FEHMARNBELT MARINE BIOLOGY
FEHMARNBELT HYDROGRAPHY

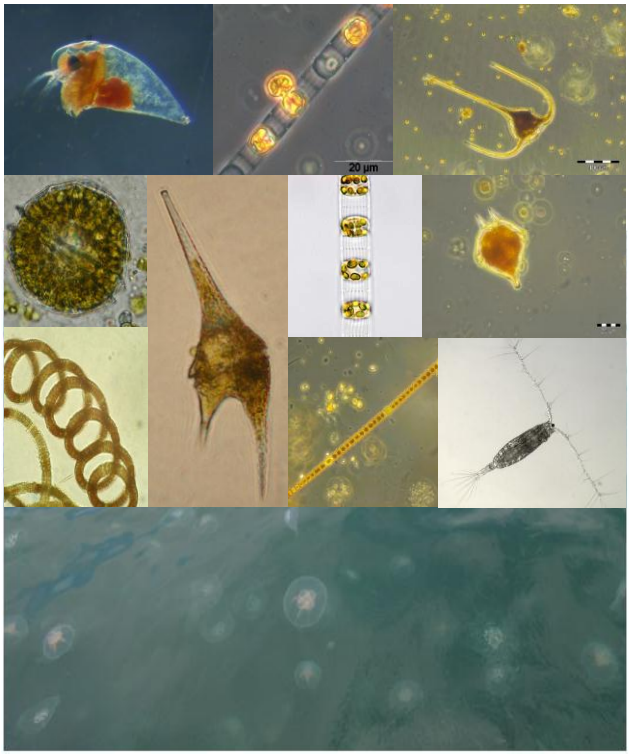
Final Report

FEHMARNBELT FIXED LINK
MARINE BIOLOGY SERVICES (FEMA)
HYDROGRAPHIC SERVICES (FEHY)

Marine Water and Fauna and Flora Impact Assessment

Water Quality and Plankton of the Fehmarnbelt Area

E2TR0021 - Volume III



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ACRONYMS, ABBREVIATIONS AND GLOSSARY

Algae blooms	Fast growth and huge increase in the density of phytoplankton on the surface of lakes and marine waters caused by nutrient enrichment and weather conditions
Biomass	The total amount in weight (wet, dry, ash free dry weight) of living organisms in a given area or volume
Cd	Cadmium – heavy metal
Climate change	Long-term changes in air, soil, and water temperature; precipitation regimes; wind speed; or other climate-related factors such as sea level rise
Biological community	Populations of various species which are co-occurring at the same time and place
Blue-green algae	See cyanobacteria
Chlorophyll-a	Light-capturing pigment present in all phototrophic plants. Often used as a surrogate for biomass in phytoplankton; Weight-ratio between chlorophyll-a and phytoplankton carbon average 1:30-40
Co	Cobalt – heavy metal; occurs in the aquatic environment in two oxidation stages (Co III and Co VI)
Cu	Copper- heavy metal
Competition	Biological interaction occurring when the demand for an resource exceeds its supply, causing organisms to interfere with each other.
Cyanobacteria	Phototrophic bacteria often dominating "phytoplankton" community in eutrophic freshwaters. Also called blue-green algae
Diatoms	Class of diatom cells encased within a cell wall made of silica (hydrated silicon dioxide). Diatoms are a dominating group within phytoplankton, often dominating during spring and autumn. Diatoms are favored by high nutrient concentrations and because of low light requirements they are favoured by vertical mixing
DDT	An organochloride used as an insecticide. Has been banned since 1969 in most western countries because of its bioaccumulation potential, persistence, toxicity and carcinogenic effects.
EQS	Environmental Quality Standards developed by EU; refers to List 1 substances to protect the environment. For each substance, aquatic EQS values exist for continuous exposure (annual average), and short-tern exposure (max.)
Eutrophication	Over-enrichment of a water body with nutrients, resulting in excessive growth of organisms (mainly phytoplankton and annual macroalgae) and depletion of oxygen concentration.
Habitat	The geographical location(s) and the associated set(s) of environmental conditions, that are necessary for the living of a particular type of plant or animal or a group of organisms.
HCB	Hexachlorobenzene is a chlorocarbon (C_6Cl_6). It is a fungicide previously used to treat seed of especially wheat. Banned due to persistence and toxicity.
Heavy metal	Metals (elements) with atomic no. larger than 40

