2012. Besides, networking resulted in initialization of new cooperation of USF with NGOs and scientific institutions.

In close cooperation with Stefanie Werner from the German Environment Agency (UBA), USF organized an expert workshop on “Umsetzung der Meeresstrategie-Rahmenrichtlinie (MSRL) – Monitoring von Abfällen im Meer” in Dessau on October 8-9, 2013. The workshop dealt with five topics, namely a) beach litter, b) marine litter in the water column, c) marine litter at the seabed, d) microplastics, and e) biological indicators of marine litter. Accordingly, four discussion groups were implemented, where the topics marine litter at the seabed and floating litter were merged to one group. Discussion within these groups contributed to a final paper (Schulz et al. 2013), which will be made available to all participants of the workshop and other committees, such as the TSG ML and the OSPAR IC Group 'Marine Litter' (OSPAR ICG ML). On the workshop, Prof. Dr. Michael Matthies gave a plenary introduction into the topics of the workshop. Dr. Marcus Schulz, Dr. Roland Krone, Dr. Jörg Klasmeier, and Stefan Weiel gave plenary talks on monitoring of beach litter, marine litter in the water column and at the seabed, microlitter, and biological indicators, respectively. All five were active as chair or rapporteur of the working groups.

At TiHo in Hanover on October 29, 2013, Dr. Marcus Schulz attended a workshop on evaluation strategies within the MSFD and gave a plenary talk on evaluation systems proposed for marine litter.

Furthermore, Dr. Marcus Schulz attended the annual meeting of the OSPAR ICG ML in Hamburg on November 5-6, 2013, as well as the Workshop on the development of an OSPAR Regional Action Plan on Marine Litter at BSH in Hamburg on November 6-7, 2013. At both occasions, Dr. Marcus Schulz gave plenary talks on results of statistical analyses of marine litter data.

Two peer-review manuscripts were written under guidance of USF and submitted to scientific journals. One of these manuscripts is in press in the scientific journal 'Marine Environmental Research' (Schulz et al. 2014), while the other manuscript is under review at 'Marine Pollution Bulletin' (Schulz & Matthies, under review).

6.6 Trends in the amount and composition of litter ingested by marine animals (Indicator 10.2.1)

6.6.1 Continuation of the OSPAR Fulmar-Litter-EcoQO approach as MSFD Indicator 10.2.1

6.6.1.1 Objective and background

Project objective was the application of indicators and monitoring concepts for measuring and assessing the impact of litter in the sea on marine life. Trends in the amount and composition of litter consumed by marine organisms were examined by analyses of stomach contents (MSFD indicator 10.2.1).

The ingestion of marine debris, mainly plastics is known from at least 111 seabird species, accounting for approximately 36 % of the world's seabird species (Laist 1997). Most prone for the ingestion of plastic are Procellariformes (Albatrosses, Fulmars, Shearwaters and Petrels) which mainly forage at the sea surface and often mistake floating plastic debris with natural food (Moser & Lee 1992). They usually do not regurgitate undigestible items but accumulate these in the muscular part of their stomach, where parts are slowly ground down to a size which may pass into the guts (Van Franecker et al. 2011). Hence Van Franeker & Meijboom (2002) chose the Northern Fulmar (*Fulmarus glacialis*) as a suitable species to monitor the ingestion of plastic debris. Fulmars are abundant in the North Sea, the North Atlantic and Pacific, feed exclusively offshore and are often found in beached bird surveys. They are also known to ingest a wide variety of marine litter items (Bourne 1976, Furness 1985, Moser & Lee 1992, Van Franeker et al. 2011, Van Franeker & The SNS Fulmar Study Group 2011). Data on plastic ingestion by Northern Fulmars is available for the Dutch coast since the early 1980s and for most other countries bordering the North Sea since 2002, when a pilot project started as part of the EU campaign "Save the North Sea" (Van Franeker 2004). Acceptable ecological quality was provisionally defined by OSPAR as the situation where less than 10% of fulmars exceed a critical level of 0.1 g of plastic in the stomach (OSPAR 2008). This target level refers to a litter situation in a reference area where the pollution level is considered to be acceptable in terms of environmental quality, i.e. the Canadian Arctic (Van Franeker et al. 2011) The methodology, metric, and data presentation were gradually evaluated and matured and then implemented by the OSPAR Commission as the Fulmar-Litter-EcoQO (OSPAR 2010a, b, Van Franeker et al. 2011). The German programme is part of the international Fulmar-Litter-EcoQO-Study coordinated and led by the Institute for Marine Resources and Ecosystem (IMARES) on Texel, The Netherlands. From 2002-2004 it was financed in the scope of the EU Interreg IIIB Programme "Save the North Sea" (Guse et al. 2005, Van Franeker et al. 2005).

From 2005 to 2010 the FTZ continued the national work on a voluntary basis to ensure an unbroken dataset. Stomach analyses were conducted by IMARES during this period. In 2011 funds from the Federal Agency of Nature Conservation (BfN) enabled the continuation of the German contribution carried out by the FTZ. This included stomach analyses of the more recent German samples and allowed the preparation of a first national report (Guse et al. 2012).

At the instigation of the Federal Environmental Ministry (BMU) and in agreement with the BfN current work on the OSPAR Fulmar-Litter-EcoQO is continued within the framework of the present project funded by the German Environment Agency (UBA).

6.6.1.2 Material and methods

The methodology follows the Fulmar-Litter-EcoQO approach that uses stomach contents of beached Northern Fulmars to measure trends in marine litter (Van Franeker & Meijboom 2002, OSPAR 2008, Van Franeker & The SNS Fulmar Study Group 2011, Van Franeker et al. 2011, Guse et al. 2012).

The continuation of the national OSPAR Fulmar-Litter-EcoQO work in Germany included the ongoing coordination of the network of volunteers, public agencies and conservation organizations involved in the collection of Fulmars during routine beached bird surveys. The coordination requires a permanent attendance and regular information of all persons and institutions involved. Project work also involved organisation of transport of birds found in 2012 and 2013 from various locations of the German North Sea coast to the FTZ. In some cases, the pick-up and transportation of collected birds was conducted by own staff members. The birds were/are stored in the institute's freezing facility.

Several dissections of Fulmars were carried out following the internationally standardized protocol of van Franeker (2004). During the current project 37 stomachs of Fulmars found in 2011, 52 samples of 2012 and several additional samples from earlier years were analysed regarding quantity and composition of ingested litter. The stomach contents were extracted, classified according to different categories and then quantified. For each litter category the incidence, abundance by number, and abundance by mass was assessed. Data was then prepared, analysed and compiled following international standards. In September 2013 the updated German dataset was delivered to IMARES for incorporation into the joint Fulmar-Litter-EcoQO database to enable a joint reporting to OSPAR. To harmonize data preparation as well as presentation we followed IMARES' lead regarding the calculation of averages, EcoQO performance, structure of figures and tables.

6.6.1.3 Results of Fulmar-Litter-EcoQO research

In addition to the data included in the last interim report (Guse et al. 2013) more analyses were carried out including all newly available data. The metric for discussion of amounts of, and trends in plastics in stomachs of Northern Fulmars for the plastic particle EcoQO focuses on the

- 'Current situation' which is described by the average for the last 5-year period (now: 2008-2012) and showing the mass of plastic in each of the different litter categories
- Development of mass of industrial and user plastics in the stomachs of Fulmars over the last 6 periods (2003-2012).
- Annual geometric mean mass of plastic in different age groups of Northern Fulmars found in Germany (2003-2012).

The following results include the latest 2012 dataset along with those of the entire German database covering 527 Fulmar stomachs from the period of 1994-2011 (see Table 6-9).

Table 6-9: Annual details for plastic abundance in Fulmars found in Germany. For separate and combined plastic categories, incidence (%) represents the proportion of birds with one or more items of that litter present, number (n) abundance by average number of items per bird, and mass (g) abundance by average mass per bird in grams. The column on the far right indicates level of performance in relation to the OSPAR EcoQO, viz. the percentage of birds having more than the critical level of 0.1 gram of plastic in the stomach. The bottom line of the table shows the 'current' situation as the average over the past 5 years. Note sample sizes (n) to be low for particular years implying low reliability of the annual averages for such years, not to be used as separate figures. Also note erratic variability in age proportions of birds in samples, where age is known to influence amount of litter in the stomach.

			INDUSTRIAL PLASTICS			USER PLASTICS			ALL PLASTICS (industrial + user)			EcoQO
Year	n	% adult	%	n	g	%	n	g	%	n	g	> 0.1 g
1994	1	0%	100%	2	0,043	100%	31,0	0,512	100%	33,0	0,555	100%
1995												
1996												
1997												
1998	1	100%	100%	2	0,022	100%	35,0	0,194	100%	37,0	0,216	100%
1999												
2000	1	0%	100%	4	0,103	100%	26,0	0,158	100%	30,0	0,261	100%
2001	2	100%	50%	2	0,034	100%	15,5	0,049	100%	17,5	0,082	50%
2002	4	50%	0%	0	0,000	100%	6,5	0,051	100%	6,5	0,051	25%
2003	32	22%	81%	3,9	0,087	94%	24,1	0,356	94%	28,0	0,443	78%
2004	155	74%	56%	2,9	0,058	93%	25,7	0,238	94%	29,0	0,296	54%
2005	71	61%	66%	2,3	0,051	94%	17,5	0,191	94%	20,0	0,242	52%
2006	11	45%	54%	3,4	0,062	100%	31,9	0,288	100%	35,3	0,351	72%
2007	66	24%	75%	4,2	0,101	95%	25,1	0,478	95%	29,3	0,579	72%
2008	50	44%	58%	2,6	0,062	94%	30,0	0,479	94%	32,6	0,541	66%
2009	87	50%	43%	1,2	0,028	100%	18,1	0,154	100%	19,3	0,182	51%
2010	9	44%	67%	1,2	0,027	89%	18,9	0,150	89%	20,1	0,178	56%
2011	37	32%	62%	1,9	0,049	86%	15,4	0,537	89%	17,3	0,587	59%
2012	52	21%	56%	3	0,070	98%	19,7	0,195	98%	22,8	0,266	58%
2007-2011	249	39%	59%	2,3	0,058	96%	21,9	0,362	96%	24,3	0,420	61%
2008-2012	235	40%	53%	2,0	0,048	96%	20,6	0,293	96%	22,6	0,341	57%

6.6.1.3.1Current situation

In the recent 5-year period (2008-2012), 96% of the 235 German Fulmars examined had plastic in their stomach (Table 6-10). On average 22.6 pieces of plastic with a mass of 0.34 g were found per bird. According to the age and sex analysis (dissection) 40 % of the birds were adult and 51% were male. 30 % of the birds belonged to the darker colourphase and probably originated from Arctic regions. Oil was recorded as cause of death for 3% of the examined birds. For all birds a body condition-index was calculated based on the status of the 3 parameters: breast muscle, subcutaneous fat and intestinal fat (for details see Van Franeker 2004). With an average condition-index of 1.1 most of the analyzed birds showed a very poor body condition.

For comparison of the different categories of plastic found in Fulmar stomachs two main groups are distinguished: industrial plastic (mainly prefabricated plastic pellets) and user plastic (summarization of various plastic items which had been in use before being discarded). In the current situation (2008-2012) 53% of the birds had industrial plastic in their stomach while user plastics were found in 96% of the birds. As the category user plastic contains a wide variety of plastic items, 5 subcategories are separated: sheet-like plastic, threads, foamed plastic, plastic fragments and other plastics. Regarding the frequency of occurrence in the composition of plastic litter in fulmar stomachs plastic fragments show the highest incidence with 91 %. These are followed by foamed plastics with 59% and sheet-like plastics with an incidence of 53 %. Looking at the average mass, plastic fragments make up almost half of all plastic found in Fulmar stomachs. Also, very prominent in the composition of stomach litter is foamed plastic (mainly polystyrene) which is underrepresented in the average mass per bird due to its low mass density but has with 5.9 items per bird the second highest incidence on average. In addition to the plastic waste found in Fulmar stomachs other rubbish of anthropogenic origin was recorded. Kitchen waste was found in 21 % of all birds with on average 1.4 items per bird. This represents the highest frequency of occurrence in this subcategory.

Table 6-10: Summary of sample characteristics and stomach contents of Northern Fulmars found in Germany for the current 5-year period 2008-2012. The top line shows sample composition in terms of age, sex, origin (by colourphase; darker phases are of distant Arctic origin), death cause oil and the average condition-index (which ranges from emaciated condition = 0 to very good condition = 9). The table lists for each litter (sub)category: Incidence, representing the proportion of birds with one or more items of the litter category present; average number of plastic items per bird stomach ± standard error; average mass of plastic ± standard error per bird stomach; and the maximum mass observed in a single stomach. The final column shows the geometric mean mass, which is calculated from ln-transformed values.

Year	nr of birds	adult	male	unsexed	LL colour	death oil	avg condition
2008-2012	235	40%	51%	3%	70%	3%	1,1

		incidence	average number of items (n/bird) ± se	average mass of litter (g/bird) ± se	max. mass recorded (g)	geometric mean mass (g/bird)
1	**ALL PLASTICS**	**96%**	**22,6 ± 2,281**	**0,341 ± 0,075**	**14,6**	**0,1007**
1.1	**INDUSTRIAL PLASTICS**	**53%**	**2,0 ± 0,264**	**0,048 ± 0,006**	**0,6**	**0,0072**
1.2	**USER PLASTIC**	**96%**	**20,6 ± 2,171**	**0,293 ± 0,075**	**14,6**	**0,0767**
1.2.1	sheets	53%	2,7 ± 0,571	0,031 ± 0,013	2,5	0,0021
1.2.2	threads	41%	1,1 ± 0,155	0,013 ± 0,003	0,5	0,0016
1.2.3	foamed	59%	5,9 ± 1,238	0,026 ± 0,006	0,9	0,0030
1.2.4	fragments	91%	10,5 ± 1,156	0,143 ± 0,039	9,0	0,0446
1.2.5	other plastics	18%	0,3 ± 0,074	0,081 ± 0,061	14,1	0,0010
2	**OTHER RUBBISH**	**29%**	**1,8 ± 0,425**	**0,102 ± 0,028**	**5,0**	**0,0023**
2.1	paper	5%	0,2 ± 0,103	0,002 ± 0,001	0,3	0,0001
2.2	kitchenwaste (food)	21%	1,4 ± 0,405	0,085 ± 0,027	5,0	0,0016
2.3	rubbish various	5%	0,1 ± 0,034	0,015 ± 0,007	1,3	0,0003
2.4	fishhook	0%	0,0 ± 0.000	0,000 ± 0.000	0,0	0,0000

6.6.1.3.2Mass of industrial and user plastic in Fulmar stomachs

Sufficient annual data on the composition of plastic litter in Fulmar stomachs in Germany exists from 2003 onwards. Since the early 2000s the average mass of user plastic per bird increased from 0.29 g (5-year average, 2003-2007) to 0.36 g (2007-2011), but dropped back to 0.29 g in the current period 2008-2012. The mass of industrial plastic stayed almost stable at around 0.05 g (Figure 6-19). Data from the Netherlands which go back to the early 1980s (Van Franeker & The SNS Fulmar Study Group 2011) show a strong decline of industrial plastic in the composition of stomach litter content until the end of the 1990s. Its proportion declined from 50 % to less than 20 % in the mass of plastic litter found in Fulmars. The German and the Dutch data on Fulmars' stomach contents show very similar patterns regarding the composition of plastic litter. Although, average mass of user plastic in German samples slightly exceeds Dutch levels in recent years.

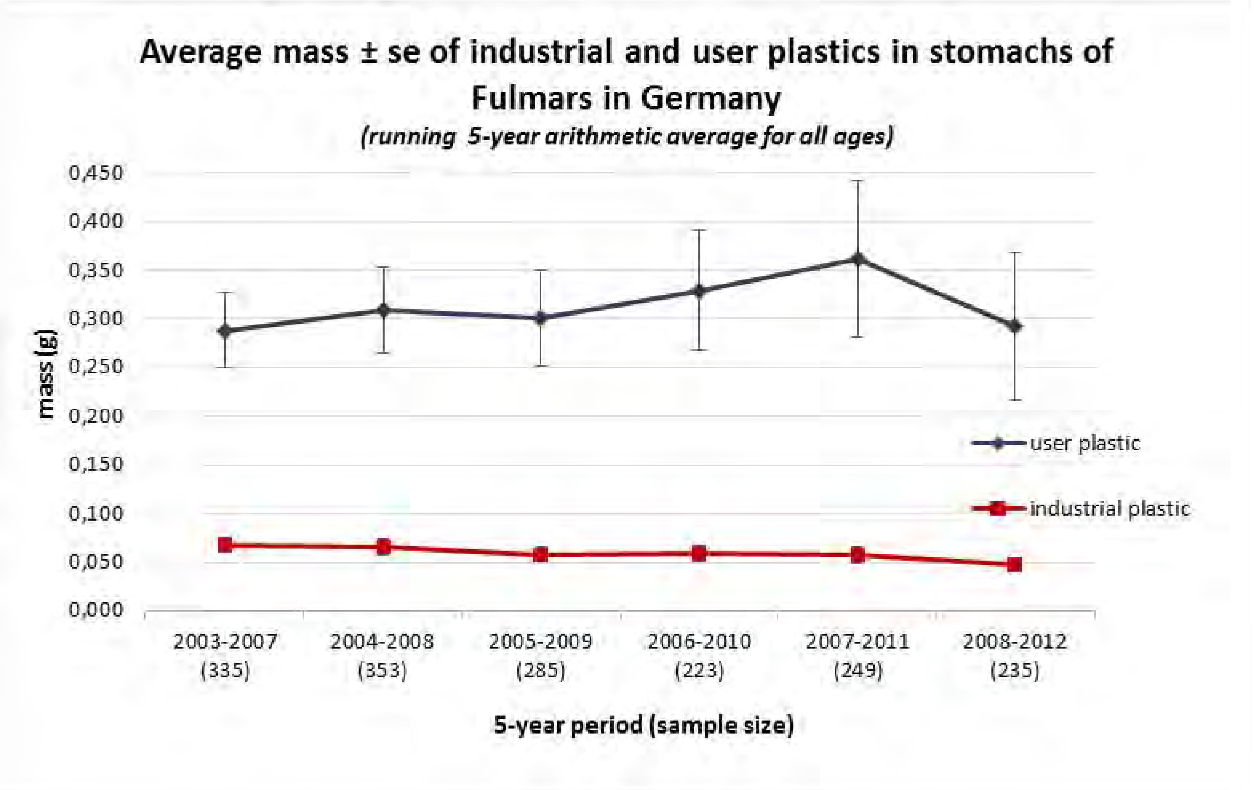


Figure 6-19: Trends of industrial and user plastics in Fulmars from Germany 2003 -2012. Trends are shown in 5-year averages for mass of plastic in stomachs of Fulmars beached in Germany (running average over 5-year periods, i.e. data points shift one year ahead at time; x-axis label show 5-year periods and in brackets the sample sizes).

6.6.1.3.3Mass of plastic in different age groups

As already demonstrated in the Netherlands by Van Franeker & Meijboom 2002, age is the only factor which significantly influences the amount of plastic found in stomachs of Northern Fulmars. Adults tend to have less plastic in their stomach than non-adults which may be explained by older birds being more experienced in deciding whether floating particles are appropriate food or plastic waste. Other factors such as body condition, sex or origin seemed to have no influence on the mass of plastic litter in beached Fulmars. In Figure 6-20 the geometric mean mass of plastic found in Fulmars beached in Germany since 2003 is shown separately for adults, immature and juvenile birds and both combined (including birds with unknown age). The geometric mean is chosen because it allows to include years were only a few birds were collected. The data is normalized by logarithmic transformation which reduces the influence of exceptionally high values. In most years with a sufficient sample size the amount of plastic in non-adult Fulmars is larger than in adults. The geometric mean data also confirms that the patterns for different age groups follow similar trends, allowing the use of pooled data over all age groups for the purpose of monitoring.

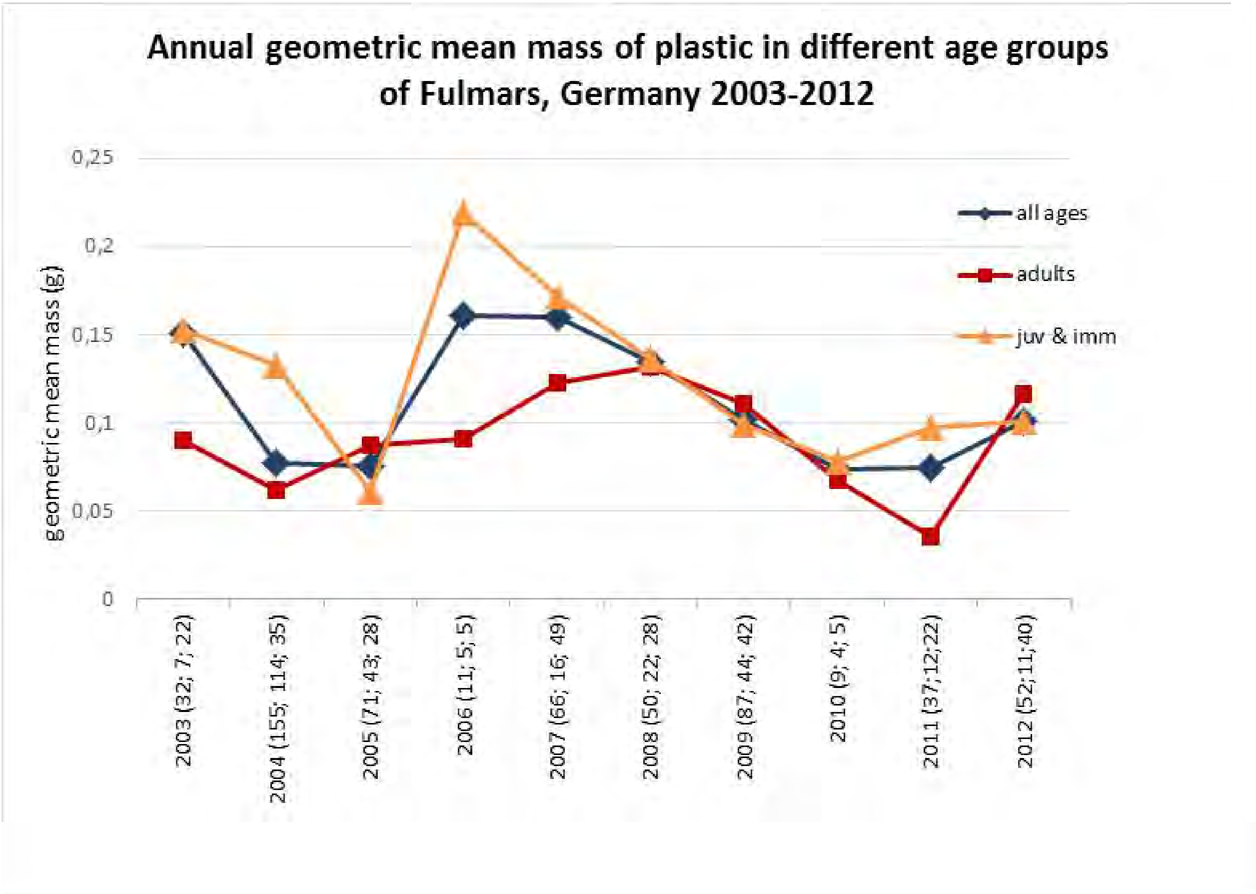


Figure 6-20: Annual geometric mean mass of plastics found in beached Fulmars from Germany 2003-2012 for all age groups combined (including birds with unknown age), adult birds and non-adults, with sample sizes in brackets in the x-axis labels.

6.6.1.4 Application of the Fulmar-Litter-EcoQO approach as Indicator 10.2.1 "Trends in the amount and composition of litter ingested by marine animals" for MSFD Descriptor 10 - Marine litter

The European Commission addressed the impact of litter on marine life with the MSFD Indicator 10.2.1 "Trends in the amount and composition of litter ingested by marine animals" (2010/477/EU). Though the indicator is based on assessing trends in ingested litter, the Commission Decision (2010/477/EU) thereby also aims to improve knowledge on the impacts of litter on marine life in general (MSFD GES Technical Subgroup on Marine Litter 2011). The MSFD GES Technical Subgroup on Marine Litter (2011) presented three monitoring tools for the monitoring of ingested litter. Of these only the Fulmar tool which describes the OSPAR Fulmar-Litter-EcoQO approach is classified as completely mature.

Fulmars qualify as monitoring species for the ingestion of plastic debris as they are abundant in the North Sea and the North Atlantic, feed exclusively offshore and are often found in beached bird surveys (Van Franeker & Meijboom 2002). Moreover, Fulmars ingest a large variety of marine litter items (Bourne 1976, Furness 1985, Moser & Lee 1992, Van Franeker et al. 2011, Van Franeker & The SNS Fulmar Study Group 2011). In addition, they have a sufficiently high incidence of ingested litter which makes them suitable for monitoring change even in times or areas of lower pollution (MSFD GES Technical Subgroup on Marine Litter 2011). The observed high incidence of ingested litter unavoidably has mechanical and chemical consequences that affect body condition of birds with negative consequences for individual survival and the capacity to reproduce (Galgani et al 2010).

Data on plastic ingestion by Northern Fulmars is available since the early 1980s for the Dutch coast and since 2002 for most other countries bordering the North Sea, when a monitoring programme started as part of the EU campaign “Save the North Sea” (Guse et al. 2005, Van Franeker et al. 2005). Thus, the most comprehensive data set available in the EU focusing on the trends of ingested marine litter or on its impacts is that on Northern Fulmars. The applied methodology was gradually evaluated and finally implemented by the OSPAR Commission as the Fulmar-Litter-EcoQO (OSPAR 2010a, b, Van Franeker et al. 2011). The methodology has been developed for the North Sea but it is applicable to most of the North East Atlantic and has recently been adapted for the North East Pacific, too (Avery-Gomm et al. 2012). It can be used to assess temporal and regional trends for different litter categories and compliance with set ecological quality targets (Van Franeker et al. 2011). It is closest to the Good Environmental Status (GES) approach in the MSFD as it includes a target value (‘ecological quality objective’) considered to represent an acceptable level of litter in the marine environment (OSPAR 2008). The OSPAR Fulmar-Litter-EcoQO, where less than 10% of fulmars exceed a critical level of 0.1 g of plastic in the stomach (OSPAR 2008), refers to a litter situation in the Canadian Arctic, where the pollution level is considered to be acceptable in terms of environmental quality (Van Franeker et al. 2011). The long term OSPAR EcoQO target for the North Sea can thus be seen as a realistic target / GES level for the longer term. In consequence, the OSPAR Fulmar-Litter-EcoQO approach for the North Sea has recently been adopted as an indicator for GES in the European MSFD (EC 2008, 2010, Galgani et al. 2010). For the shorter term 2020 MSFD GES target however, an intermediate definition might be a more realistic option (e.g. a 'statistically significant reduction' according to the established method for testing of trends in the OSPAR EcoQO).

Overall, the OSPAR Fulmar Litter approach currently is the only fully applicable methodology for Indicator 10.2.1 in the MSFD (MSFD GES Technical Subgroup on Marine Litter 2011). Quality of methodology is guaranteed by applying an agreed OSPAR methodology that has been developed over a number of years and was confirmed by scientific peer review (see Van Franeker et al. 2011, Avery-Gomm et al. 2012). Beyond the scientific approval the Northern Fulmar is one of the most powerful public relation tools with regard to creating awareness of the consequences of marine litter. The Fulmar is a symbol for the negative effects of littering on the marine environment. During the last few years the work related to the Fulmar Litter EcoQO has been featured in countless TV documentaries, newspaper articles, radio shows and internet feeds both nationally and internationally.

6.6.1.5 Additional activities

Together with the German Environment Agency (UBA) the members of workpackage 5 organized a workshop on monitoring marine litter which took place on the 9 and 10 October 2013 in the UBA headquarters in Dessau. 35 scientists and members of NGOs whose work relates to marine litter participated. A member of the FTZ gave a talk about the Fulmar litter project in the opening plenum and moderated the discussions in the sub working group on litter ingested by biota.

The outcome of the workshop supplemented a guidance paper with monitoring concepts for marine litter in the North and Baltic Seas which was written by the members of workpackage 5. This monitoring guidance is planned to be published soon by the German Environment Agency.

On request of the German Environment Agency (UBA) scientific advice was given to the authors of the MSFD GES TSG Marine Litter – Thematic report: Monitoring Protocols-Litter in Biota, Entanglement. The FTZ compiled information on existing protocols on entanglement of seabirds in marine litter both in colonies (litter as nesting material) and at sea as documented by beached bird surveys (BBS). Based on the available information from Europe and North America it was outlined which points need to be considered if a standard type of protocol on entanglement of seabirds in marine litter should be developed as a monitoring tool for European waters. The strengths and drawbacks of the existing protocols were highlighted.

In addition, several press and public relation activities were undertaken.

6.6.1.6 Acknowledgements

We would like to thank the German Environment Agency (UBA) for funding the German Fulmar Litter research within the current project.

We are grateful for the good cooperation with Jan van Franeker and his colleagues at the IMARES institute that characterized our involvement since 2002.

We would like to thank all agencies, institutions and NGOs that supported our work throughout the years: Nationalparkverwaltung im Landesbetrieb für Küstenschutz, Nationalpark und Meeresschutz Schleswig-Holstein (LKN-SH), Schutzstation Wattenmeer e.V., Der Mellumrat e.V., Niedersächsischer Landesbetrieb für Wasserwirtschaft, Küsten- und Naturschutz (NLWKN), Landesmuseum Natur und Mensch in Oldenburg, Öömrang Ferian e.V., Seehundstation Nationalpark-Haus Norden-Norddeich, Naturschutzbund Deutschland e.V., Verein Jordsand e.V., Naturschutzgemeinschaft Sylt e.V..

We are particularly grateful to the hundreds of individuals that have helped in the collection of beached Fulmars along the German North Sea coast since 2002 on a voluntary basis. Without their motivation and support the realisation and continuation of the Fulmar Litter monitoring programme would not be possible.

6.7 Trends in macroscopic litter on the seafloor and 'ghost nets'

6.7.1 Introduction

Within the framework of the implementation of the Marine Strategy Framework Directive (MSRL), an ecosystem-based evaluation of the current state of the marine environment in the German Exclusive Economic Zone (German EEZ) of the North Sea and the Baltic Sea is to be conducted. Thereby, the aim is to define a 'good state' which is to be achieved in the future.

In the sub-project "Trends in macroscopic litter on the seafloor and 'ghost nets' " which is integrated into the above evaluation, litter present on the seafloor and valued as an anthropogenic stressor was used as an index for the environmental status. In addition, the spatial distribution and abundance of fisheries nets trapped and entangled in ship wrecks was recorded in order to investigate the harmful effects of the potential stressor 'ghost net'.

In all the world's oceans, inorganic litter is present and increasing in abundance (Barnes et al., 1909; UNEO, 2009). In part, the litter is irregularly distributed. Locally, however, it is massively concentrated at the water surfaces as well as at the coasts and the seafloor (Galgany at al., 1995; Gregory, 1999; Moore et al., 2001; Ramirez-Llodra et al., 2011; Bergmann and Klages, 2012). Marine pollution, particularly through plastic litter, poses serious threats and harmful impacts on organisms living in and above the sea and on humans consuming food from the sea (Van Franeker et al., 2011; Rochman et al., 2013). Quantifying the amounts of litter in the sea is a prerequisite for better estimating the danger that litter poses and for developing, designing and verifying strategies and target settings for the reduction and ultimately for the elimination of litter. In the German Bight, the amounts of litter washed up to the shores of the East and North Frisian Barrier-Islands are regularly recorded (OSPAR commission, 2007) and are also described for the island of Heligoland (Vauk and Schrey, 1987). The amount and spatial distribution of plastic litter drifting at the water surface within the EEZ has been described as well (Thiel et al., 2011).

So far, the amount and spatial distribution of litter on the seafloor of the EEZ has been estimated only roughly through few bottom trawls (Galgani et al., 2000). The Alfred Wegener Institute Helmholtz-Centre for Polar and Marine Research has regularly conducted beam trawl and otter trawl catches in the German EEZ and in the Wadden Sea of the North Sea between 1998 and 2010 in order to monitor the development of the benthic faunal communities over a longer spatial period. In the reports of these scientific fishery hauls, the amounts of litter caught in the nets have been recorded as well. In the present study these data were used to:

- estimate the amount and spatial distribution of litter on the seafloor in the German Bight and to
- describe the development of the amounts of litter over time.

According to the results,

- areas were selected and methods recommended for a long-term monitoring on the development of the amounts of litter on the seafloor and
- a suggestion was made on the degree of pollution and the quality of the litter that should represent an achieved 'good state' of the seafloor *sensu* MSRL.

The results were provided to compare the amounts of litter at the seafloor with the amounts of litter at the shores of different Barrier-Islands.

In addition, also ’ghost nets‘ pose a worldwide threat to the marine fauna. Crabs, fish, sharks, turtles, mammals and birds perish in large numbers through given up or lost and freely floating fishery gear made of filamentous material and which remains to be effective even after decades (Richards, 1994; Gilardi et al., 2010; Good et al., 2010; Kasperek and Predki, 2012). For the German Bight, quantitative data on the types, abundances and spatial distribution of such abandoned fisheries nets and information on their direct and indirect harmful effects (through trapping and killing of animals and as a source of soluble chemical substances) are lacking so far. From the Baltic Sea it is, however, known that considerable amounts of ‘ghost nets’ can be trapped and entangled in ship wrecks (Kasperek and Predki, 2012) and from faunal surveys on wrecks (Krone und Dederer, pers. observation) this has also been shown for the North Sea. More than 1.000 wrecks are scattered over the German EEZ (Krone und Schröder, 2012). It can be assumed that a large amount of freely drifting nets is caught by these obstacles. The amount of wrecks, therefore, may reflect an estimation of the amount of all ‘ghost nets’ present in this sea area.

The Federal Maritime and Hydrographic Agency (BSH) conducts inspection dives at the ship wrecks in the German EEZ documenting their status in order to warrant safety along the major shipping routes in the German Bight. The video recordings of the inspection dives provide the possibility to roughly estimate the amount of ‘ghost nets’ for a large number of wrecks without the employment of own scientific divers. In the present study, the inspection video recordings were used to:

1. estimate the percentage coverage of wrecks through nets in the German EEZ and to determine the types of nets caught in the wrecks,
2. depict the spatial distribution of the ‘wreck ghost nets’ in the German EEZ,
3. estimate the total surface area of ‘ghost nets’ on ship wrecks in the German Bight and
4. identify selected wrecks at which harmful impacts directly effected through the ‘ghost nets’ could be investigated in the future.

6.7.2 Materials and methods

6.7.2.1 “Ghost net” survey

6.7.2.1.1 The ship wrecks

The assessment of the amount of ‘ghost nets‘ which were trapped and entangled at ship wrecks was evaluated by inspecting video recordings of the wrecks which had been taped earlier by the BSH. In total, video inspections of 64 wrecks with known position data (North Sea, German Bight 59 and Baltic Sea 5) were evaluated. The descriptions of the wrecks were taken from the reports on wreck-search surveys of the BSH. For 51 wrecks information of surface (length x width) is available and for 25 wrecks the date of the actual sinking is known. The wrecks considered in the present study relate to ships sunken between 1883 and 2008. The duration of exposure of the wrecks at the time being sampled (i.e. inspection video recordings) is on average 51±32 years. The evaluated inspection videos were recorded between 2004 and 2011. The average wreck surface is 1,026±1,506 m^2 and the average height of the wrecks above the surrounding seafloor amounts to 5.5±3.2 m. The wrecks are located in water depths between 7 and 40 m (-LAT) (compare Figure 6-22).

6.7.2.1.2 Analysis

For each wreck an individual inspection recording of 13.5±5.0 min per video was evaluated. The divers recording the video would swim over along the wreck maintaining a distance of approximately 1 m. A digital camera was fixed to the diver’s mask. On the video recordings, the visible image area and, thus, the visible surface area of the wreck could be of different size during each dive. On average, the visible image area of a wreck surface was approximately 1 m². The evaluation of the recording started as soon as a diver would start passing over along the wreck. While observing the dive, the estimated percentage coverage of a wreck through nets was recorded every 30 seconds (video step or unit). At the same time the quality of the nets reflecting a gradient of different mesh sizes and netting twines (hereafter referred to as 'fraction') was documented for each video step. Hereby, eight fractions of mesh sizes or types were classified (Table 6-11). In addition, the positions of the nets were identified per video step. The positions of the nets were distinguished into nets attached entirely to the wreck (attached nets), nets attached loosely to the wreck and thereby partly floating up to 1-2 m above the wreck (floating nets) and nets partly attached to the wreck but floating several meters above it (pelagic nets). In order to avoid a bias, all evaluations were conducted by the same observer.

Table 6-11: Qualitative classification of net fractions identified during the evaluation of wreck inspection video recordings of the BSH in the 'ghost-nets' at ship wrecks in the German Bight.

Code	Name	Description
F	fine	A net made of thin netting twine (ø 1-2 mm), with differing mesh widths
Fs	fine, small meshed	A net made of thin netting twine with small mesh widths (2-4 cm)
Fc	fine, coarse meshed	A net made of thin netting twine with large mesh widths (4-8 cm)
Fm	fine, monofilament	An net consisting of monofilaments ('gillnet-like')
Cm	coarse, mixed	A net made of coarse netting twine (ø 2-5 mm), with differing mesh widths
Cs	coarse, small meshed	A net made of coarse netting twine, with small mesh widths
Cc	coarse, coarse meshed	A net made of coarse netting twine, with coarse mesh widths
P	Protection Nets	Very coarse, mostly orange coloured nets made of strong material with coarse meshes (large mesh widths) (protective layers often fixed underneath bottom trawls and beam trawls)

In consideration of all video steps per wreck the percentage coverage of each respective wreck through nets was determined. For those wrecks for which the wreck surface (projected as basic area onto the seafloor) was known, the absolute surface covered with nets was calculated and given as average net surface per wreck. The spatial distribution of the investigated wrecks and the percentage coverage through nets were depicted using the GIS-programme ArcMap 10.1®. A linear regression analysis was applied in order to test, whether the percentage coverage of the wrecks through nets was correlated with the age of the wrecks, the exposition time (duration from the year of the sinking until the recording of the inspection videos), the size or height of the wrecks (programme GraphPad Prism® 5.0), type of nets and under consideration of technical shipping safety aspects, wrecks were identified where scientific diving investigations on the catchability of 'ghost nets' should preferably be conducted.

6.7.2.2 Seafloor litter survey

6.7.2.2.1 Data ascertainment

For the assessment of the amounts of litter on the seafloor in the North Sea, reports from 1642 fishery hauls by the Alfred Wegener Institute (AWI) were sighted. So far, these data have been available in written protocols only. In total, 534 otter trawls (OT, time period 2000 – 2010, Table 6-12) and 449 beam trawls (BT, time period 2000 – 2010, Table 6-12) were undoubtedly assigned to geographic positions in the open North Sea and evaluated (Figure 6-21). They relate to data from fisheries catches of stationary long-term series as well as to catch data from temporarily restricted and spatially dispersed investigations by the AWI. Therefore, the distribution of the catches across the North Sea is heterogenic and variably dense. In addition to the planned recordings, the amounts of litter from 503 BTs which were conducted for evaluation purposes (by the fishing vessel Polaris, "Polarisprotokoll") focussing on the impacts of the EUROGAS-pipeline on the benthos in the Wadden Sea off Langeoog between 1998 and 2007, were recorded as well. Information on the abundances of litter was drawn from the present catch reports and positioned on the basis of station protocols and the data base of the AWI for the benthic communities in the North Sea (principal manager Dr. J. Dannheim). For the analysis, these data were then converted into the unit g catch-1 m-2 (with reference to the starting point of the respective hauls). 29 fishery hauls (17 BT, 12 OT from the year 2007) were additionally recorded along with the protocols of the 100m-OSPAR beach litter monitoring (categories of litter), in order to assess whether a differentiated ascertainment of litter is more appropriate than following evaluation according to the protocols of the 'FS Heincke' for offshore areas. The OSPAR protocols also serve to gain more indices on the percentage composition of the inorganic litter.

Table 6-12: List of used fishery gear (BT = beam trawl, OT = otter trawls) and parameters (SD = standard deviation).

Area	Period	Fishing gear	Mesh width [mm]	opening width [m]	Speed ground [kt]	Average sampling time on ground [min ±SD]	Samples n
Offshore	2000 - 2010	BT	10	3	3-4	9±20	449
Offshore	2000 - 2010	OT	10	15	3-4	21±22	534
Wadden Sea	1998 - 2007	BT	10	3	2-3	10±3	503

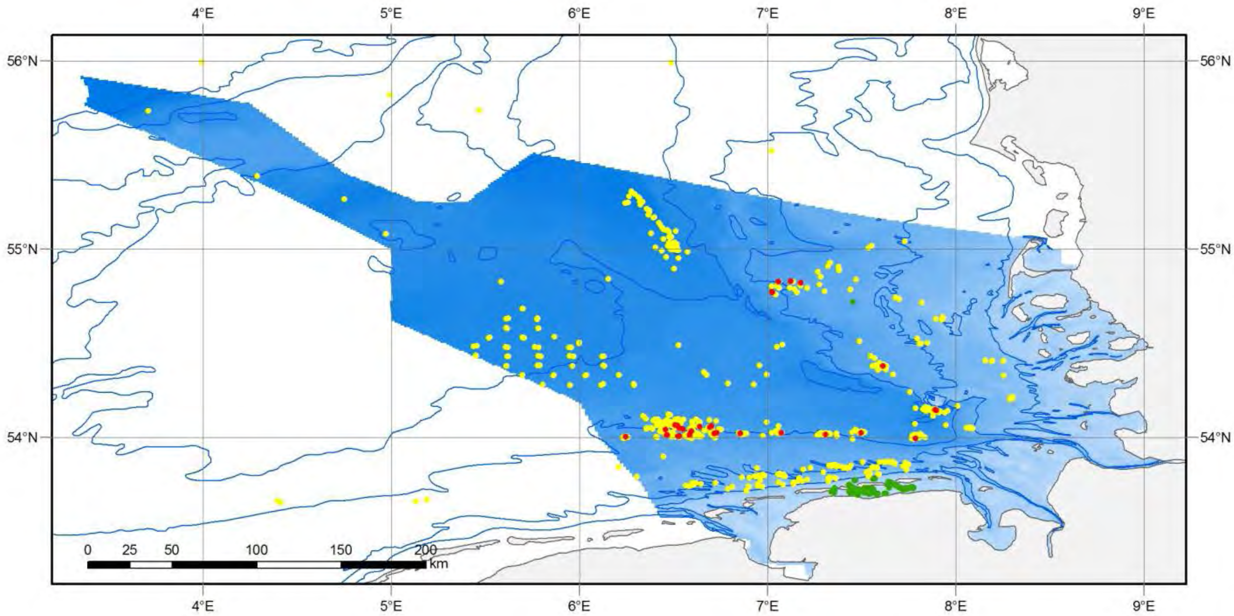


Figure 6-21: Distribution pattern of otter and beam trawl data (yellow) by the AWI inside the open North Sea of the German EEZ and coastal waters (sampled by the research vessel '"FS Heincke') and beam trawl catches by the AWI inside the Wadden Sea (green points, sampled by the 'FS Polaris'). Each point marks the middle of a trawl length. 29 selected offshore litter-catch-data were entered into 100 m OSPAR beach litter monitoring protocols (red points).

6.7.2.2.2Analysis

Along with faunal observations in the Heincke and Polaris protocols, 13 different types of litter and their respective wet weights were recorded. Throughout the monitoring, the same methodological protocols were used per ship. It must, however, be assumed that the litter was differently recorded on the various expeditions, because the original purpose of its recording lay predominantly in the aim to distinguish its share in weight from that of the monitored faunal species. The differentiated recording of litter types was not the aim of the investigations and other than in the case of the faunal monitoring, a consistent procedure of record entries in the catch protocols could not be shown. Therefore, the present analyses were possible only for the total abundances and the spatial distribution of the inorganic litter and not for individual categories (fractions) of litter.

The inorganic litter category (plastic litter) comprises the following fractions:

Inorganic remains: rubber, net, net yarn, rope, styrofoam, inorganic litter, plastics, clothing, buoy, bottles, knife handles, plastic bags, rope, indifferent.

Aluminium, wire, glass bottles, cans, glass, hemp rope iron and metal are not counted to the category of inorganic litter. These types of litter were identified as single findings on few occasions only (in less than 1.08% of all catches) but took up large shares in weight, which would have significantly skewed the results. Because metal and glass litter are considered clearly less harmful for the marine environment than plastic material, the exclusion of the former litter types was considered irrelevant in the present investigation. The entire data set comprising beam trawls and otter trawls was plotted in geographical reference using the GIS programme ArcGIS 10.1®. The investigation area was subdivided into spatial and, where possible, into temporal clusters (separately for beam trawls and otter trawls). A cluster comprises at least ten samples. Similarly, single time periods for comparison in one cluster consisted of a minimum of ten samples and of durations of at least three years. This minimum time period was chosen in order to incorporate possible outliers causing high variability due to spontaneous findings - for example after storms. The amounts of litter in the respective clusters and time periods were calculated using the Kriging method (programme ArcGIS 10.1®) for a visual comparison (Kriging-Type: simple; Parameters: Output type: prediction; Searching neighborhood: standart; Neighbors to include: 5; Include at least: 2; Sector Type: 8 sectors; Variogram: Semivariogram; Number of lags: 12; Model type: K-Bessel; Anisotropy: no; Lag size, Partial sill, Range, Parameter: manually adjusted in each cluster). Because of the very heterogenic spatial distribution of the samples in all clusters and because of the patchy distribution of the litter (high amounts of zero-findings), we refrained from applying parametric and non-parametric statistical tests for the comparison of data sets (here: the litter samples per cluster). Kriging is a geo-statistical interpolation method which calculates specific weighted values for locations from surrounding samples under consideration of spatial variability65. Within this interpolation, the result is projected as a rectangular area, where the limits are defined by its most eastern, western, northern and southern positions. From the interpolation results a result grid was determined with a pixel size of 0.005° (which relates to approximately 330 x 555 m in the investigation area). The grid creation bears the advantage that large scale irregularities from the interpolation are compensated through the coarse grids. In addition, the grid can be trimmed with the help of a mask. This is reasonable in cases, where the regular distribution of the measured data points is restricted to a certain section of the rectangular interpolation area. Creating coarse grid pixel which partly exceed the interpolated data, the limited measurement points in the grid do not exactly align with its border any more. In part, some clusters were trimmed with a mask to a consistent size (clip by mask) for the multi-temporal comparison. These masks were created by hand and the position of the measuring points in the respective time units is referenced to the largest common area. In this way, some points were included into the interpolation which really lies outside the trimmed grid. The total litter values were calculated from the means (± standard deviation) of the interpolation results using Kriging. Therefore, the size of the respective cluster was calculated in m² and multiplied with the mean value. Using Kriging under ArcGIS 10.1® it is not possible in the interpolation to consider obstacles (e.g. island groups) as barriers. The areas are part of the interpolation result although they were cut out of the results (clip function) after the interpolation and grid creation, in order to calculate the values for the total amount of litter from the grids. For clusters, in which beam trawl and otter trawl catches were conducted at the same time and in overlapping areas, the results of the two methods were compared with each other. For the ongoing monitoring of litter on the seafloor within the MSRL, clusters are recommended in which high amounts of litter were detected. Hereby, such clusters were selected for which sufficient data are available from the past years and which allow continuing a time series in monitoring the amounts of litter. Under consideration of the current deficits in collecting litter with scientific fisheries hauls and the presently used protocols, a sampling design is proposed.

[65] The variance is calculated from a semivariogram from which the parameters can be modified in a way that the estimated residual variance is as small as possible. Using Kriging, it is assumed that the spatial variation of the weighted values is statistically homogeneous over the entire surface, meaning that it is possible for each value to appear at any location.

6.7.3 Results and discussion

6.7.3.1 "Ghost nets"

The analysis of the wreck data has been completed. Data ascertainment and analyses have been completed successfully. Although the inspection videos were originally not designed to assess net coverages on wrecks, we succeeded in generating a quantitative description of the status quo of the 'ghost nets' and in identifying wrecks for future investigations on harmful effects. Most of the investigated wrecks consisted of expanses of debris from steel boat hulls which were broken apart. Only sporadically, well preserved boat hulls of smaller vessels were recorded (for example a sailing yacht that sunk a few years back). On most of the ship wrecks a large number of mobile demersal megafauna was observed.

6.7.3.1.1Amounts of nets and their spatial distribution

The investigated wrecks are irregularly spread across the southern German Bight (Figure 6-22) and the Baltic Sea (Figure 6-23). Especially along frequented shipping routes in the southern German Bight, approximately at the 30 m depth contour, video recordings are relatively numerous because notably high numbers of wrecks exist in these areas which are preferably monitored by the BSH for reasons of security in shipping traffic.

The average coverage of the wreck surfaces from all 64 wrecks with nets amounts to 4.7±7.2%. Most of the wrecks with increased coverage lie in the most southern part of the investigation area, a few nautical miles off the East Frisians. This is most likely due to the local and spatially highly concentrated inshore fisheries on site. Other tendencies in the spatial distribution were not detected. From the central parts of the German Bight no inspection videos are available that could allow for a comparison. The largest share is made up by nets made of coarse netting twine with different mesh widths followed by the protective nets and the fine nets made of monofilamentous twine (Figure 6-24).

The by far largest shares of the observed nets (99.2±3.0%) refer to those lying directly on the surfaces of the wrecks. These nets are attached directly and fixed tightly to the wrecks. Most likely they have been entangled and strongly woven into protruding wreck debris and dominating fragments through constant circulating tidal currents and wave dynamics. Only a very small share of nets was observed that floated 1-2 m (0.5±2.4%) or further (0.3±1.4%, at three wrecks) up above the wrecks and moved freely in the water column though attached to the wreck. Freely floating delicate nets (for example those made of monofilamentous material), which had partly sunken to the ground due to the weight of trapped animals such as those reported from other marine areas, have not been observed at the presently investigated wrecks. Large drift nets are not in use in the southern German Bight. It is, however, thinkable that such nets have followed incoming drifts from other sea areas into the open German Bight and can therefore be found at wrecks in the central German Bight. Benthic organisms such as the brown crab (Cancer pagurus) or the velvet crab (Necora puber) have repeatedly been observed on coarse nets tightly fixed to the wrecks. Apparently, the animals use the net structure without harm as habitat (Figure 6-25). So far, no organisms have been observed that were in any way strangled or trapped in coarse or protective nets. The inspection videos, however, did not allow for any systematic investigations on harmful effects beyond the presently recorded net qualities and quantities. Most of the tangly knotted and muddled nets made of finer twines could not be spontaneously controlled for trapped organisms. In addition, organisms that get trapped in nets mostly likely fall prey to predators and are thereby removed from the nets. This could also add to an underestimation of the catchability of the nets.

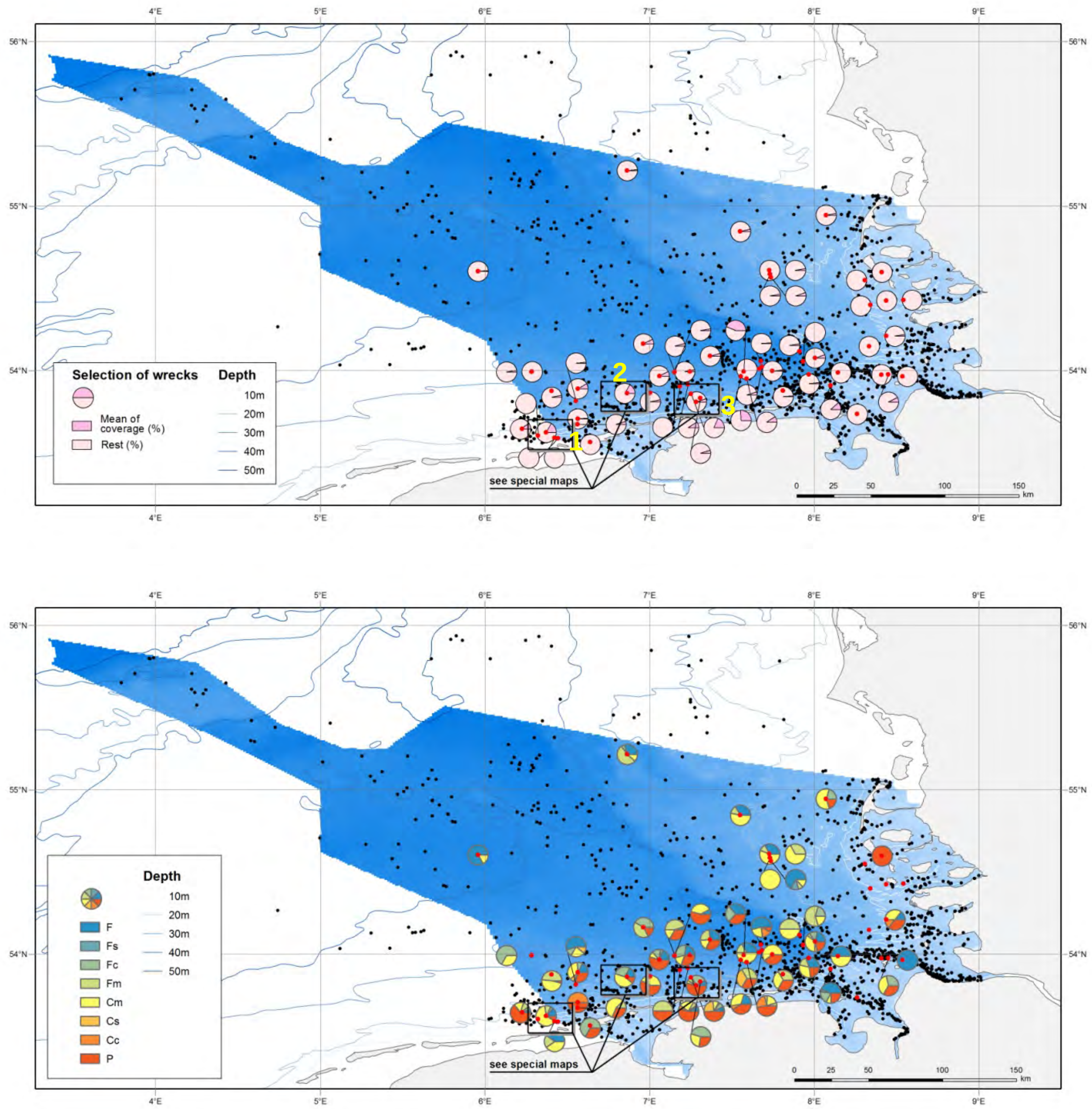
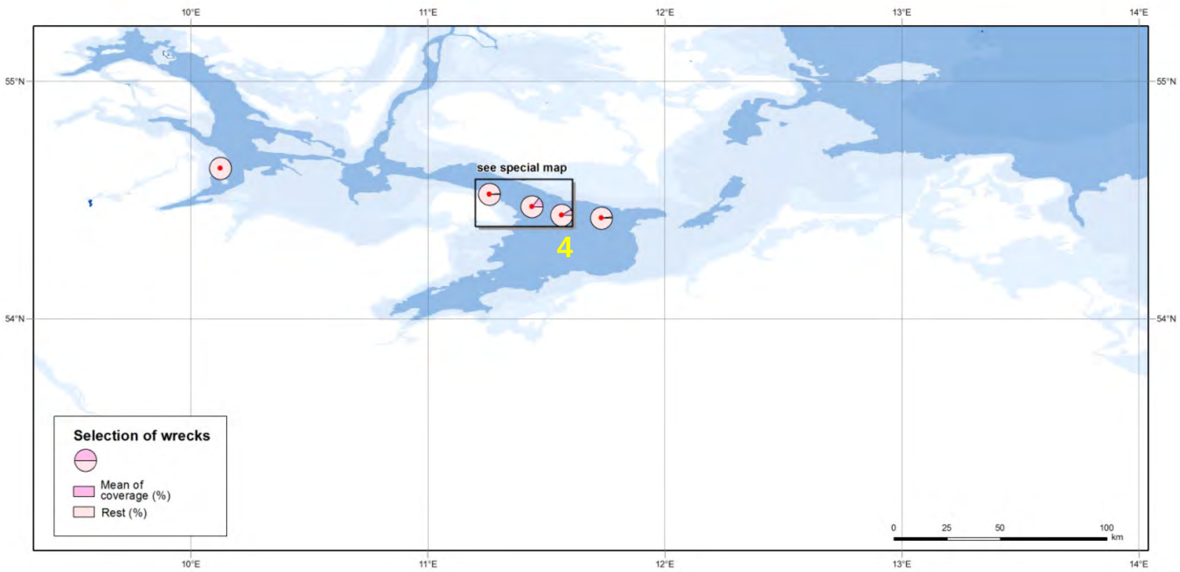


Figure 6-22: Spatial distribution of aprox.1,300 catalogued wrecks (BSH) (black dots) in the North Sea and the percentage coverage through nets (upper figure) and the percentage share of net fractions (compare table 1) on ship wrecks investigated by divers (red dots). Frame = position of wrecks recommended for further investigations.



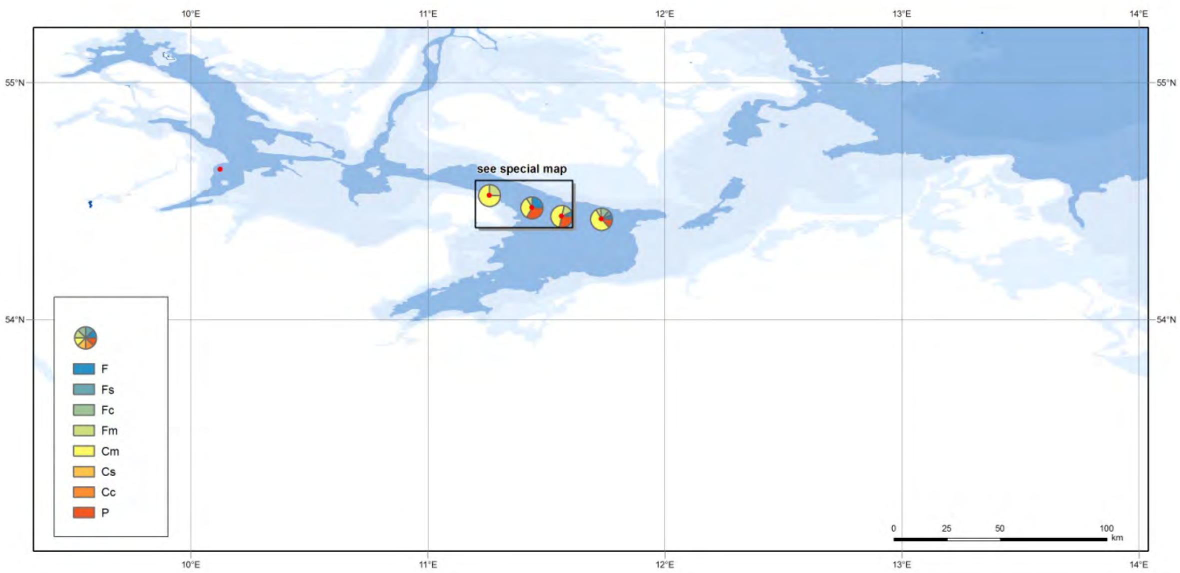


Figure 6-23: The percentage coverage through nets (upper figure) and the percentage share of net fractions (compare table 1) on ship wrecks in the Baltic Sea investigated by divers (red dots). Frame = position of wrecks recommended for further investigations.

The degree of the coverage of the wrecks neither increases with the age of the wreck nor with the duration of exposition at the time of the inspection (Figure 6-26). An increase in the abundance of 'ghost nets' in the German Bight within the past 100 years is therefore not evident by simultaneously considering all available wreck videos. The results did not indicate any specific trend in the abundances of the 'ghost nets'. It must, however, be taken into consideration that only a limited amount of wrecks was included into the analysis, because for 39 wrecks information on the exact sinking date was not available. In order to observe local trends in the 'ghost net' abundances, selected wrecks should be sampled repeatedly in specific time intervals.

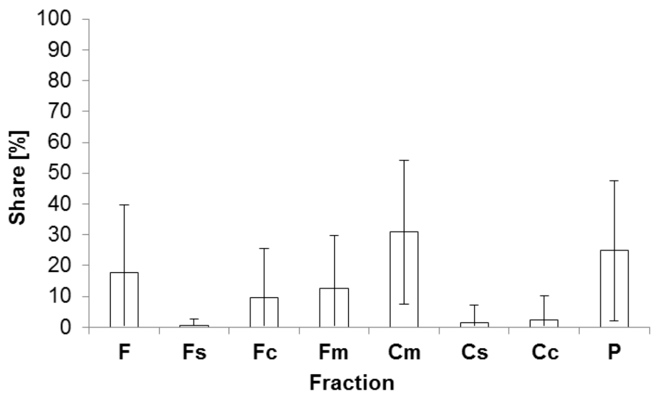


Figure 6-24: Average shares (±SD) of different net fractions in the coverage of the investigated wrecks.

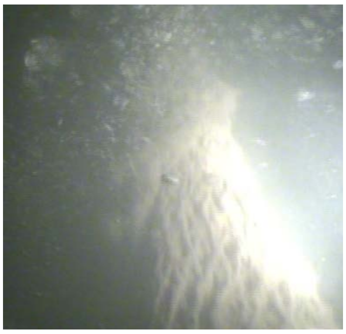
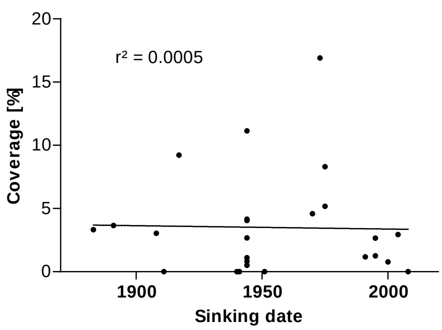


Figure 6-25: A strong coarse 'ghost net' drifting several meters above a wreck (left side: screen shot, C. Schneider) and a coarse net tightly fixed to a ship wreck, colonized by a brown crab *Cancer pagurus* (right side: R. Krone).



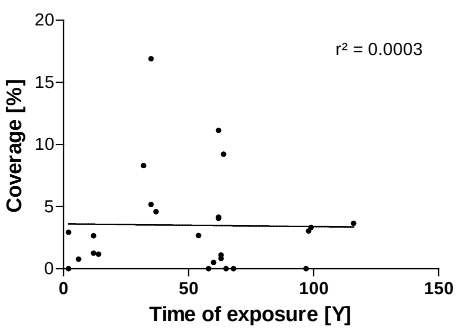


Figure 6-26: Linear regression analyses between the coverage through nets and the age of wrecks (n = 25) (left) as well as the duration from time of ship wrecking until time of inspection (video recording), respectively (n = 25) (right).

Similarly, as above, the size of the wrecks as well as their areal extent and height also did not have an effect on the degree of coverage through nets ($r^2 = 0.01$ and 0.03, respectively). In addition, neither did the age of the wrecks nor the spatial expansion or the height of the wreck (or obstacle, respectively) correlate with the percentage share of the individual net fractions (F, Fm, P) ($r^2 < 0.03$). Finer, lighter nets are also less often trapped and entangled by wrecks protruding higher up from the seafloor. This could indicate that there are only very few nets in the investigation area floating freely in the water column. Although larger wrecks represent larger obstacles than smaller wrecks, they are detected much better by echolocation and thus can be more easily avoided by trawlers. In this way the risk of potentially loosing fishing gear and the probability of a net being caught in a wreck can be reduced. A single, large and freely drifting net could probably cover up most parts of a small wreck, whereas large wrecks represent large obstacles that are potentially more likely to be encountered. It is likely that more nets are trapped and entangled in larger wrecks more often than in smaller wrecks. In the present investigation the degree of coverage was identified for each wreck, but not the actual number of nets covering a respective wreck. The average surface of a wreck covered with nets was 37.4±65.5 m². Transferred and extrapolated to the number of about 1.300 wrecks in the southern North Sea this amounts to 48.600 m² of stationary 'ghost nets' in the German Bight. This surface area of 'ghost nets', however, must be viewed as the minimum amount because the percentage coverage of the wrecks only applies to the basic area of the wreck and not to its actual and typically very rough surface. Information on harmful effects of the net surfaces and their different qualities on individual faunal organisms are not available so far.

6.7.3.1.2Recommendations for 'ghost net' surveys

In order to name and quantify harmful effects of 'ghost nets' trapped at ship wrecks, selected wrecks should be subjected to systematic investigations. Hereby, such wrecks should be selected that lie in water depths that allow for sufficient bottom time for the diving staff to investigate the nets in detail. In addition, wrecks with high percentage coverage of nets consisting of qualitatively different net fractions should be selected. Preferably, wrecks with nets directly attached to it should be selected because this type of net location was observed most frequently so far, although floating 'ghost nets' should be included as well. Although the latter type of 'ghost nets' is comparably rare, its harmful effects - i.e. its catchability per net - is potentially many times higher than that of the locally stable and directly attached nets and is therefore highly important for the holistic evaluation of the relevance of the litter type 'ghost net' in the German Bight. Species trapped and entangled in the respective net types should be quantified. According to the above criteria, five wrecks (4 North Sea, 1 Baltic Sea) were identified and are suggested for further investigations (Figure 6-27). The wrecks with the identification numbers (WKN) 930, 981 and 1149 reveal high degrees of coverage through directly attached nets (17, 16 and 17%) and are located in water depths of 21, 24 and 14 m (measured inside the scores). The wrecks with the WKN 1027 (depth 30.7 m) and 1543 (depth 27.6 m) revealed the highest shares of freely floating nets (5 and 9%). At these wrecks, the number of nets, their different quality (fractions) and their covering surfaces should be recorded. Subsequently and in order to monitor the temporal development of the net abundances, a regular ascertainment of net data at wrecks is highly recommended.

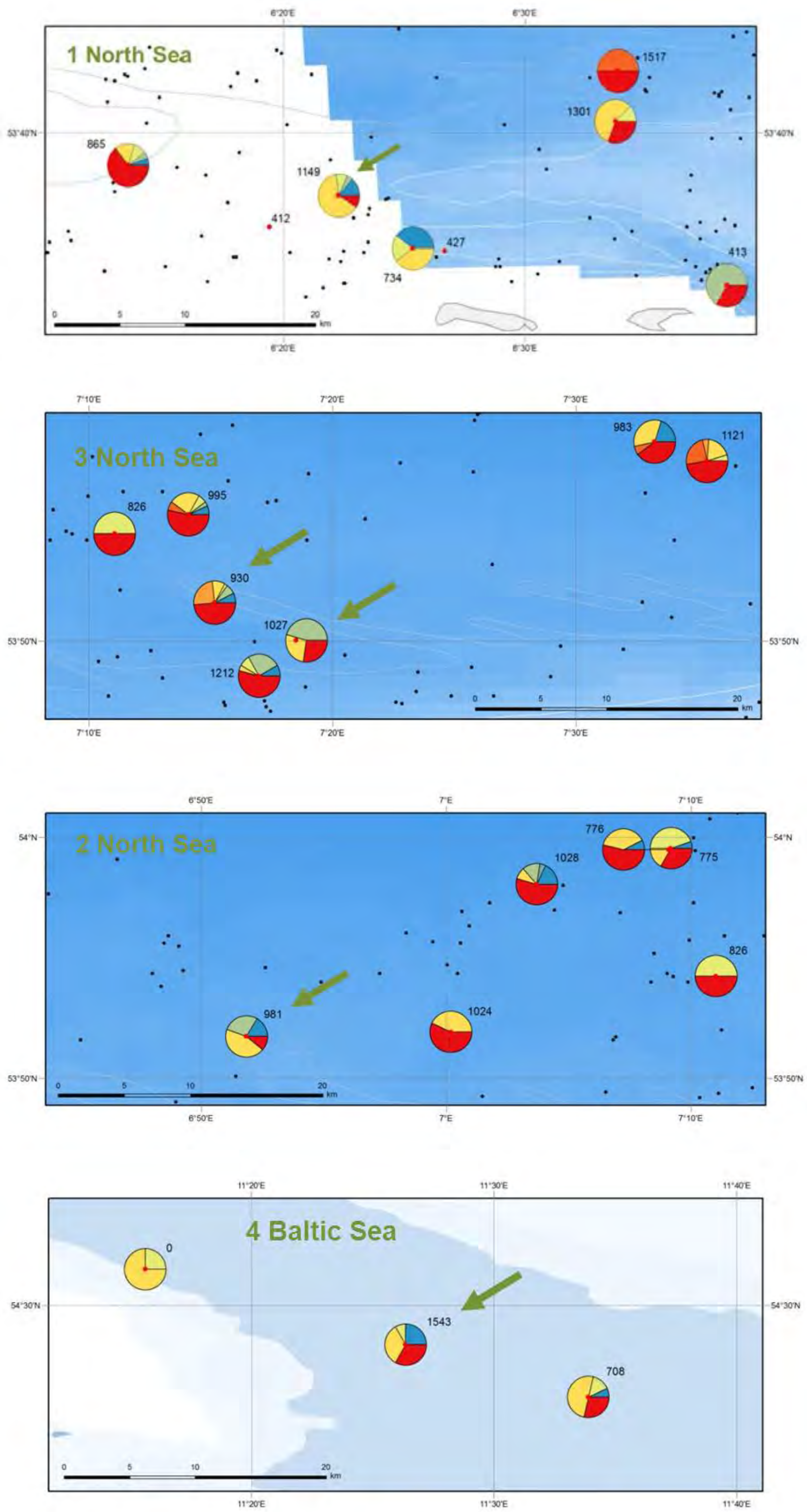


Figure 6-27: Positions of selected wrecks (green arrows) located in the German Bight (North Sea) and the Baltic Sea, recommended for scientific diving investigations on trapped and entangled ‘ghost nets’. Map numbers compare figures 2 and 3.

6.7.3.2 Litter on the Seafloor

From a total of 1642 fishery hauls, 1486 could be assigned to 16 spatial clusters (Figure 6-28). All clusters were compared independent of the time of data ascertainment with reference to the total amount of inorganic litter. The spatial overlaps and distributions of the samples allowed for a comparison between five beam trawls and otter trawls. In four clusters a temporal comparison was possible.

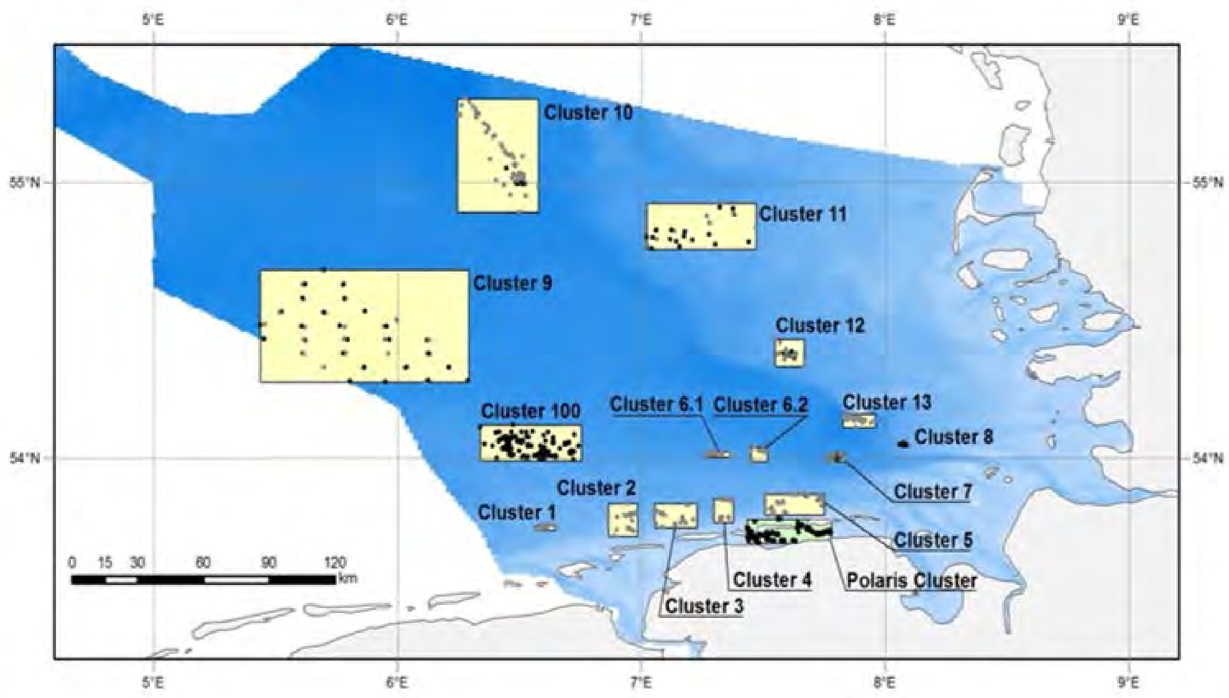


Figure 6-28: Overview map of all processed cluster (black dots – beam trawl; grey dots – otter trawl).

6.7.3.2.1 Beam trawl results

The distribution of the beam trawl catches allowed for the distinction of eight clusters of different sizes (Table 6-13). Seven clusters were located in the open German Bight. A cluster encompassed the seaside and Wadden Sea-side of the East Frisian Island Langeoog. The amount of inorganic litter varied between the clusters by a decimal power. In the north-western German Bight, in cluster 10, the smallest amount of litter with approximately 1 kg km^{-2} was detected. In cluster 7, at the most eastern border of the investigation area, the highest litter densities were detected with approximately 29 kg km^{-2}. Together with the remaining beam trawl data it becomes apparent that far from the coast the amounts of litter are lesser than nearer to the coast. An exception to this is cluster 8, which lies very close to the coast in the most inner part of the German Bight, yet it does not reveal the highest litter densities.

The values of litter from cluster 8 are probably only limited in being representative because this cluster with 1.8 km^2 area is the smallest sampled unit. To some extent, the spatial heterogeneity of generated litter within the individual clusters can be substantial (Figure 6-29) and therefore it can be stated already at this point that clusters for comparison should be of consistent size in order to the representativeness of the results.

The spatially unequal distribution of the litter within a cluster reflects on one hand the patchy distribution of litter. On the other hand, it can't be excluded that the in part strongly irregular sampling of a cluster will lead to such a result. In addition, areas for which only one sample is recorded will underlie strong distortions if this very sample by chance reveals very high amounts of litter. In order to minimize the effects of unequal distributions in the samples the Kriging method was presently applied. Nonetheless, the unsystematic sampling must be considered in all further evaluations. Using Kriging, irregularly distributed beam trawl catches from single clusters were presently used to generate a value for a position (the cluster). In this way the demonstrated spatial distribution of the amounts of litter across the German Bight (in amounts of litter per km²) represents a strong expansion of the description of the litter distribution in the German Bight using fewer individual samples per position (compare Galgani et al., 2000). The latter does not allow for considerations of small scaled local differences. We attribute the fact that in the German Bight more litter is found closer to the coast than far away from it, to the southerly concentrated fisheries industry, shipping traffic and coastal tourism, because a large amount of the litter most likely consists of fishing twines and nets as well as plastic bags (compare chapter 1.7.2.4). The reduction in the amounts of litter with increasing distance to the coast is confirmed by Thiel et al. (2011) who focussed on the amounts of drifting litter in the sea. Carefully averaged over all beam trawl catches, a mean litter contamination of 10 kg km^{-2} can be calculated for the German EEZ (North Sea). Transferred to this area covering 28,539 km^2, this refers to a total amount of 285,390 kg of inorganic litter. This amount must be viewed as minimum amount, because an unknown number of particles will be passing through the meshes of the beam trawls, because the continuous contact to the seafloor is most probably not warranted at all times during a trawl and because probably not all pieces of litter that are partly or entirely covered by sediment are picked up by the trawls.

Table 6-13: The amounts of inorganic litter (±SD) in eight clusters in the German Bight (compare Figure 9). The data were calculated with the Kriging method. Given are number of samples as well as information on area and time. Sampling was conducted via beam trawling (BT).

Fishing gear	Cluster	Number of samples	Area [km^2]	Period	Inorganic litter [kg $Cluster^{-1}$]	Inorganic litter [kg km^{-2}]
BT	7	12	14.60	2001 - 2008	209.04±74.38	28.69±10.21
	8	9	1.81	2001 - 2010	37.56±32.59	9.84±8.54
	9	28	2507.13	2003	6467.29±3153.71	5.29±2.58
	10	13	967.06	2001 - 2010	6.49±9.31	1.30±1.87
	11	18	541.84	2004 - 2007	1116.22±271.85	2.37±0.58
	12	15	83.56	2001 - 2010	34.59±7.19	3.82±0.79
	100	353	389.03	2000 - 2008	3694.90±1892.32	9.52±4.88
	Polaris	443	223.39	1998 - 2006	4936.06±2105.79	18.42±7.86

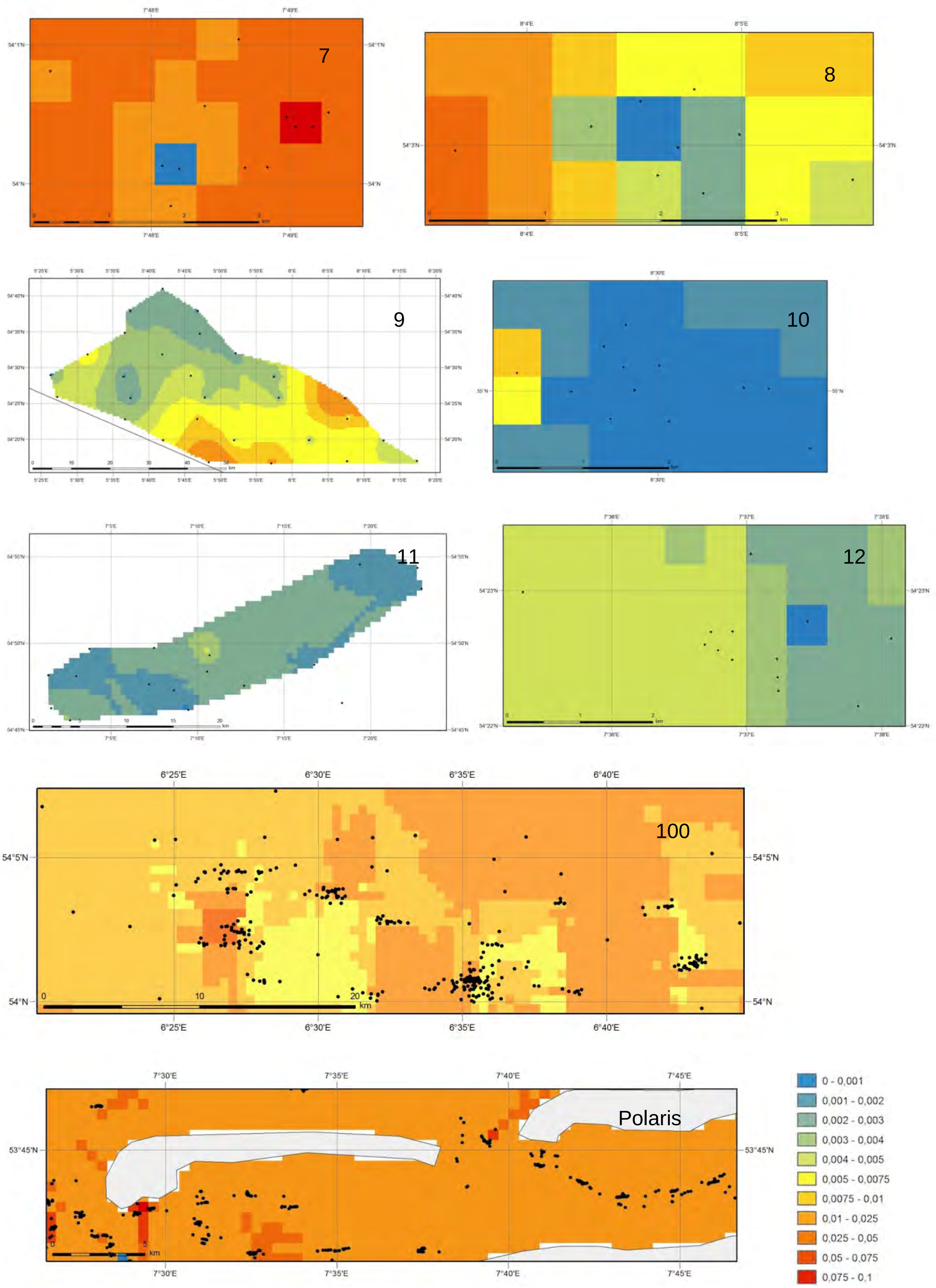


Figure 6-29: Spatial inorganic litter distribution in eight clusters inside the German Bight (whole sampling period) gained by beam trawling. Results [g m^{-2}] were obtained using data generated by the Kriging method. Black dots: samples.

For cluster 100 (the area northwest off the island of Borkum) (Figure 6-30) and the cluster 'Polaris' (Figure 6-32) temporal comparisons on the amounts of litter were possible. In cluster 100, the amount of litter declined from the time period of 2000-2002 until 2006-2008 with an increase during the middle time period (Figure 6-31). In the cluster 'Polaris', the amounts of litter around the Island of Langeoog had at first tripled and then declined by the factor 5 (Figure 6-33).

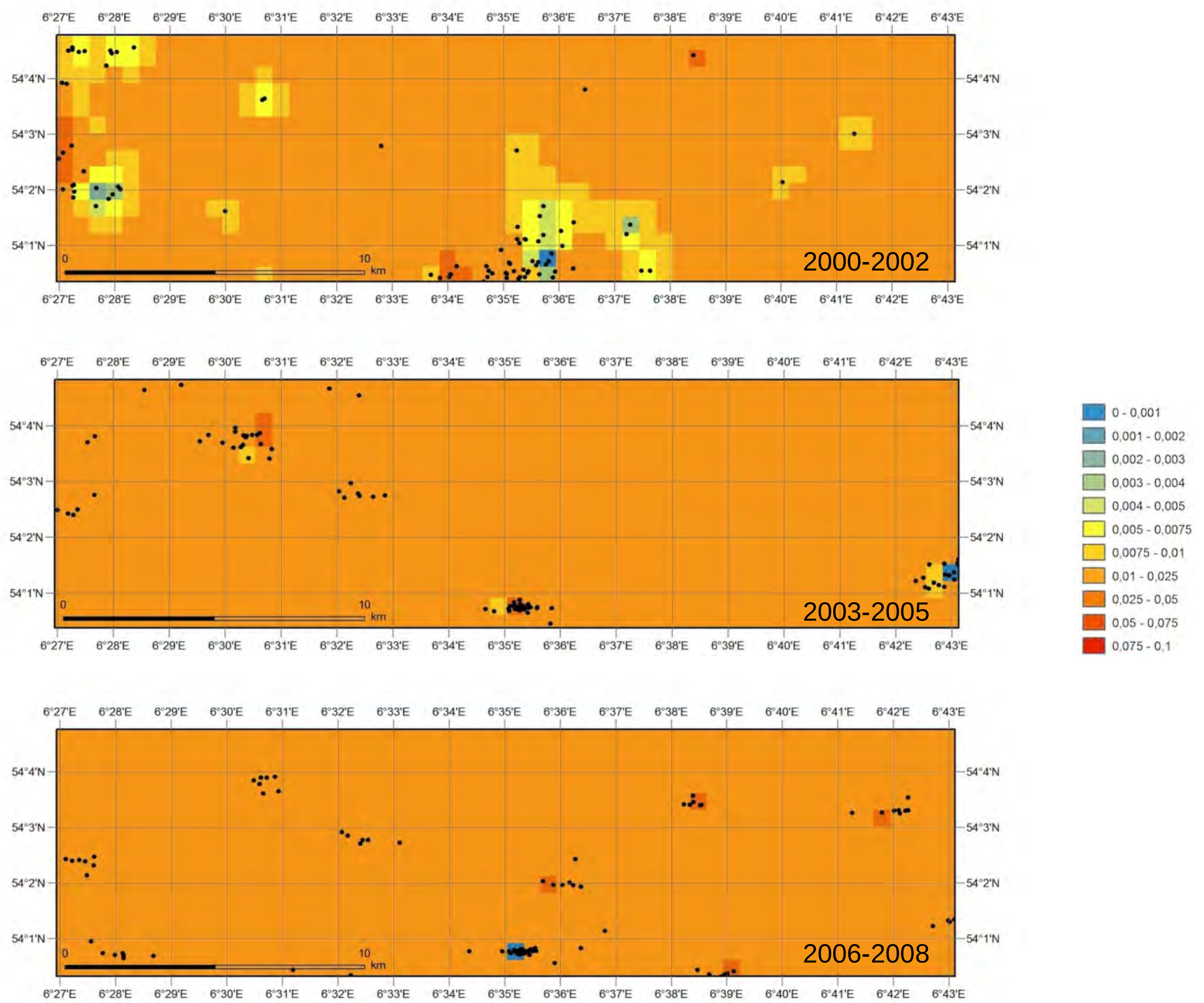


Figure 6-30: Multi-temporal comparison of the amounts of litter [g m^{-2}] in cluster 100 in three time periods. Results calculated with data using the Kriging method. Black dots: samples gained by beam trawling.

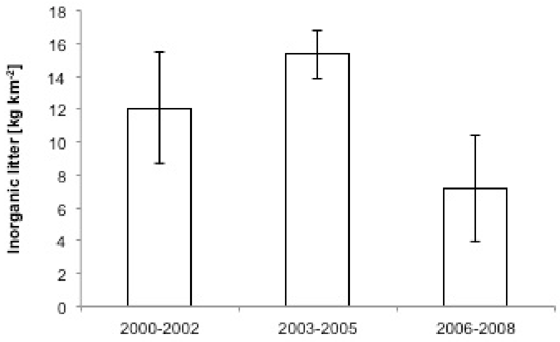


Figure 6-31: Average amounts of inorganic litter (±SD) in three time periods for cluster 100. Samples gained by beam trawling.

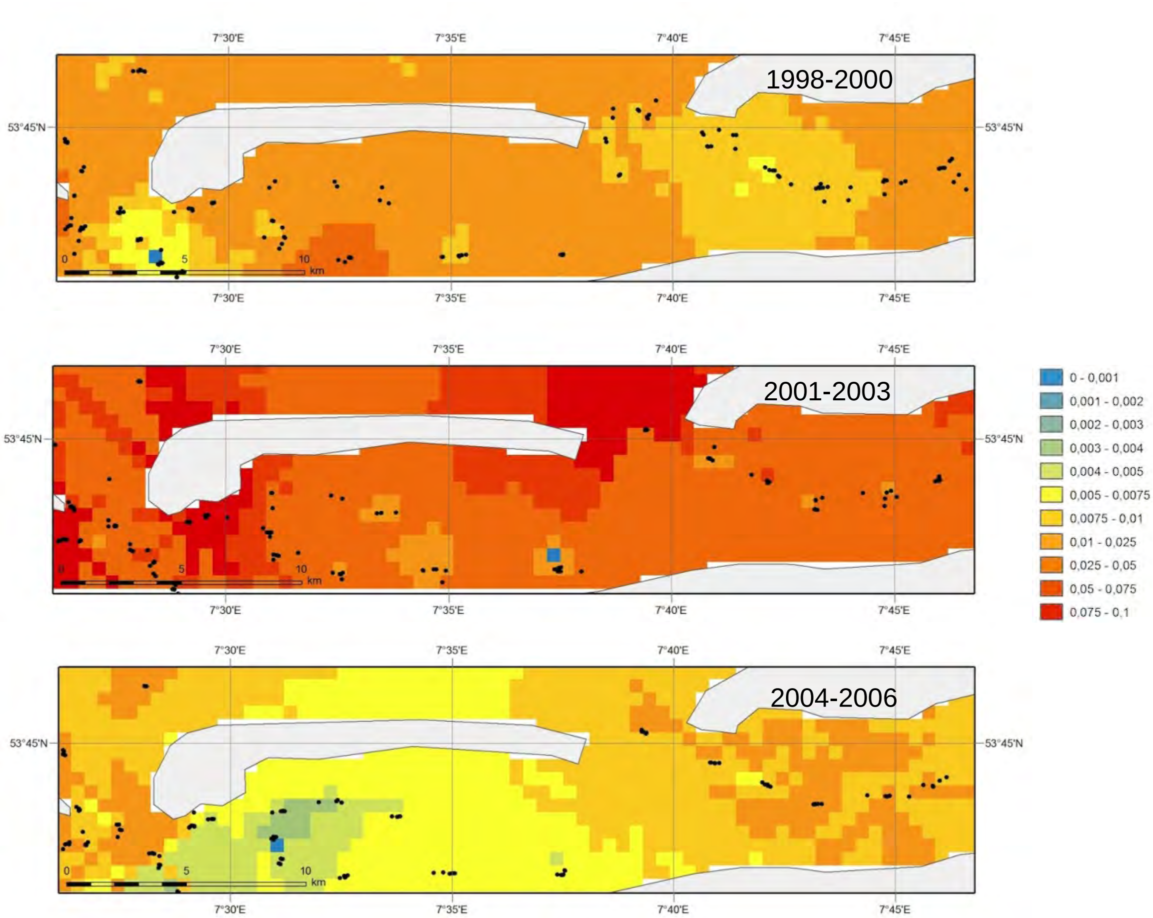


Figure 6-32: Multi-temporal comparison of the average amount of inorganic litter [g m^{-2}] in the area of the island Langeoog (cluster 'Polaris') in three time periods. Results calculated with data using the Kriging method. Black dots: samples gained by beam trawling.

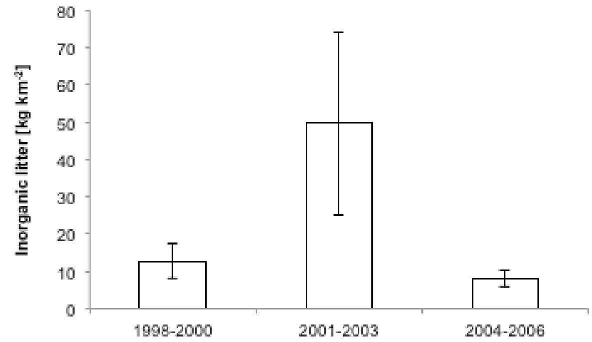


Figure 6-33: Average amounts of inorganic litter (±SD) in three time periods for cluster 'Polaris' (area: island of Langeoog). Samples gained by beam trawling.

Reasons for either increase or decrease of litter are not shown here. Only longer monitoring periods can show, if these trends are stable. These two comparably large clusters can be used as examples to demonstrate that changes can be detected in the areal litter contamination of specific areas.

6.7.3.2.2Otter trawl results

The distribution of the otter trawls allowed for a division into 14 clusters (Table 6-14, Figure 6-34 and Figure 6-35).

Table 6-14: The amounts of inorganic litter (±SD) in 14 clusters in the German Bight (compare Figure 14 and 15). The data were calculated with the Kriging method. Given are number of samples as well as information on area and time. Sampling was conducted via otter trawling (OT).

Fishing gear	Cluster	Number of samples	Area [km^2]	Period	Inorganic litter [kg Cluster^{-1}]	Inorganic litter [kg km^{-2}]
OT	1	10	10.04	2004 - 2009	170.12±25.54	12.88±1.93
	2	21	103.43	2001 - 2009	1331.96±1063.41	11.64±9.29
	3	17	111.51	2001 - 2009	922.61±360.13	7.36±2.87
	4	17	49.42	2000 - 2008	1700.49±984.12	30.34±17.56
	5	31	135.96	2000 - 2009	757.45±338.56	5.17±2.31
	6.1	20	13.25	2001 - 2009	399.52±76.89	24.79±4.77
	6.2		25.27	2000 - 2009	108.35±20.93	3.84±0.74
	7	36	14.60	2001 - 2010	130.44±159.93	6.82±8.36
	9	29	2507.13	2000 -2003	16335.08±21716.29	6.44±8.56
	10	54	967.06	2000 - 2010	7299.48±889.51	7.38±0.90
	11	11	541.84	2006 - 2007	4474.55±1745.11	8.69±3.39
	12	24	83.56	2000 - 2010	410.08±51.10	4.53±0.56
	13	69	46.84	2000 - 2010	779.85±453.83	14.46±8.42
	100	191	389.03	2000 - 2009	2899.03±1112.01	7.56±2.90

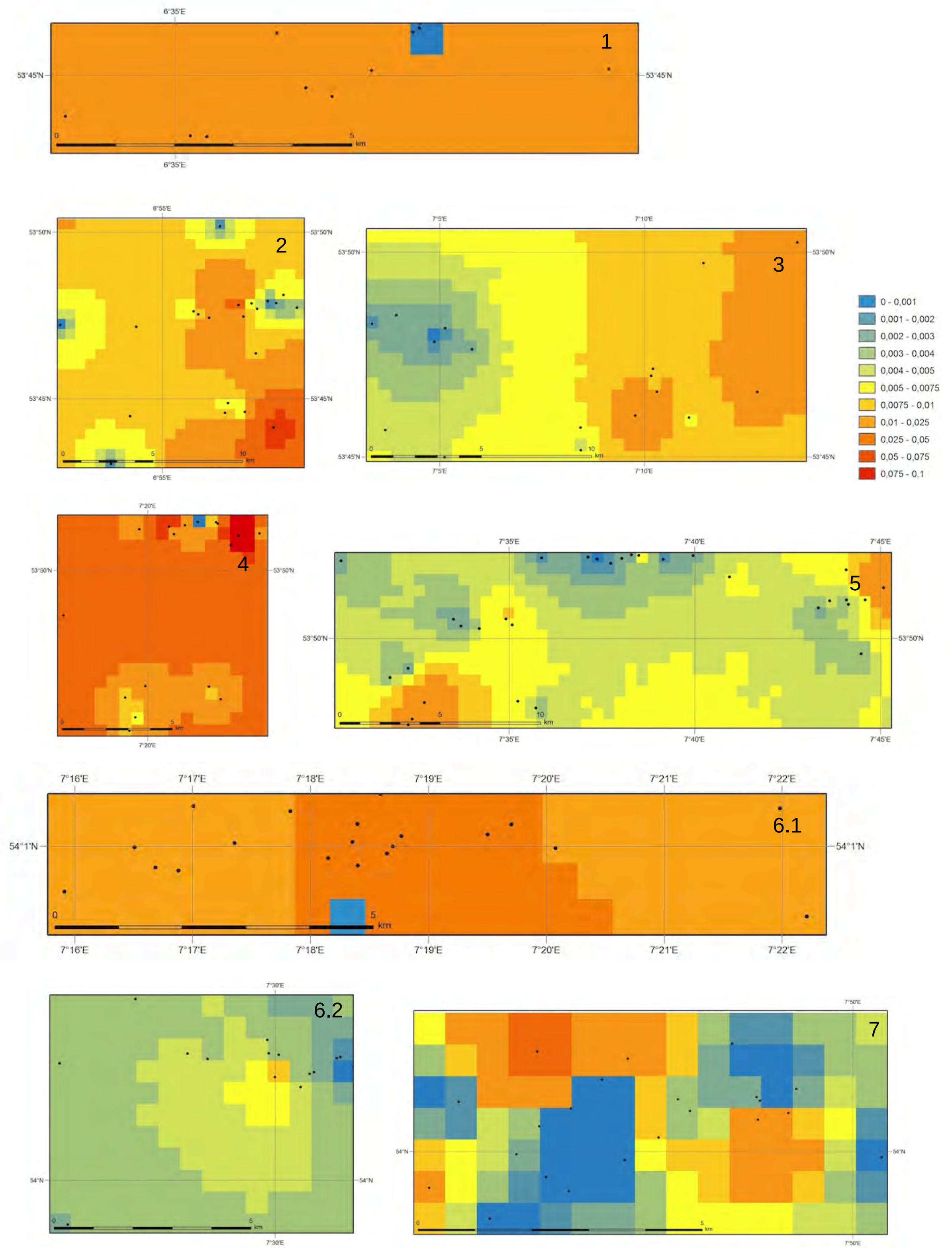


Figure 6-34: Spatial inorganic litter distribution in 8 of 14 clusters inside the German Bight (whole sampling period) gained by otter trawling. Results [g m^{-2}] were obtained using data generated by the Kriging method. Black dots: samples.

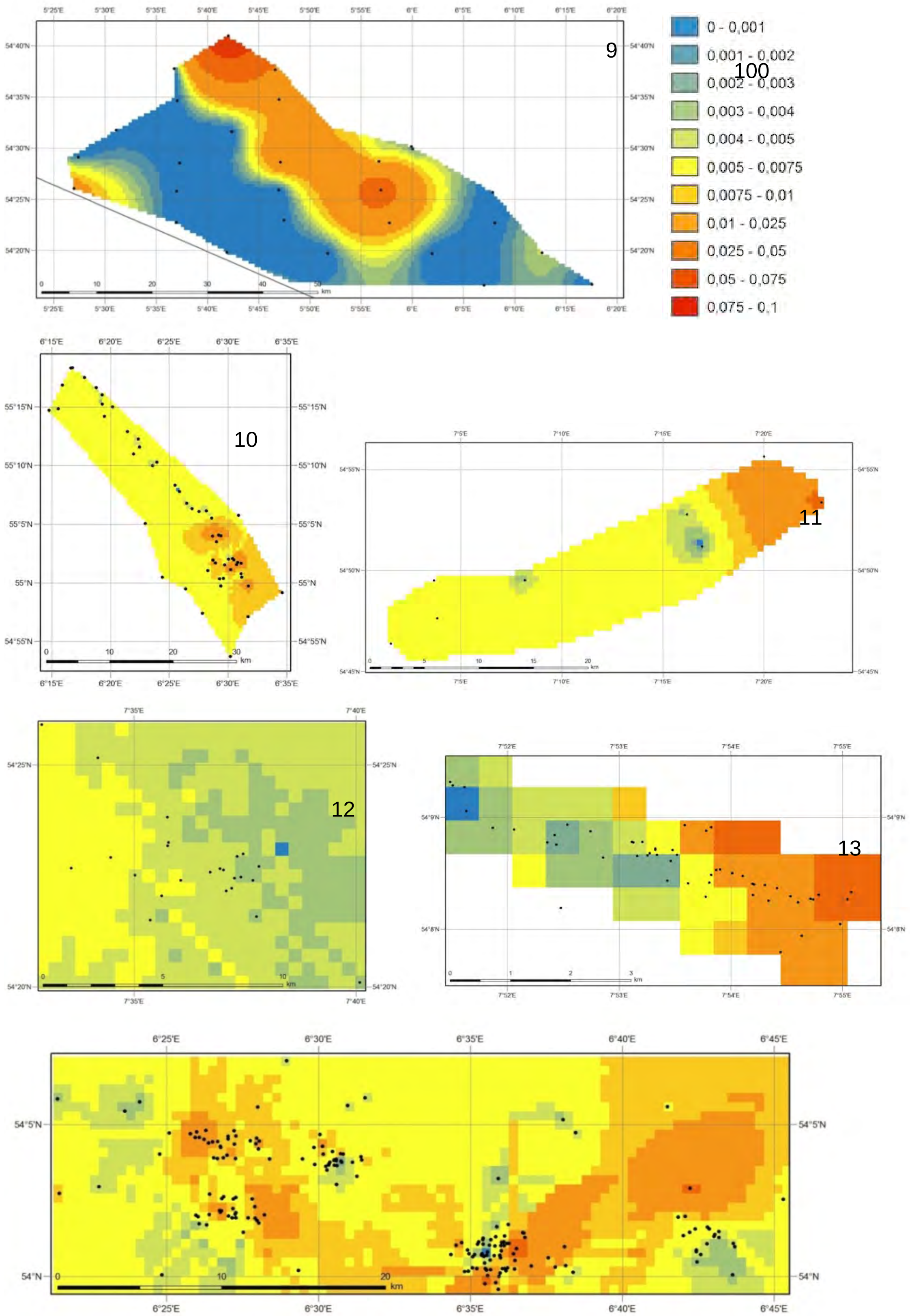


Figure 6-35: Spatial inorganic litter distribution in 6 of 14 clusters inside the German Bight (whole sampling period) gained by otter trawling. Results [g m^{-2}] were obtained using data generated by the Kriging method. Black dots: samples.

As described for the beam trawls, the otter trawls revealed a similar distribution of litter in the German Bight. In close distances to the coast more litter was found than in distances further away. Close to the coast, off the island of Baltrum, the highest amounts of litter were found revealing approximately 30 kg km^{-2}. Very low values of approximately 6-7 kg km^{-2} were found in the clusters of the central German Bight. On average and as in the results for the beam trawls, the litter contamination detected through otter trawls amounted to ~11 kg km^{-2}. Similar as for the beam trawl results, therefore, the minimum amount of inorganic litter detected through otter trawls in the EEZ amounts to 313,929 kg.

The temporal comparison was possible for clusters 10 and 13 (Figure 6-36 and Figure 6-38). In the coastal-near cluster 13 the amount of litter had doubled (Figure 6-37) while it had increased fourfold in the coastal-far cluster 10 (Figure 6-39). Just as in the case of the beam trawls, a reason for this trend is not detectable presently. Here as well, further monitoring periods are needed to show whether the present trends are stable. The otter trawl data from three positions far away from each other suggest a general increase in the amounts of litter in the offshore areas of the German Bight.

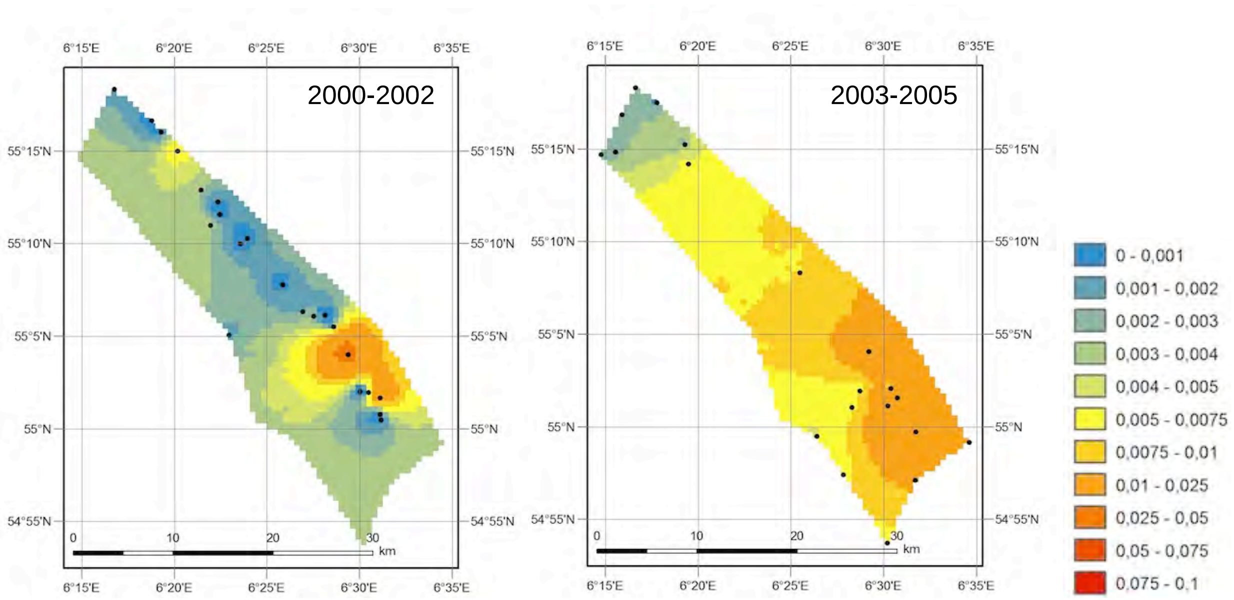


Figure 6-36: Multi-temporal comparison of the average amount of inorganic litter [g m^{-2}] in cluster 10 in two time periods. Results calculated with data using the Kriging method. Black dots: samples gained by otter trawling.

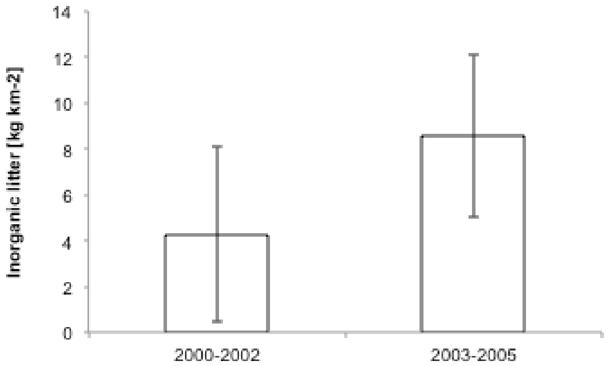


Figure 6-37: Comparison of the average amounts of inorganic litter (±SD) in two time periods for cluster 10. Samples gained by otter trawling.

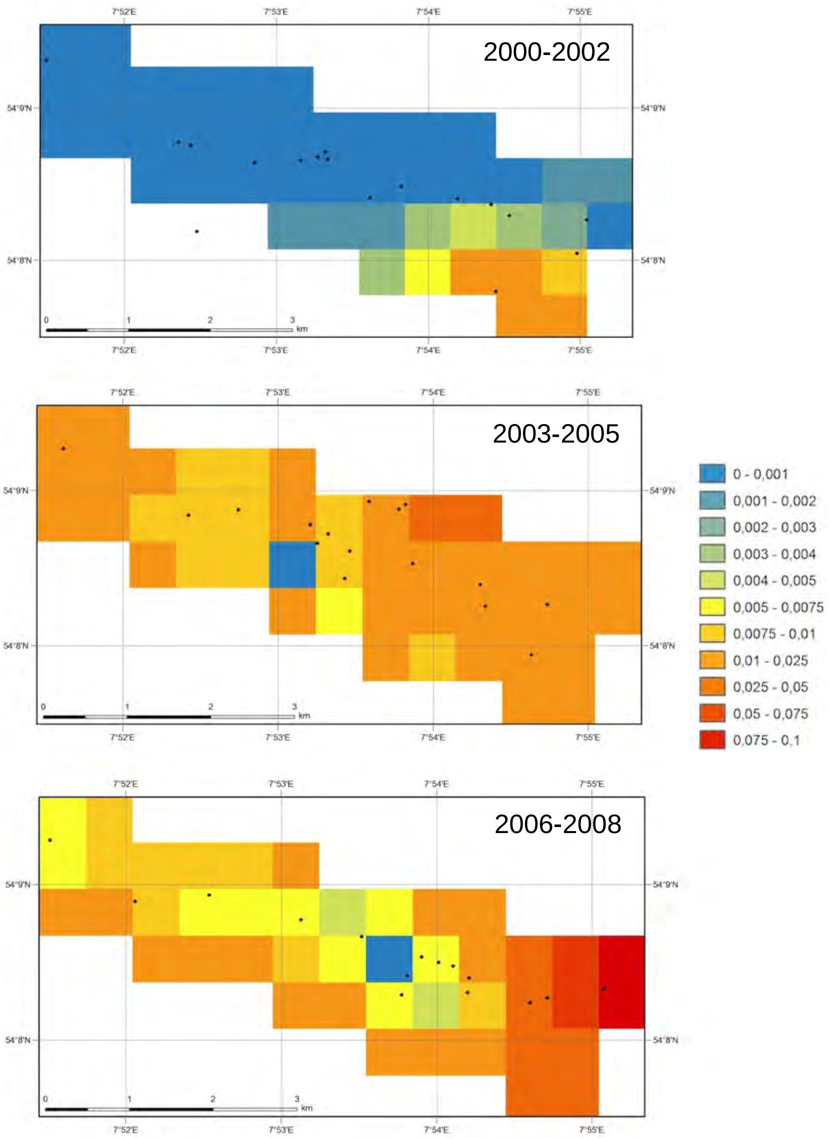


Figure 6-38: Multi-temporal comparison of the average amount of inorganic litter [g m^{-2}] in cluster 13 in three time periods. Results calculated with data using the Kriging method. Black dots: samples gained by otter trawling.

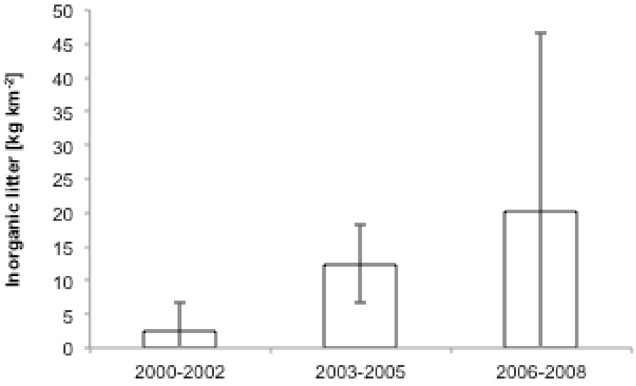


Figure 6-39: Comparison of the average amounts of inorganic litter (±SD) in three time periods for cluster 13. Samples gained by otter trawling.

6.7.3.2.3Beam trawl versus otter trawl

The comparison of the catchability of the beam trawl and the otter trawl method was possible in five clusters, which are of the same size and which were sampled at the same times (Figure 6-40). No consistent difference in their catchability for inorganic litter was detected between the beam trawls and the otter trawls. In two clusters the same high amounts of litter were detected in both methods. In two clusters (10 and 11) the amounts of litter obtained through otter trawls were about threefold higher than those obtained through beam trawls. Contrarily, in one cluster (7) the amounts of litter obtained through beam trawls were about fivefold higher than those obtained through otter trawls. The contact to the seafloor is probably less close and more inconsistent in otter trawls (through rolling wheels connected to the bottom rope) than in beam trawls, which are connected to a chain which penetrates into the seafloor and causes animals to be chased out of their environment. Nonetheless, and in contrast to the expectations, a higher catchability of the beam trawls per unit area could not be shown. The partly very different results probably are on account of the patchy distribution of the litter and the unequal distribution of the samples from both methods within the individual clusters. Both nets reveal the same mesh width of 1 cm and it can be, therefore, assumed that both trawling methods select for the same particle size and that subsequently small litter particles are similarly underrepresented in both trawling methods. Because the individual compartments in the group of inorganic litter cannot be recorded systematically, it is impossible to detect a selective catchability in one of the two trawling methods. An improved and systematic comparison of the two methods could be reached through an equal distribution of samples obtained by beam trawls and otter trawls in one cluster and through the application of differentiated protocols and uniform recordings. Because, however, there seems to be no systematic difference between the two methods and under the consideration that with both methods nearly the same mean amount of litter in the clusters was obtained (for OT approximately 11 kg km^{-2} and for BT approximately 10 kg km^{-2}), according to the present state of knowledge, the two trawling methods can be applied and treated in equal manner.

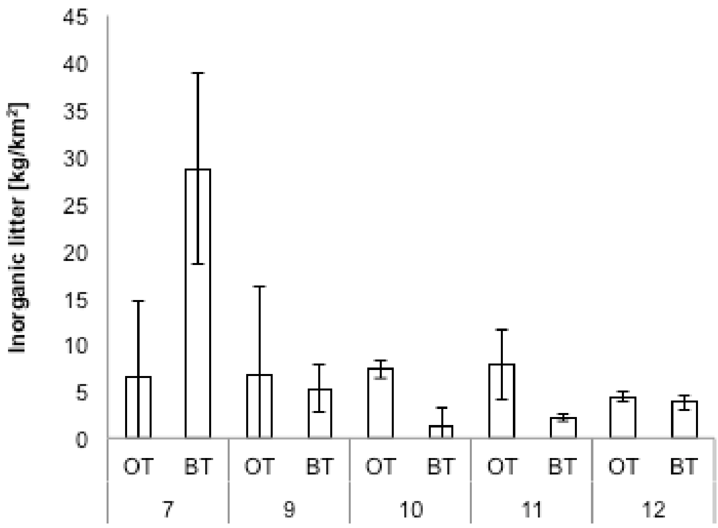


Figure 6-40: Methodological comparison of otter trawls (OT) and beam trawls (BT). Average amount of inorganic litter (±SD) per cluster and area over the whole sampling time period.

6.7.3.2.4 OSPAR and AWI protocol

In 2007, 29 fishery hauls (17 beam trawls and 12 otter trawls) were recorded with both the AWI protocol and the OSPAR-beach litter monitoring protocol for 100 m of beach line (compare Figure 6-21). Deviating from the routine method and for comparability purposes, the wet weights of the litter fractions were integrated into the beach litter protocol as well as into the AWI protocol. With the Heincke protocol only four resp. three different fractions were distinguished in the considered beam trawl and otter trawl catches. With the finer resolution in the beach litter protocols, the litter could be classified into 13 (beam trawl) and 11 (otter trawl) categories, respectively (Table 6-15). It can be assumed that if more otter trawls hauls were applied, the number of distinguishable categories would reach similar numbers as in the beam trawls. Different selectivities in the methods are not visible. The application of protocols which reveal higher resolutions than the AWI protocol are reasonable, because offshore litter is obviously not bashed up into small and unidentifiable fractions, but can certainly be partitioned into distinguishable categories and its origins or sources can be determined as well. In the Heincke protocols the different types of litter were commonly distinguished without proper procedure guidance and conducted each time through different staff members. The OSPAR protocols, however, were consistently performed by the same staff members. The percentage shares of the litter assigned to the categories of the OSPAR protocols can, therefore, be transferred as estimated values to all clusters and catches following the AWI protocol.

The actual composition of inorganic litter on the seafloor in the individual clusters is determinable only through actually repeated and differentiated recordings of litter. The presently summarized data can only be viewed as estimations. The differently sized rope parts and net fragments taken together form the largest distinguishable group with a share of approximately 50 % for beam trawls and 40 % for otter trawls. This overarching category is most likely on account of shipping and fishery industry. Industrially produced wrapping and packaging material accounts for a share of approximately 15 %. A reduction of litter from these specific fractions could result in a measurable reduction of the entire amount of generated litter.

Table 6-15: Comparison of AWI protocols and the 100 m beach litter monitoring protocol applied to identical catches inside the German Bight at different sites in 2007. Presence/ absence (upper table) and the percentage of litter (> 1%). Orange colour = group of net like categories.

Heincke protocol								OSPAR BEACH LITTER MONITORING [g]
Gear Sample	Fisching area [m²]	Glass	Plastics	Inorganic litter ind.	Net	Rope	Number of cathegories	Number of cathegories
BT 1	2681						1	1
BT 2	2861						1	1
BT 3	2808						1	4
BT 4	4371						1	2
BT 5	2580						1	2
BT 6	2427						1	1
BT 7	2245						1	2
BT 8	3397						1	2
BT 9	2597						1	2
BT 10	3077						1	1
BT 11	1815						1	1
BT 12	4158						1	3
BT 13	3039						1	2
BT 14	2677						1	2
BT 15	2787						1	3
BT 16	2418						1	3
BT 17	2755						1	2
Total							3	13
OT 1	30002						1	4
OT 2	29569						1	2
OT 3	37387						1	2
OT 4	31997						1	1
OT 5	31598						1	3
OT 6	32449						1	3
OT 7	32555						1	2
OT 8	30804						1	2
OT 9	36406						2	2
OT 10	36208						1	1
OT 11	35172						1	4
OT 12	36551						1	2
Total							3	11

OSPAR item columns (presence marked by shading): Plastic/ polystyrene: 4/6-pack yokes, Bags (shopping), Small plastic bags, Drinks, Cleaner, fast food containers, Cosmetics, Engine oil <50 cm, Caps/lids, Cups, Rope/cord/nets < 50 cm, Rope/cord/nets > 50 cm, Tangled nets/cord, Fibre glass, Hard hats, Shotgun cartridges, Plastic/polystyrene pieces < 50 cm, Industrial packaging, plastic sheeting, Styropor<50cm[g], Strapping bands; Rubber: Balloons, Boots, Tyres and belts; Cloth: Clothing, Sacking, Rope/ strings; Paper: Bags; Metal: Crab/lobster pots, Drink cans, Bottle caps, Aerosol/Spray cans, Wire, wire mesh; Glass: Bottles, Light bulbs/tubes, Other glass items; Sanitary waste: Condoms, Cotton bud sticks, Tampons; Faeces: Human, Animal; Other: Knife handle.

Gear	Heincke protocol [%]					OSPAR BEACH LITTER MONITORING [%]														
						Plastic								Cloth		Metal	Glass		Other	
	Plastics	Anorganic litter ind.	Net	Rope	gesamt	Bags (shopping)	Small plastic bags	Rope/cord/nets < 50 cm	Rope/cord/nets > 50 cm	Tangled nets/cord	Plastic/polystyrene pieces < 50 cm	Industrial packaging, plastic sheeting	Strapping bands	Clothing	Rope/ strings	Drink cans	Bottles	Other glass items	Knife handle	gesamt
BT	41	47	6	6	100	6	5	22	16	11	5	16			5	5		3	6	100
								49												
OT	58	42			100	24	2	14	23	1		14	6	8	1		7			100
								38												

6.7.3.2.5 Suggestions for seafloor litter monitoring

For the monitoring of the amounts of litter in the German Bight we suggest to choose five positions which lie in the presented clusters (Figure 6-41). Hereby, the degree of contamination through litter can be distinguished for coastal-near and coastal-far positions and for the Wadden Sea area. These positions revealed differently strong gradients of litter contamination and are distributed over the entire investigation area. In this way, the development of the amounts of litter in the German Bight can be estimated representatively.

Specifically, in order to properly and actually monitor the amount of litter in coastal-near areas, the clusters 100, 7 and 13 are recommended to be sampled. The coastal-far sampling procedure is recommended to be conducted in cluster 10. The generated litter in the Wadden Sea should be monitored in the area of the island Langeoog (cluster Polaris), representing the only location in direct vicinity of the coast. At these four above mentioned stations, further investigations can be built on previously conducted temporal comparisons of the AWI data. At the same time, the amounts of litter in these areas are high. In the areas of the island Langeoog, locally stationed commercial fishermen could be included into a monitoring procedure (K. Wätjen, pers. comm.). These fishermen use beam trawls for their fishing hauls.

In the offshore area, the clusters should be sampled through systematically conducted fishery hauls. For the sampling methodology, beam trawls as well as otter trawls can be applied. The monitoring is recommended to be conducted using a uniform design in all clusters; however, different methods should be applied therein. Apart from the Langeoog area (tidal zone with tidal creeks and isolated areas) uniform and equal cluster sizes should be chosen. The data evaluation should be conducted in a uniform way, for example with the presently applied Kriging method, in order to improve the comparability. Hereby, the size of cluster 13, which was sampled with beam trawls, could be chosen as reference (approximately 47 km^2). With the selected grid size of 0.005°, a total of 91 quadrates can be determined within cluster 13. We suggest applying at least 10 regularly distributed samples per cluster and sampling date in order to sample 10% of the calculated unit area. When taken from positions regularly distributed across the clusters, this minimum of samples allows not only the Kriging of the data, but also the application of simple parametric and/or non-parametric statistical tests for analysis. High resolution protocols should be applied. These allow for gaining more information on the composition and origin of the litter. In a subsequent approach and for comparison purposes with the AWI data, the category 'inorganic litter' can be generated according to its presently used definition. For recording the amounts of litter, the categories of the OSPAR beach litter monitoring protocols can be used. This enables the sufficient differentiation of litter and optionally allows for the direct comparison of the amounts of litter on the seafloor and at the beaches.

The investigations have shown that in all areas of the German Bight inorganic litter is present on the seafloor. Bearing in mind that the harmful effects of inorganic litter are insufficiently described so far, the 'good or best state of the sea' would be achieved by completely ridding the sea from any kind of litter. Under consideration of ongoing intakes of litter into the sea and difficult to identify and hardly quantifiable outtakes of litter from the sea, we recommend in a first step to reduce the intakes to an extent where the presently in the offshore areas detected intake of litter is stopped. We understand that a 100% ridding of litter from the German Bight is not possible. Furthermore, we suggest to define <10% of the presently determined amounts of inorganic litter per cluster as being the 'good state' of a specific cluster. Using the example of the island Langeoog it was shown that fluctuations of the amounts of litter can reach up to 50%. A reduction of the intake of litter and, therefore, a reduction of the total amounts of litter by 90% (over long time) appears to us as realistic and implementable aim. Focussing only on the reduction of industrially produced wrapping and packaging material and the litter produced through the shipping and fishery industry, this could contribute to over 60% in the reduction of the amounts of litter (compare table 5). The presently shown amounts of litter could be used as reference therefore. The respective smaller values [kg km^{-2}] calculated with Kriging from the clusters 10, 100 and 7, where otter trawls and beam trawls are performed (during the entire investigation time period), should be used as reference basis.

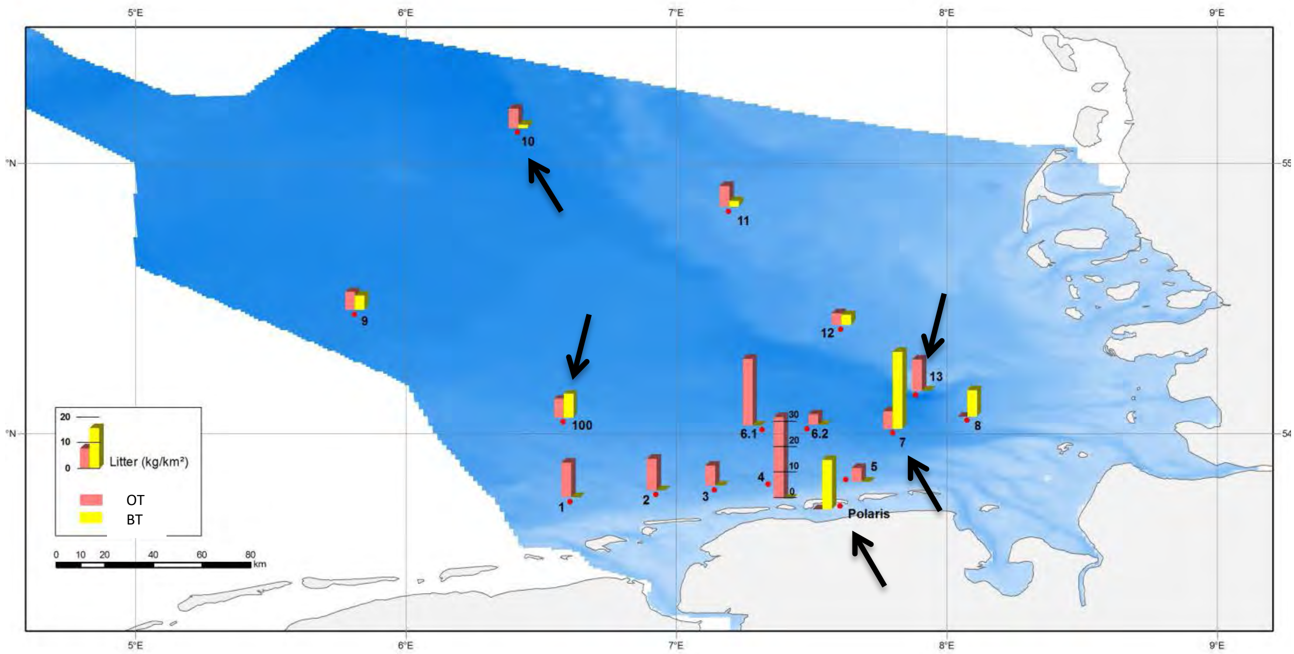


Figure 6-41: Overview of the average amounts of inorganic litter (kg km^{-2}) in the German Bight. Selected clusters for a long-term monitoring are marked (arrows). Applied methods are beam trawling and otter trawling. For clusters 1, 2, 3, 4, 5, 6.1, 6.2 and 13 no beam trawling data exist. For the cluster 8 and Polaris no otter trawl data exist.

6.7.3.2.6Outlook

The evaluation and analysis of the 'ghost net' data and the fishery hauls is completed. In a further project work the implementation of a specific sampling design will be discussed. Thereby, and in participation of institutions conducting scientific fishery surveys, it will be determined whether existing beam trawl and otter trawl surveys in the North Sea can be incorporated into a monitoring programme in order to sample the proposed clusters and it will be discussed when the next sampling surveys could be conducted.

6.7.4 Acknowledgement

We thank the German Environment Agency (UBA) for financial support of the project. We are grateful to the Federal maritime and Hydrographic Agency (BSH) and in particular to H. Pietrek for the provision of the wreck data. We thank all the divers of the BSH and the crew of the WEGA for their support in the wreck video evaluation. Many thanks to C. Schneider for his reliable sightings of the wreck videos. J. Dannheim and L. Gutow provided the Benthos Data Base of the Alfred Wegener Institute for Polar and Marine Research (AWI), for which we are very thankful. We thank K. Wätjen and R. Knust for the provision of the Polaris data. We thank J. Eisenberg for the data processing by Kriging and for the graphical display of the data with ArcGis. We are grateful to K. Boos for the revision of the text.

7 Work package 6: Aggregation of pressures to an overall assessment (Integrated Ecosystem Assessment)

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7.1 Introduction

The concept of the Marine Strategy Framework Directive (MSFD) is a holistic approach based on the assessment of the European marine ecosystems. It has the objective to ensure the sustainable use of marine goods and services and simultaneously preserve the marine environment. Additionally, the terms 'environmental status' and 'good environmental status' (GES) are defined in Article 3 of the Directive 2008/56/EC of the European Parliament and of the Council (European Commission, 2008). Both definitions refer to the overall state of the environment. This implies that any assessment should result in an ecosystem assessment evaluating this state and deciding whether the GES is reached or not. This would lead to two classes for the assessment results. However, the status can also be classified in five classes as it has been done within the Water Framework Directive (European Commission, 2000; European Commission, 2005). This would yield more comprehensive assessment procedures, additionally reflecting positive or negative trends within the ecosystem. In general, 'Integrated Ecosystem Assessments' comprise three types of procedures: a fully integrated assessment including all ecosystem components, a sectoral assessment of specific human activities or a thematic assessment of specific ecosystem components (Rice et al., 2010). It has been recommended to separately assess pressure and state variables of the ecosystem. This would follow the DPSIR ('Driver-Pressure-State-Impact-Response') approach which is a causal framework interlinking the state of an ecosystem with the human impacts, their causes and consequences and the management responses (EEA, 2005). The integration can take place at the levels of indicators, criteria and descriptors and several approaches are conceivable. The methods have already been evaluated and discussed in detail by Altvater et al. (2011). The following chapter will give a short summary of the underlying basic principles.

7.2 Existing methods

7.2.1 'One-out, all-out' (OOAO) approach

The simplest way of an integrated ecosystem assessment is the 'one-out, all-out' (OOAO) approach. That means that the overall status is determined by the worst status of the components used in the assessment. This procedure has several strengths and weaknesses. On the one hand, using OOAO is a precautionary approach and thus, may function as an early warning system. On the other hand, the assessed system will tend to be downgraded depending on the number of parameters used and their respective reliability (Borja and Rodríguez, 2010; Altvater et al., 2011). Single parameters might get a significance that is too high, and groups of parameters with a causal relationship might get the same level of significance, but would not really reflect the relationships and importance in the ecosystem. In contrast to the MSFD, the WFD CIS guideline (WFD CIS, 2003) describes aggregation rules and states, in order to apply the OOAO on the biological quality element level using averaging or weighting of parameters. This holds only for parameters related to the same pressure. Within the MSFD, the OOAO principle might be applied to the status descriptors (D1, D3, D4 and D6) as recommended by the BLMP (2011) due to their importance and overlapping properties. Following this approach, the overall good status of the ecosystem cannot be reached, if one of these status descriptors fails to meet the conditions for the good environmental status. Especially when integrating a higher number of assessment criteria the OOAO can be extended alternatively to two criteria (two-out, all-out, TOAO) or a certain proportion of criteria which thus determine the bad overall status if not achieving the limits. This might decrease the probability of downgrading the whole system by a single criterion that is not of eminent importance.

7.2.2 Averaging and weighting

Averaging is another simple method of aggregation. The averaging can be done in several ways, e.g. by using the mean or the median for all assessment parameters, and can be additionally supported by weighting the single elements. One critical point about averaging is the 'averaging-out' effect: This means that information on parameters with an exceptional high or low value or status will be smoothed by averaging. The more elements are used in the averaging, the higher is the possibility of averaging out an impact factor. Furthermore, the methods used for averaging may have an effect on the overall result. Assessment results may be highly sensitive to the aggregation rules, as Ojaveer and Eero (2011) showed in their study about methodology in marine environment assessment by comparing several ways of averaging.

Weighting is not able to eliminate the 'averaging-out' effect, but can be an appropriate method for ranking the elements used for the aggregation, because these often are not directly comparable to each other. Usually, expert judgment is used for applying the weighting procedure, which can lead to subjectivity in the evaluation process. To overcome this problem, as many experts as possible should be included in the assessment. The criteria for the weighting should be clear and transparent. One method could be an adaption of the criteria described by Halpern et al. (2007). They used the vulnerability of an ecosystem to the threat as a rating scale and defined this vulnerability by five criteria: spatial scale, frequency, functional impact, resistance, and recovery time. Borja et al. (2011) developed an integrated assessment using the indicators and descriptors of the MSFD for the example of the Basque Country with a weighed averaging system. They also included the reliability for indicators and weighed and averaged them as well. However, they did not discriminate between pressures and state descriptors.

7.2.3 Decision tree

Another method that can be utilized for assessing the overall status of an ecosystem is the decision tree. Due to the possible schematic presentation with clear alternatives it may be easy in application and support the decision. The simplicity and visual approach may increase the understanding and acceptance by stakeholders. However, the order of indicators and the decision rules have to be chosen and this may be a subjective issue. In several studies decision trees were used for an ecosystem assessment (Campbell, 2008, Borja, 2009; Bundy et al., 2010). Within the WFD a decision tree (Figure 7-1) is used in combination with weighed averaging and the OOAO approach for assessing the status.

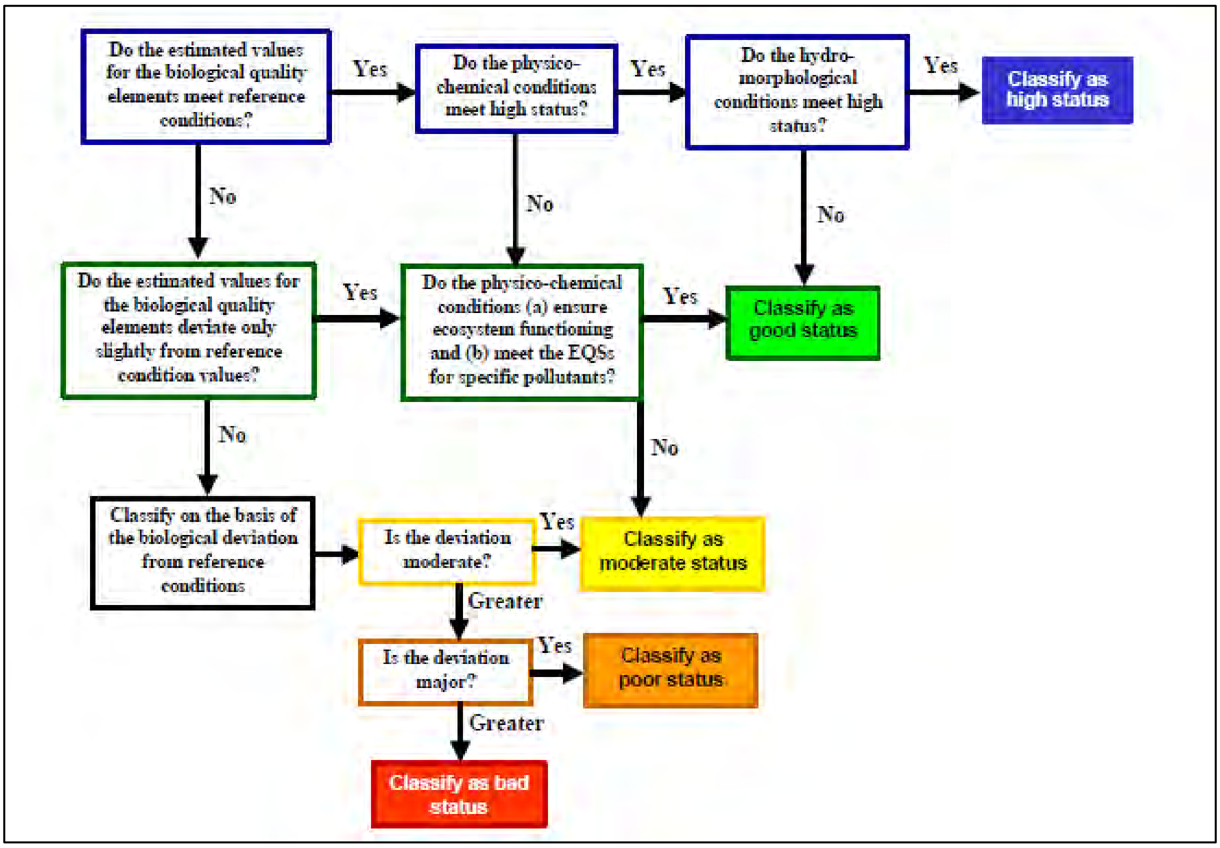

Figure 7-1: Overview of the relative roles of biological, hydromorphological and physico-chemical quality elements in ecological status classification according to the normative definitions in WFD Annex: 1.2 (EC, 2005; from Altvater et al., 2011).

7.2.4 Decision Matrix

In a decision matrix all interactions between impacts and ecosystem components can be presented as a schematic overview in form of e.g. a triangular matrix or layered matrices (Altvater et al., 2011; Dixon and Montz, 2005). This visualisation may help to understand the complex interactions. Furthermore, thematic assessments may be carried out as described by the SEAMBOR (Science Dimensions of Ecosystem Approach to Management of Biotic Oceans Resources) working group (Altvater et al., 2011). Assessments may be carried out either with regard to a selected sector as one aspect of a marine environment, e.g. the plankton, or with regard to a particular human activity such as dredging or other (Figure 7-2). In this way, various sectors, ecosystem components and socio-economic factors may be integrated in a fully integrated assessment.

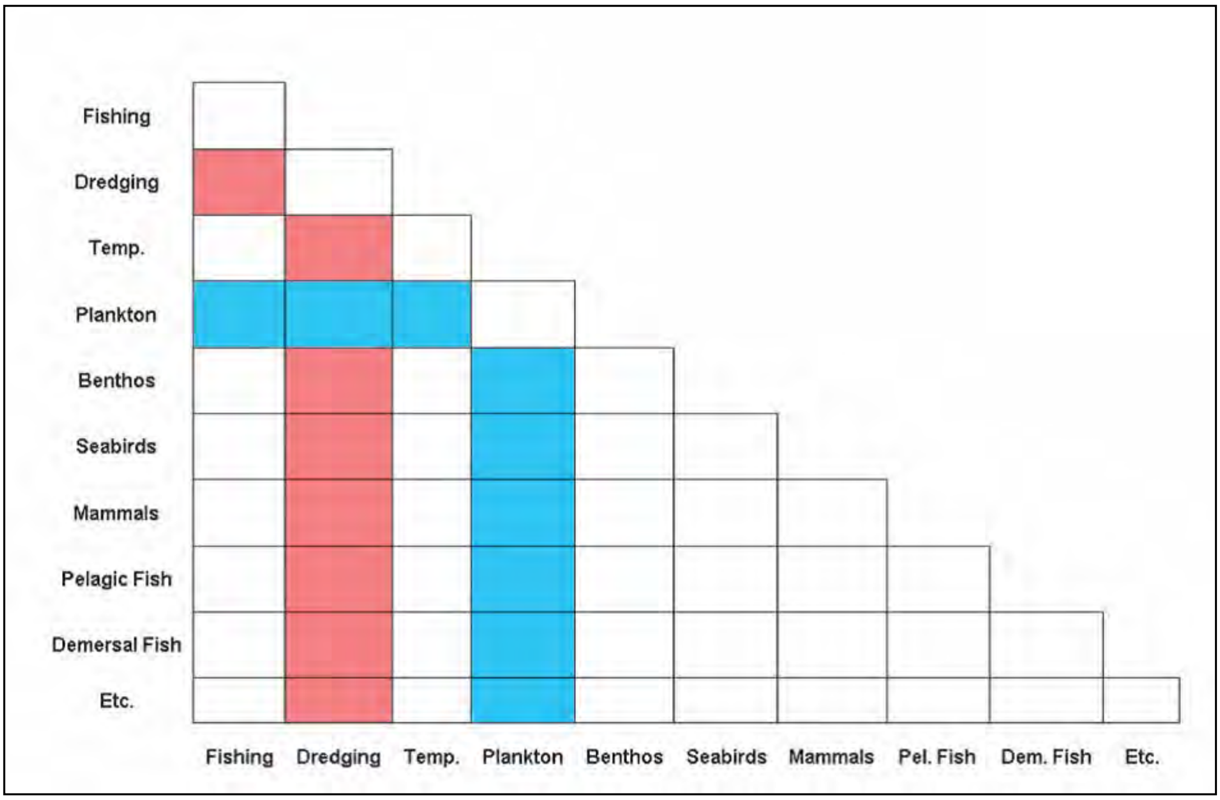

Figure 7-2: Example of a matrix approach used to describe the relationship or degree of interconnection between human pressures (sectoral activities such as fishing) and ecosystem components (such as benthos). The specific interactions between all sectors and ecosystem components can be readily observed. For example, the specific interactions (as impacts) between dredging and all other components of the system can be documented (highlighted in red), this would be an example of a sectoral or sector-specific assessment. In addition, the interactions between plankton and all other ecosystem components, including sectoral pressures, can be evaluated, and this would be described as a thematic assessment (highlighted in blue); from Altvater et al. (2011).

7.2.5 Different Indices

Another possibility to assess the overall state of an ecosystem is the translation of the human impact into an index. Rombouts et al. (2013) have reviewed and discussed several methods for indices to assess the health of marine ecosystems. Several indices were also described in detail by Altvater et al. (2011), for example, the Ocean health index (Halpern et al., 2012), the Chesapeake Bay Health Index or the combination of the three indices for the Assessment of Estuarine Trophic Status (ASSETS) by NOAA. HELCOM developed one index for the pressures, the Baltic Sea Pressure Index (BSPI), and another index for the impacts, the Baltic Sea Impact Index (BSII) (HELCOM, 2010). Halpern et al. (2008) calculated cumulative human impacts scores and mapped them globally. Dependent on the method of calculation, the pressures and impacts and their relationships can be reflected and integrated in an overall assessment result referring to the status.

7.2.6 Multivariate Analyses

Some methods use multivariate analyses for assessing multiple parameters. An example is the approach by Tett et al. (2007b, 2008, 2013 in press,) which defines a domain in a two to multidimensional space that can be considered a 'reference' state for the system (Figure 7-3). The divergence of the actual state to the reference area then reflects the disturbance in the system and can be used to calculate an index for the overall status as well. Several more are discussed in the report of the Task Group 6 (Rice at al., 2010).

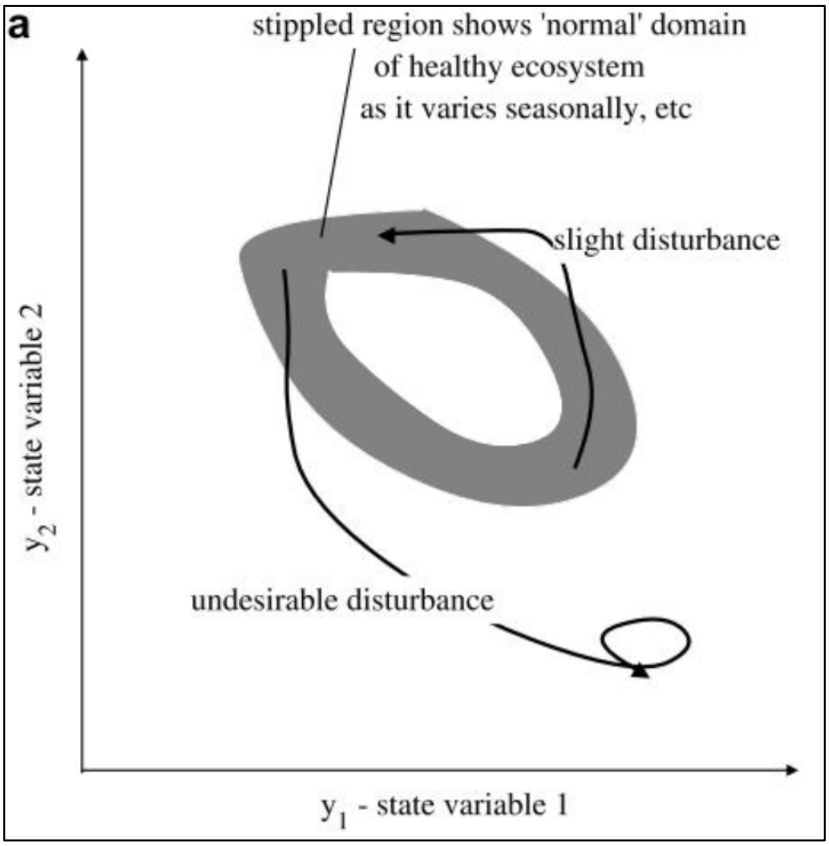

Figure 7-3: Domain of 'reference' state defined by state variables (grey doughnut shaped area) and diverging situations outside this region reflecting disturbances. From Tett et al. (2007).

7.2.7 Cumulative assessment

For a fully integrated ecosystem approach the cumulative effects of all pressures have to be taken into account. The combined effects of all stressors may be different from the individual effects of each stressor. This is due to interaction between the stressors. The combined effects may be additive, synergistic or antagonistic and also interact in a complex way. The cumulative effects and models for their assessment are currently investigated in another R&D project funded by UBA (UFOPLAN 3711 25 216).

7.2.8 Combinations

Various combinations of different methods are possible, and such combinations are recommended in order to combine the advantages of the individual methods and to outweigh or improve the disadvantages. One example is the combination of the OOAO approach and the weighed averaging with a decision tree used within the WFD, explained in detail by Altvater et al. (2011). By combining the OOAO approach with weighting and grouping of parameters, the precautionary approach is improved by taking into account ecological relationships and meanings without averaging out possible indicative values of the biological quality elements. A similar concept is the assessment method, HELCOM (2010) suggested for the integrative assessment. Even a first cumulative procedure is described within that approach (for further information see Altvater et al. (2011) and the references therein).

7.3 First draft of an integrative ecosystem assessment for the implementation within the MSFD

As already described in previous chapters, there are different methods for combining partial classifications on different levels to an integrated overall assessment of the ecosystem status. The MSFD is dealing with 11 descriptors (4 status and 7 pressure descriptors). Additionally, cumulative effects will have to be included and assessed. For a particular marine region to be classified, this will result in 12 individual assessments that have to be aggregated to an integrated overall assessment. In order to achieve that goal, three conceptual approaches will be described in the following. One pre-condition for applying this approach is the development of individual assessments systems for each descriptor.

7.3.1 Weighted integration of descriptor assessments

The simplest way of integrating the results of all 12 (i.e. 11 descriptors + cumulative effect) individual assessments into an overall assessment would be to calculate just an average value. But this would mean that all values taken into account will have the same weighting. This might not reflect the real significance of the variable for and its impact on the ecosystem. For that reason, a weighting procedure should be carried out, being similar to the approach of Halpern et al. (2012) for calculating the Ocean Health Index. The weighting factors will have to be determined by expert judgement based on the most recent research results. The calculation of the Integrated Ecosystem Assessment (IEA) will then be carried out as follows:

$$IEA = \sum (f_{D1} \cdot AV_{D1} \dots f_{D11} \cdot AV_{D11} + f_{kum} \cdot AV_{kum})/12$$

Where f is the weighted factor for each descriptor and the cumulative effects, AV is the corresponding assessment value.

As mentioned before, there are only two classifying stages for the implementation of the MSFD: either the 'Good Environmental Status' (GES) is reached or not. If the GES target was missed, this could well mean that either the conditions were far away from the good status or that the GES status has just been missed by a short distance. For that reason, it will be recommended to employ a five-stage classification system for assessing the individual descriptors as well as for calculating the overall assessment. Especially, if weighted mean values are used for the overall assessment, it may make a great difference for the final results whether a five-stage classification system is used or only a two-stage system. Moreover, a more detailed classification scheme will provide much more information on respective trends within the ecosystem, which is important for the application of possible management measures. For the calculation of the integrated overall assessment values, normalised values from the partial assessments will have to be used.

7.3.2 Combination of "One Out, All Out" and weighted assessment

Stringently applying the OOAO or TOAO approach to the 11 descriptors of the MSFD as well as to the cumulative effects would mean that the GES would not be reached if only one or two of the individual assessments missed the target. In that case, the question will arise whether this can be justified in all cases, of course being strongly dependent on the importance of the factor or the factors causing the negative classification. It should be considered whether one or two negative assessment values could be allowed without hampering the GES. The consequence would then be that some important descriptors could have a negative evaluation and nevertheless, the GES would be reached.

In the first place, the four MSFD status descriptors (D1, D3, D4 and D6) will give direct information on the condition of the ecosystem components and their trophic interrelationships. They are of course strongly influenced by the other pressure descriptors and the cumulative effects. In order to stress the importance of these status descriptors, the OOAO approach might be applied according to the respective guideline for the implementation of the MSDF (Krause et al. 2011). That would mean that the GES could only be achieved, if each of the four descriptors reached the GES target. If one of the descriptor assessments failed, the target would be missed then. In this case, the assessment results of the remaining descriptors and of the cumulative effects would not be of any interest any longer.

If all four status descriptors yield a positive assessment result, the individual assessments of the other descriptors will be included in the overall assessment. For this procedure, two alternative ways exist: First of all, the weighted averages for all components can be used, as already has been described in the previous chapter. Secondly, the status descriptors and the pressure descriptors including the cumulative effects will be separately calculated and averaged as two groups. For the status descriptors, it has to be considered whether they will be weighted within this procedure or be treated as equivalent. Any accumulation of impacts should be weighted. In any case, the five-stage assessment system adapted to the two-stage MSFD system should be applied.

7.3.3 Integrative Assessment of the ecological status based on the results of the indicators

In contrast to the previously described methods suggesting methods for the aggregation of the results of the descriptors, the following proposed methods for an overall assessment are based on the aggregation of indicators, which are described in the Commission Decision (2010/477/EU) for the MSFD (2008/56/EU). The basic idea was to regroup the indicators for different descriptors to describe different aspects of the ecological status and being able to separate aspects describing the status and aspects describing the anthropogenic pressures.

7.3.3.1 Ecological index for the identification of areas of concern

In the following proposal, the indicators are grouped into status variables and pressure variables. Here, the indicators describing the status variables are compared to the indicators describing the intensity and effects of anthropogenic pressures (see report UFOPLAN 3711 25 216 for details).

Put in short terms, indicators representing similar aspects of different descriptors will be compiled to different categories. These categories will then be divided into two groups of variables: status variables and pressure variables (Table 7-1, Figure 7-4). The results of the cumulative assessment can be integrated by either completely integrating the pressures into the cumulative effect assessment or by combining the pressures in a first step and multiplying the results with the index of the cumulative effects assessment. The cumulative assessment can provide an index indicating how much the cumulative effects increase the effects of the anthropogenic pressure on the different ecosystem components in comparison to a simple addition of the effects of pressures.

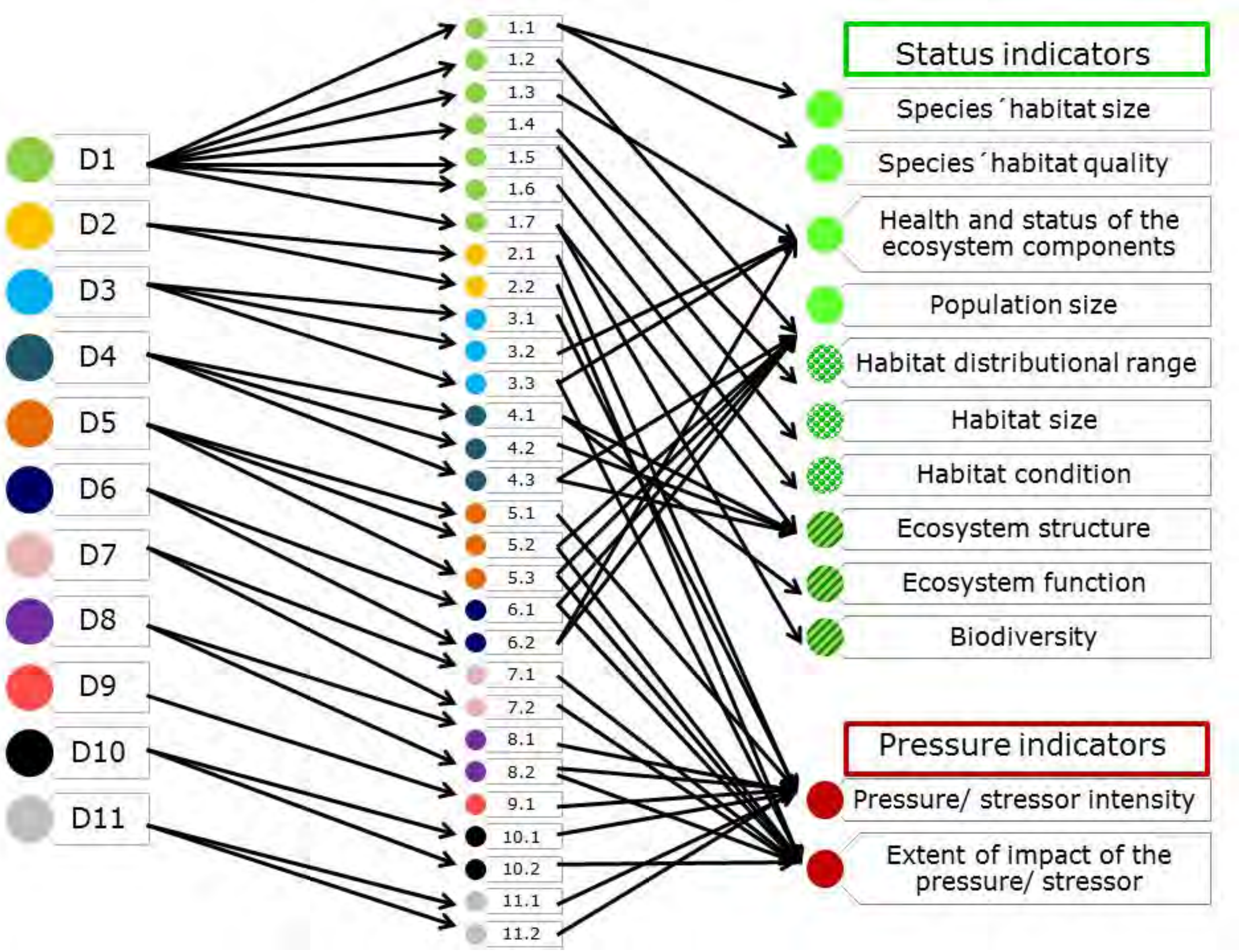

Figure 7-4: Regrouping of the indicators listed in the MSFD to different aspects of ecological status and anthropogenic pressures.

Additionally, the sensitivity and the recovery potential of the occurring ecosystem components will be taken into account. The relation of the status variables and the pressure variable will indicate if the ecological value is very high and at the same time is threatened by a high number and/ or a high intensity of pressures, following Coll et al. (2012) (Table 7-1). Figure 7-5 shows exemplary approaches for an integrative ecological assessment based on MSFD indicators.

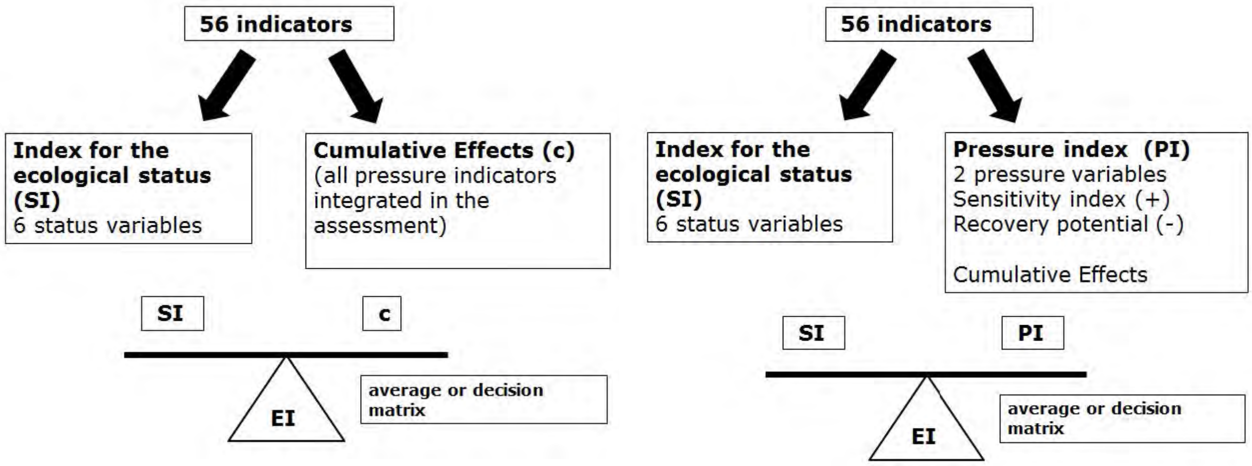

Figure 7-5: Summary of two approaches for an integrative ecological assessment based on MSFD indicators. The ecological index (EI) is determined by comparing the index for the ecological status (SI) and the pressure index (PI) with each other. The ecological index can also be determined by a decision matrix.

In general, we tried to propose a concept following the requirements of the SMART concept (ICES, 2005): The assessment method should be specific, measurable, achievable, realistic and time-bound. Furthermore, the system should consider the aspects of the DPSIR framework and include drivers, pressure, state, impact and response as process parameters (Rapport and Friend, 1976; EEA, 2007; reviewed in Altvater et al., 2011).

Many indicators refer to particular organism groups or indicator species based on the consideration that different species and species groups show individual sensitivities towards different pressures or stressors. Therefore, all status variables referring to the ecosystem status should first be treated separately for each ecosystem component and then be summed up. It should be defined how many ecosystem status variables and thus ecosystem components should be included and how the ecosystem status variables should be selected to provide comparability of the ecological status between different eco-regions. For example, a certain number of Red List species could be chosen, a certain number of key species, a certain number of umbrella species and functional groups could be chosen with regard to the species groups highlighted in descriptor 4 and 6. However, all species and species groups mentioned in the MSFD will have to be included.

Table 7-1: Status variables and pressure variables sorted by assigned criteria (letters indicate that one criterion is split into different categories).

Category	Criteria
Status variables	
Species/ species groups	
Habitat size of the species/ species group	Distributional range (1.1.1) , area covered by the species (1.1.3)
Habitat quality of the species/ species group	Distributional pattern, where appropriate (1.1.2), possibly more aspects not particularly mentioned in the MSFD, if appropriate

Category	Criteria
Health and status of the ecosystem components	Population demographic characteristics (1.3.1), population genetic structure, where appropriate (1.3.2), Spawning Stock Biomass (SSB) (3.2.1), Biomass indices (3.2.2), Proportion of fish larger than the mean size of first sexual maturation (3.3.1), Mean maximum length across all species found in research vessel surveys (3.3.2), 95% percentile of the fish length distribution observed in research vessel surveys (3.3.3), presence of particularly sensitive and/or tolerant species (6.2.1), Multi-metric indexes assessing benthic community condition and functionality, such as species diversity and richness, proportion of opportunistic to sensitive species (6.2.2), Proportion of biomass or number of individuals in the macrobenthos above some specified length/size (6.2.3), Parameters describing the characteristics (shape, slope and intercept) of the size spectrum of the benthic community (6.2.4),
Population size/ abundance/ biomass	Population abundance and/ or biomass, as appropriate (1.2.1), special emphasis on abundance/distribution of key trophic groups/species (4.3), sensitive and tolerant species (6.2.1), macrobenthos (6.1.1), algae (5.2.1, 5.2.3) and perennial seaweeds and seagrasses (5.3.1).
Habitat	For each habitat type:
Habitat distributional range	Habitat distributional range (1.4.1), distributional pattern (1.4.2)
Habitat size	Habitat area (1.5.1), habitat volume, where relevant (1.5.2),
Habitat condition	Condition of the typical species and communities (1.6.1), Relative abundance and/or biomass, as appropriate (1.6.2), Physical, hydrological and chemical conditions (1.6.3).
Ecosystem structure (including also food web indicators)	Composition and relative proportions of ecosystem components (habitats and species) (1.7.1).
Ecosystem function	Needs further consideration, see proposals in concept 4 below
Biodiversity	e.g. Shannon-Weaver, Simpson, Richness, 1.7
Pressure variables	
Stressor/ pressure intensity	Abundance and state characterisation of non-indigenous species, in particular invasive species (2.1), Fishing mortality (F) (3.1.1), Ratio between catch and biomass index (hereinafter 'catch/biomass ratio') (3.1.2),

Category	Criteria
	Nutrients concentration in the water column (5.1.1), Nutrient ratios (silica, nitrogen and phosphorus), where appropriate (5.1.2), Concentration of the contaminants mentioned above, measured in the relevant matrix (such as biota, sediment and water) in a way that ensures comparability with the assessments under Directive 2000/60/EC (8.1.1), Occurrence, origin (where possible), extent of significant acute pollution events (e.g. slicks from oil and oil products) and their impact on biota physically affected by this pollution (8.2.2), Actual levels of contaminants that have been detected and number of contaminants which have exceeded maximum regulatory levels (9.1.1), Frequency of regulatory levels being exceeded (9.1.2), Trends in the amount of litter washed ashore and/or deposited on coastlines, including analysis of its composition, spatial distribution and, where possible, source (10.1.1), Trends in the amount of litter in the water column (including floating at the surface) and deposited on the sea-floor, including analysis of its composition, spatial distribution and, where possible, source (10.1.2), Trends in the amount, distribution and, where possible, composition of micro-particles (in particular micro- plastics) (10.1.3), Distribution in time and place of loud, low and mid frequency impulsive sounds (11.1), Continuous low frequency sound (11.2)
<u>Extend of impact of stressor/ pressure</u>	Ratio between invasive non-indigenous species and native species in some well studied taxonomic groups (e.g. fish, macroalgae, molluscs) that may provide a measure of change in species composition (e.g. further to the displacement of native species) (2.2.1) , Impacts of non-indigenous invasive species at the level of species, habitats and ecosystem, where feasible (2.2.2), Size at first sexual maturation, which may reflect the extent of undesirable genetic effects of exploitation (3.3.4),

Category	Criteria
	Chlorophyll concentration in the water column (5.2.1), Water transparency related to increase in suspended algae, where relevant (5.2.2), Abundance of opportunistic macroalgae (5.2.3), Species shift in floristic composition such as diatom to flagellate ratio, benthic to pelagic shifts, as well as bloom events of nuisance/toxic algal blooms (e.g. cyanobacteria) caused by human activities (5.2.4), Abundance of perennial seaweeds and seagrasses (e.g. fucoids, eelgrass and Neptune grass) adversely impacted by decrease in water transparency (5.3.1), Dissolved oxygen, i.e. changes due to increased organic matter decomposition and size of the area concerned (5.3.2), Extent of the seabed significantly affected by human activities for the different substrate types (6.1.2), Extent of area affected by permanent alterations (7.1.1), Spatial extent of habitats affected by the permanent alteration (7.2.1), Changes in habitats, in particular the functions provided (e.g. spawning, breeding and feeding areas and migration routes of fish, birds and mammals), due to altered hydrographical conditions (7.2.2), Levels of pollution effects on the ecosystem components concerned, having regard to the selected biological processes and taxonomic groups where a cause/effect relationship has been established and needs to be monitored (8.2.1), Trends in the amount and composition of litter ingested by marine animals (e.g. stomach analysis) (10.2.1),

Basically, each value of the indicators needs to be compared to a respective specific ecological goal. Proposals for specifications of ecological goals have been defined and are published in the report for the description of the good environmental status for the North Sea and the Baltic Sea (BLMP 2012a and BLMP 2012b).

In a first step, the environmental values of the indicators referring to the status of a species such as indicators for the species distribution, population size and population condition need to be grouped by multiplication of these aspects for each species/ species group. The results for each group can be summed up in a second step. The same approach can be applied for the different habitat types. The different aspects describing each of the habitat types can be multiplied as well and the results of all habitat types can be summed up. An index for the environmental status based on the indicators (SI) is calculated as described below.

In a second step the index is normalised with the ideal values for each of the indicators based on the specifications of the respective ecological goals (BLMP 2012a and BLMP 2012b). The ecological goals are represented as 100% resulting in an overall index for the status indicators between 0 and 1 by deviation of the two indices.

7.3.3.2 Calculation of the status index

The status index (SI) for a certain spatial area could be defined as:

$$\mathrm{SI} = \left(\sum_{\mathrm{n}=1}^{\mathrm{N}} (\mathrm{SD_n} \cdot \mathrm{SH}Q_n \cdot \ \mathrm{PS_n} \cdot \mathrm{PC_n}) + \sum_{\mathrm{n}=1}^{\mathrm{N}} (\mathrm{HD_n} \cdot \mathrm{HE_n} \cdot \mathrm{HC_n})\right) \cdot \mathrm{ES} \cdot \mathrm{EF} \ \cdot \mathrm{Div}$$

where $\mathrm{SD_n}$ is the species distribution of a certain species group, $\mathrm{SHQ_n}$ is the habitat quality of the species, $\mathrm{PS_n}$ is the population size for a certain species group or species, and $\mathrm{PC_n}$ refers to the health of a certain species (group) or to the population condition of a species. The other summand refers to all indicators describing habitat characteristics. The results of the summands will be added up and weighted by the indices for the ecosystem structure, ecosystem function and biodiversity. The index SI will be normalised with the value 0 reflecting a good status, and 1 standing for a bad status by a comparison with the ecological goals as described above.

Alternatively, the calculation of the index could be calculated by adding up the five components: species group related measures, habitat type related measures, ecosystem structure, ecosystem function, and biodiversity, and optionally introduce a weighting of these different aspects.

7.3.3.3 Calculation of the pressure index

The pressure index (PI) would be calculated by multiplying the intensity of all occurring stressors (p) and the effect of each species group or species ($\mathrm{e_n}$) occurring at a particular place group by each ecosystem component. Equivalent to the calculation of the indicator status index, the aspects referring to a certain species/ species group are multiplied and then these results are added up. In addition to the pressure index, the sensitivity and the recovery potential of each ecosystem component would be defined, based on literature data and expert judgement. One purpose of a sensitivity index (s) and recovery potential index (r) is to include a temporal dimension. Very long-lasting pressures should result in a higher pressure index than pressures, which are only temporary or short-living. Another aim is to include properties, which are specifically related to sensitivity and recovery. One example for a sensitivity index was described by Bernem et al. (2000). The criteria defining the sensitivity index should be dependent on the type of pressure. For example, van Bernem et al. (2000) developed a sensitivity index for oil spills in the North Sea. Similar indices could also be developed for further pressures. Halpern et al. (2008) and other publications, which are based on their method, included a sensitivity index in their calculation of an ecological status as well. The sensitivity index and the recovery index could be included in the pressure index by addition and subtraction. The maximum value of the sensitivity index and the recovery potential will have to be < 1 and must have the same normalisation range. The fine scaling of these variables will require further research.

In some cases, in the list of indicators of the MSFD (2008/56/EU), either no effects are considered for some species/species groups or the pressure intensity is not defined and just measured indirectly by the effect on an indicator species. In such cases these indicators could simply be added up to the results of the indicators grouped by species/ species group. Another alternative would be to add additional indicators to provide symmetry between the indices representing pressure intensity and the indicators representing the effects of anthropogenic pressures on habitat types and species.

In a last step the pressure intensities and the effects for each species would be normalised with the ecological goals for the indicators as described above for the calculation of the status- indicator index, where 0 indicates that a pressure is not present or meets the ecological goal, and with 1 indicating the highest pressure intensity.

In a further step, the assessment results of the cumulative effects (Cumulative Index – CI) could be integrated into the equation by multiplication. By taking into account the cumulative effects of stressors, the respective assessment indicates the magnitude of cumulative effects in comparison to a simple addition of effects or intensities of pressures. The index will also yield values between 0 and 1. Therefore the pressure index can be weighted by the cumulative index.

The pressure index would be calculated as follows:

$$\mathrm{PI} = \sum_{n=1}^{N} (\mathrm{p} \cdot \mathrm{e_n} + \mathrm{s} - \mathrm{r}) \cdot \mathrm{CI}$$

The overall calculation of the ecological index (EI) for the MSFD would be expressed as the mean of the pressure index and the status index (c.f. Coll et al. 2012):

$$\mathrm{EI} = (\mathrm{PI} + \mathrm{SI})/2$$

Alternatively, the cumulative effects assessment could be excluded for the calculation of the Pressure Index (PI) (see above). Instead, a cumulative ecological index could be calculated by replacing the pressure index PI by the cumulative index CI based on the indicators representing pressures in this equation:

$$\mathrm{EI_c} = (\mathrm{SI} + \mathrm{CI})/2$$

The lower the ecological index (EI), the better the ecological status will be. It will also be possible to divide the index into a distinct number of classes for indicating the ecological status. Moreover, the results of the PI or CI and the SI could be interpreted with a decision tree: If the pressure index PI *and* the status index SI were very low, then the overall ecological index EI would be low, too, and indicate good ecological conditions. If the indices PI *or* CI were low *but* the status index SI for the ecological status was high, it would have to be considered whether the effects of pressures, which have occurred at that location, were still lasting. Areas, where the pressure index PI *or* cumulative index CI were high and the status index SI was low, should be prioritised for management actions. This way, the index could also give a feedback about the effectiveness of the assessment method if these cases can be excluded. If the status variables were very low and the pressure index was high, the assessment method should be reconsidered and possible reasons for the results should be analysed. For example, it might then have been the case that several aspects had not been considered, although they were ecologically very important, or that the applied expert judgment had not been realistic. Furthermore, such a result might indicate that the uncertainty was too high and that the monitoring programme would have to be improved. It should also be considered to test the model for errors itself. In all these cases described above, the proposed indices might play an important role for the ecosystem assessment and give valuable information on the further development and management.

7.3.4 Ecological index based on modelling of the ecological system

In general, this method follows the principle of the ecological quality ratio (EQR), where the actual environmental status is compared to a reference status. The EQR is mathematically expressed as a division and results in a value between 0 and 1 (WFD 2000, van de Bund and Solimini 2007).

In contrast to the approach in the WFD, where in most of the cases a historical reference state is used, this method refers to a reference state in the future.

The future reference state describes how the ecological health would recover without any negative human influences. This value is calculated by subtracting the percentage alteration of the ecosystem, which can be explained by the influence of anthropogenic pressures (SI_p), from the current ecological status and modelling the recovery over a certain time period. The future reference state of a certain time point (SI_R) is compared to the scenario in which present conditions with regard to anthropogenic pressures would not change (SI_R).

$$\frac{SI_p}{SI_R}$$

The ecological status is not only defined by the intensity of the anthropogenic pressures, but also on interaction effects between different aspects of the ecosystem, which can indicate a possible disturbance of the ecological balance. Such a description of the ecological status can be obtained by applying an integrative matrix system (Table 7-2). The general method of using matrices as assessment tools is described more in detail in the report of the UFOPLAN 3711 25 216. This method has a strong focus on cumulative effects. Interaction effects between stressors as well as interactions between organisms and even between social-economic aspects can be integrated. The matrix consists of all species and species groups considered for the assessment as well as of measures for the ecosystem structure, ecosystem function, biodiversity, and eventually social-economic aspects. Moreover, cumulative effects such as interaction effects, temporal, and spatial effects are closely linked to the results of the indicators and criteria, which define the aspects of the ecosystem the integrative matrix is based on.

7.3.4.1 Ecological aspects used in the integrative matrix

Species and species groups

Four different aspects contributing to the status of each species/ species group are considered in the matrix. These groups comprise the aspects habitat size, habitat quality, overall health of the organism and population size. The terms habitat size and habitat quality refer here to the traditional definition of habitat with regard to a certain species or species group as discussed for example in Nehring and Albrecht (2000). The motivation for using this categorisation was the consideration that interactions between the different components of the matrix often occur between these specified aspects. Two species might for example compete for space, which would be indicated by an interaction factor between habitat size of species A and habitat size of species B in the matrix. However, also ‘cross-interactions’ are possible: the population size of one species might positively influence the habitat size of another species, for example by the provision of a suitable substrate to settle on. The overall health of a species can affect the health of its predator negatively and the population size of a certain species in a habitat of another species might significantly increase the habitat quality, for example by provision of visual structures serving as protection from predators. The variety of interactions between species and habitats aspects is huge. Besides these kinds of interactions, some aspects might interact on a higher level: for example, the population sizes of some key species are assumed to interact with the ecosystem structure as a whole.

It is essential that each interaction and each status of the ecological aspect is described in a short text, so that it is possible to evaluate if an interaction really occurs in a certain scenario. This is particularly relevant for indirect interactions: If one species is e.g. poisoned by some chemical substances affecting its overall health this might affect the predator significantly; however, if the impairment of overall health of the species is affected by genetic drift, this kind of reduction of overall health might be a negligible impairment of health from the predators´ perspective and no actual interaction might occur between these two species regarding health.

Species habitat size (distributional range/ area covered by the species)

The actual habitat size of the species needs to be provided by descriptor 1. For mobile species it is appropriate to use the distributional range, whereas for sessile species it is more reasonable to use the area covered by the species as a measure. The unit used for calculations is any appropriate measure of area. The cumulative analysis sums up the area covered by structures constructed by human for different purposes. Furthermore, anthropogenic pressures, which spread spatially and cause a complete avoidance of a corresponding area (possibly with certain stressor intensity), are considered as well. One example for such an anthropogenic pressure is noise. It should be kept in mind that spatial restrictions of habitat due to anthropogenic pressures cannot always simply be added up because some of some spatial overlaps which need to be taken into account. Programs such as the Geographic Information Systems (GIS) can be used to conduct such kind of calculations (esri.com). The value for the habitat size used in the matrix is either a measure of area or volume. This distinction is also made in the MSFD and depends on the species of interest.

Species habitat quality (distributional pattern, where appropriate; further variables influencing habitat quality)

The factors determining the species habitat quality are very diverse and depend on the species-specific requirements. The distributional pattern is one factor indicating habitat quality because the probability to find individuals at a spot of high habitat quality is higher than at a spot of lower habitat quality. Moreover, at sites of high-quality habitat the population density is probable to be higher than at low habitat quality sites. If such information is available, this way characteristics of the habitats used can be derived. In a multivariate statistical analysis these habitats can be categorised by defining the distributional pattern or presence / absence data as a variable, which is influenced by a set of habitat characteristics considered probable to define habitat quality. Such a method was for example applied to define the relevant factors for presence/ absence data of coral reef fishes using generalized linear models (GLM) and multivariate statistics (Harborne et al. 2011). In a statistical test the most relevant characteristics with the highest explanatory power can be identified (Harborne et al. 2011) and could also be utilized for the characterisation of sites, which might be suitable as potential habitat sites.

Instead of using the distributional pattern or abundance data as an indicative variable for habitat quality other measures can be applied depending on the suitability for the respective species. For some sessile species habitat quality might for example be indicated by growth performance as described by Berglund et al. (2012). For some species it might be more appropriate to use a combination of several parameters as an indication for good habitat quality. A condition index, growth performance of otolithes and RNA/ DNA ratio was for example used for determining nursery habitat quality for plaice (Selleslagh and Amara 2013).

Explanatory variables can be physical, hydrological and chemical conditions but also biological factors such as species composition, abundance data or morphological characteristics of occurring species.

If characteristics, which determine good habitat quality, are well known and if their respective relevance can be estimated, habitat quality can be determined directly by the application by applying multivariate statistics. It is difficult to provide a detailed general concept for estimating habitat quality, which can be applied for any species due to the huge variety of relevant factors and the different level of information available. Therefore, these examples can only serve as proposals for possible approaches.

The indicator 'distributional pattern within the latter (the distributional range), where appropriate' might also indicate habitat quality more directly: Fragmentation and thus the distributional pattern might play a major role for some species and relativize the habitat size. The distributional pattern of a species or the patchiness of its habitat is not easy to determine. However, Kefi et al. 2007 developed a method for indexing such patterns in a geographical analysis by laying randomly lines over the areas and analysing the pattern on these lines, representing the discontinuity of the habitat. Suitable habitat might for example be coloured black and unsuitable habitat might be coloured in grey. The number of discontinuities is used as a basis for calculating an index for patchiness. The index would be normalized according to the requirements of the species, so that it gets a value between 0 (bad) and 1 (good). The single aspects of habitat quality could be added up for each species for the index of habitat quality, which can be used in the integrative matrix later on.

Overall health of the species/ condition of the occurring populations

Overall health is a wide-ranging notion comprising many characteristics. We use this term here to pool all aspects mentioned in the MSFD referring to the condition species. This aspect is described in the MSFD by several indicators referring to different species groups with a special emphasis on the health of fish stocks and shellfish comprising the following indicators:

- 3.2 Reproductive capacity of the stock (referring to fishes)
- 3.2.1 Spawning Stock Biomass (SSB) (primarily indicator for the reproductive capacity of the stock)
- 3.2.2 Biomass indices (secondary indicators (if analytical assessments yielding values for SSB are not available)
- 3.3 Population age and size distribution (referring to fishes)

Primarily indicators. Healthy stocks are characterized by high proportion of old, large individuals. Indicators based on the relative abundance of large fish include

- 3.3.1 Proportion of fish larger than the mean size of first sexual maturation
- 3.3.2 Mean maximum length across all species found in research vessel surveys
- 3.3.3 95% percentile of the fish length distribution observed in research vessel surveys

Secondary indicator

- 3.3.4 Size at first maturation, which may reflect the extent of the undesirable genetic effects of exploitation

The MSFD focuses furthermore on the health of the macrobenthic community as emphasised by the indicators

- 6.2 Condition of benthic community

and

- 6.2.3 Proportion of biomass or number of individuals in the macrobenthos above some specified length/ size

However, two indicators of D1 (biodiversity) also refer to the health of a species or in other words to the condition of the population, which consider all species groups:

- 1.3.1 Population demographic characteristics (e.g. body size or age class structure, sex ratio, fecundity rates, survival/ mortality rates)
- 1.3.2 Population genetic structure, where appropriate

The condition of each species or species group does not need to be described by the same number of aspects and might depend on the focus of the MSFD as well as on species characteristics. A description of an age class structure is for example not relevant for all species groups mentioned in the MSFD. Moreover, the indicative list of characteristics, pressures, and impacts points out health issues and contamination of biota (MSFD, 2008/56/EG). Such health issues might not only affect the species themselves but also the consumers of those species. Contaminants will accumulate in the food web and also affect human health as a consequence.

The same applies to some diseases, which can be transmitted to other species through the food web or through physical contact affecting possibly several species, which interact with it (Smith et al. 1978).

All aspects of the health of a species change over time. Contaminants accumulate in the body, the genetic structure changes slowly and also demographic characteristics change over time. Such temporal models already exist and do not need to be new invented: ICES uses a predictive modelling approach for the calculation of the maximum sustainable yield for fisheries management (SISAM 2012), where demographic characteristics are modelled as well; the dynamic energy budged model (DEB model) (Jager et al. 2010) is used to model maturation, size and contaminant accumulation in the body of organisms and has been modified for the use with regard to the MSFD (see report UFOPLAN 3711 25 216 for details). Other population characteristics such as genetic uncertainty, resilience and demographic uncertainty can be modelled with population viability analyses (PVA) (Morris and Doak 2002). The modelling in a temporal dimension is critical for this concept because the concept is based on a predictive approach.

Population sizes, abundances or biomass

This group of indicators describes the measures of the sizes of populations. Depending on the organism group certain measures are typically used for indicating population sizes. The main indicator for describing this aspect is mentions in the indicator

- 1.2.1 Population abundance and/ or biomass, as appropriate

Special emphasis is placed in the MSFD on key species (4.1 - 4.3), sensitive and tolerant species (6.2.1), macrobenthos (6.1.1), algae (5.2.1, 5.2.3) and perennial seaweeds and seagrasses (5.3.1). Partly those species groups are used as indicators for anthropogenic pressures. It should be kept in mind that those species groups are particularly relevant; however, it should be avoided to integrate these measures twice in the overall assessment (e.g. in indicator 1.2.1 and an additional one such as 5.3.1).

Habitats

According to the MSFD different habitat types need to be defined, which refer to the biotope as a whole including abiotic characteristics as well as characteristics referring to the biological community (MSFD 2008, 2880/56/EC). It would be reasonable to use the structure of the classification of habitats as described in the Commission Decision (2010/477/EU) also for the integrative matrix. Therefore, the following subgroups for each of the habitats are proposed for the matrix with regard to the Commission Decision (2010/477/EU) and considering their relevance for interactions effects: Habitat distributional range, habitat distributional pattern, habitat extent, and habitat condition. The habitat distribution is divided into distributional range and pattern because habitat fragmentation might for example be relevant for some benthic organisms. These single aspects can be added up for each habitat type. It can be considered if a weighting of habitat type should be applied according to conservation considerations such as rarity. Such a weighting of habitats though can first be considered after the main matrix calculations are conducted.

Ecosystem structure

The ecosystem structure is described In the MSFD as the 'composition and relative proportions of ecosystem components (habitats and species)'. The integrative matrix might cover this aspect by its structure itself; however, some species have a strong influence on the ecosystem structure as a whole and these interactions should be covered with the matrix method as well. Some alien species might for example influence the ecosystem structure dramatically, which needs to be considered in the assessment. One possibility for the evaluation of ecosystem structure is calculating an index for ecosystem structure by carrying out a network analysis based on the indicators (see report UFOPLAN 3711 25 216 for details).

Ecosystem structure can be described with the indicator 1.7.1 Composition and relative proportions of ecosystem components (habitats and species) and a food web analysis based on the indicators and species groups mentioned under descriptor 4. The influences of species on the ecosystem structure as a whole can just be seen as an overall measure of the magnitude of alteration of the ecosystem. Specific interactions can rather be covered with the integrative matrix. Influences of species on the whole ecosystem should be described in an explanatory text for transparency.

Ecosystem function

Ecosystem functions such as productivity have a high relevance for all ecological components. This field of research has gained much attention particularly during the last years and it can be assumed that many interactions referring to this concept can be found in the literature (Reiss et al. 2009). As ecosystem functions can be subdivided into many aspects, a selection of ecosystem functions, which should be used in the integrative matrices, needs to be made and the groups need to be chosen carefully. The following ecosystem functions are considered as particularly relevant in the context: gas regulation, climate regulation, disturbance prevention, productivity, nutrient cycling, filtration, biodiffusion, and bioturbation (see e.g. Norling et al. 2007, Hiddink et al. 2009). However, this list only serves as a first proposal and the selection requires a more detailed literature search.

Biodiversity

This aspect covers the classical biodiversity indices such as the Shannon Weaver index, Simpson or Richness (reviewed in Magurran 2004). There might be some overlaps concerning the contents with the aspect Ecosystem structure. However, the focus of the aspect Ecosystem rather lies on the composition of the ecosystem, whereas the aspect biodiversity stresses the number of present species present.

The integrative matrix system

The results of the different aspects can be combined in an integrative matrix (Leopold matrix) (Leopold et al. 1971). This allows taking into account all kinds of interactions between species such as competition advantages and disadvantages as well as the importance of the different ecosystem components on the ecosystem structure, ecosystem function, and biodiversity (Table 7-2). Optionally, even social and economic influences could be integrated.

The matrix analysis is conducted twice: One integrative matrix represents the present status of the ecosystem, whereas the other one represents the system subtracted by the influences of anthropogenic pressures. This implies that the percentage influence of anthropogenic pressures first needs to be calculated for each of the aspects described above. The cumulative analysis provides a method for estimating these values. However, the data for each of the aspect need to be derived by combining the cumulative analyses with the data of the indicators. At the current stage the estimation of the percentage of cumulative human influence can just be shown exemplary and needs to be elaborated more in detail and transferred to all aspects and species.

An interaction factor is calculated in each cell of the integrative matrix (Table 7-2) representing an increase or decrease of the aspect in the column per a certain time period. Moreover, the relationship might depend on the values of the aspects where required. Complementing the interaction factor with a short explanatory text can be very helpful for providing transparency and to understand the ecological system in depth. The interaction factor should be based on literature data wherever possible. In the simplest case a linear relationship defines the interaction between two components, where the interaction factor describes the slope of the relationship. However, those relationships are usually time-dependent, which should be included in the equation by introducing the factor t, representing a certain period of time. However, it needs to be kept in mind that it must refer to the same unit and value in the whole matrix.

The matrix is read column by column from the left to the right. Each cell of the column shows the magnitude of the influence the aspects presented in the lines has on the aspect of the column (highlighted in grey in Table 7-2). In the 'balance line' the cumulative effect based on the interactions shown in the corresponding cells of the column are summed up for each aspect. The actual interaction is not only dependent on the interaction factor but also on the size of the two ecological aspects: A higher number of individuals will for example have a higher influence on a certain habitat type. Therefore, the interaction factor needs to be weighted by the corresponding environmental conditions of each aspect by multiplying the value of the aspect referring to the present environmental conditions. In contrast to some other matrix systems described in report UFOPLAN 3711 25 216, no normalisation needs to be conducted because the matrix of the present system is divided by the matrix subtracted by the percentage of negative human influence including the same aspects and thus resulting in a scalar value as a final assessment value. Alternatively, the matrix of the present status can be compared with a matrix based on the ecological goals described in the MSFD (2008/56/EU) and specified in BLMP (2012a and b).

Table 7-2: Integrative Matrix including the aspects species' habitat size (HS), species' habitat quality (HQ), overall health/ population condition (S), habitat distributional range (Hdr), habitat extent (He), habitat condition (Hc), ecosystem structure (ES), ecosystem function (EF), productivity (pro), climate (cli), and biodiversity (B).

		Species A (Sa)				Species B (Sb)				Habitat 1 (H1)			Habitat 2 (H2)			Ecosystem structure (ES)	Ecosystem function (EF)			Biodiversity (B)
Effects on / Influence of		HS	HQ	S	P	HS	HQ	S	P	Hdr	He	Hc	Hdr	He	Hc		pro	cli	…	
Species A	HS																			
	HQ																			
	S																			
	P																			
Species B	HS																			
	HQ																			
	S																			
	P																			
Habitat 1	Hdr																			
	He																			
	Hc																			
Habitat 2	Hdr																			
	He																			
	Hc																			
Ecosystem structure																				
Ecosystem function	pro																			
	cli																			
	…																			
Biodiversity																				
Interaction index · values																				
SIN		$\mathrm{HS_A} \cdot \mathrm{HQ_A} \cdot \mathrm{S_A} \cdot \mathrm{P_A}$				$\mathrm{HS_B} \cdot \mathrm{HQ_B} \cdot \mathrm{S_B} \cdot \mathrm{P_B}$				$\mathrm{Hdr_1} \cdot \mathrm{He_1} \cdot \mathrm{Hc_1}$			$\mathrm{Hdr_2} \cdot \mathrm{He_2} \cdot \mathrm{Hc_2}$			ES	$\mathrm{pro} \cdot \mathrm{cli} \cdot \ldots$			B
Final index	$\sum_{i=1}^{S}(\mathrm{HS_i} \cdot \mathrm{HQ_i} \cdot \mathrm{S_i} \cdot \mathrm{P_i}) + \sum_{i=1}^{H}(\mathrm{Hdr_1} \cdot \mathrm{He_1} \cdot \mathrm{Hc_1}) + \mathrm{ES} + \mathrm{EF} + \mathrm{B}$																			

The results of these equations should be verified with literature data. In some cases, such concrete literature data might not exist even though the relationship between to ecological components can be derived with a relatively low risk of uncertainty. For example, a relationship between a certain species and the size of a certain habitat type might be likely to be similar to a relationship between a similar species inhabiting the same ecological niche inhabiting the same habitat type. In such cases expert knowledge can be a good option and the overall uncertainty is likely to be smaller than in case of integration of an uncertainty factor for the lack of knowledge, which is difficult to estimate as well. However, the cells, where an expert judgment is used should be counted and allow to evaluate the overall uncertainty, which is added by the integrative matrix.

In other cases, though the relationship might be very well known, a lot of literature data exist and the relationship can be described in a precise equation. The flexibility of the matrix allows using such detailed information as well as approximate data (Dixon and Montz 1991). However, the equation might need to be transformed because the result in the cell should always be a percentage decrease or increase under certain environmental conditions even if the value varies depending on factors used in the equation. These factors can include any relevant environmental conditions but also e.g. density related measures of the influencing aspect. Calculations can be conducted in the program Matlab, which has a strong focus on matrix calculations and facilitates such methods; but also other programs for mathematical calculations might be applicable.

In the 'balance line' all interactions are summed up resulting in the 'netto effect' of all influences. If the environmental data referring to the aspect in the column are added to this value, the expected environmental data for this aspect for the next time point at the end of a certain time period can be calculated. Based on this principle, the expected values for each ecological aspect can be calculated for a certain time in the future. It should be assured that all ecological aspects change at each time period, wherever they are used in the matrix. How far into the future the system should be modelled depends on the data availability and the resulting uncertainty as well as on the political need for using a certain time frame.

As explained above an equivalent matrix could be constructed for the situation without the influence of anthropogenic pressures. This matrix represents a theoretical recovery potential of the ecosystem. However, it needs to be considered that in reality it is not possible to release the marine environment from all negative human influences anymore because e.g. certain substances are not possible to eliminate and some alien species are likely to remain in the system for a long time. However, especially long-lasting anthropogenic pressures should not be denied, because they can have relevant effects on the ecosystem. On the other hand, a system with a bad ecological quality might have a small recovery potential and the ecological status might be overestimated. Therefore, the comparison of the matrix modelled into the future assuming the same intensity of anthropogenic pressures as in the assessment year with a matrix representing the ecological goals might be a more suitable approach and fits better to the approach of the MSFD.

The relation between the results of the two matrices indicates the final assessment value of the overall assessment referring to a certain area. These data can be used for constructing a map as well as the other methods for an overall assessment proposed above.

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