Hg	Mercury - heavy metal
Holozooplankton	Zooplankton organisms that spend their entire life cycle in the water colum
Hypoxia	Reduced concentration of dissolved oxygen (compared to full saturation) in water which can be detrimental to aquatic organisms
Inorganic nutrients	Dissolved salts containing nitrogen, phosphorus and silicon. Inorganic nutrients (NO_2^-, NO_3^-, NH_4^+, PO_4^{3-}) are essential for all algae and higher plants; SiO_2^- is also essential for diatoms.
Light absorption	Attenuation of light due to absorption by organic material (dissolved, particulate, dead or living)
Light attenuation	Reduction of light due to combined effects of scatter and absorption
Light scatter	Scatter of light by particulate material (organic or inorganic) in the water column
List 1 substances	13 substances that are priority substances and priority hazardous substances (toxic, persistent and likely to bioaccumulate). Regulated under Water Framework Directive. Environmental Quality Standards have been set to protect environments
List 2 substances	20 substances that are priority substances (not necessarily hazardous) regulated under the Water Framework Directive
Meroplankton	Larvel forms of benthic invertebrates that live in the water column for a short period (1-8 weeks)
Ni	Nickel – heavy metal
Nitrogen fixation	The natural process, either biological or abiotic, by which atmospheric nitrogen (N_2) is converted into ammonia (NH_3). In the aquatic environment, cyanobacteria are the main responsible for nitrogen fixation. The capacity is particularly important in freshwater and brackish waters, because large colony-forming cyanobacteria species cannot life at salinities higher than 10 ppt.
NTU	Nephelometric turbidity units, refers to the way a nephelometer, measures how much light is scattered by suspended particles in the water. Often measured by an optical back-scatter (OBS) intrument
Oxygen deficiency	Reduction in oxygen concentration below saturation (determined by salinity and temperature) in near-bed waters. Deficiency is a result of respiration by bacteria, plants and animals. Short-term oxygen deficiency can develop if reduced substances are released momentarily e.g. by dredging.
Oxygen demand	The capacity of organic matter and other substances in water to consume oxygen during decomposition or chemical oxidation (e.g. $H_2S \rightarrow H_2SO_4$)
PAHs	Polycyclic aromatic hydrocarbons are potent atmospheric pollutants. PAHs occur in fossil fuels and are produced as byproducts of fuel burning (whether fossil fuel or biomass). They are of concern because some compounds are carcinogenic.
Pb	Lead - heavy metal
PCB	Polychlorinated biphenyl covers a group of more than 200 compounds with 2-10 Cl atoms on the compound core consisting of two benzene rings. Previously used as dielectric in transformers and capacitors. Banned because of persistence, bioaccumulation and toxicity

Pesticide	A substance or mixture of substances intended for preventing, destroying, repelling, or mitigating any pest. Persistent pesticides can end up in marine sediments after aerial or water-mediated transports from farmed fields.
Phytoplankton	Microscopic aquatic plants, often unicellular members of the plankton. Main contributors to primary production in lakes and marine waters
Plankton	Organisms which are unable to maintain their position or distribution independent of the movement of water masses.
Primary production	Productivity by autotrophic organisms, such as higher plants or algae. Measured as biomass accumulated over a unit of time (net primary production), or by the amount of carbon fixed, e.g. using labeled CO_2 (gross primary production).
Resuspension	Stirring up of mud or fine sand from seabed due to wave or current action
Runoff	Storm water from paved areas and adjacent domestic or commercial properties that may carry pollutants of various kinds into the sewer systems and from there to streams and the sea.
Secchi depth	A measure of the clarity of water. Secchi depth is measured using a circular plate, known as a Secchi disc, which is lowered into the water until it is no longer visible
SS	Suspended solids; suspended sediments
TBT	Tributyl Tin; organo-metal previously used as antifouling agent on vessels. Very toxic to aquatic life, banned on commercial vessels since 2008
Toxic	Substances which are harmful to living organisms.
TSS	Total suspended solids (sediments)
VKKgen	General water quality criteria (i.e. concentration) set by Danish authorities for metals and compounds to protect aquatic environment. Concentrations averaged over the release periods must not exceed VKKgen
VKKmax	Max acceptable water quality criteria (i.e. concentration) set by Danish authorities for metals and compounds to protect aquatic environment. Max concentration during the release periods must not exceed VKKmax
Water Quality	A collective term used to describe the physical, chemical and biological characteristics of water with respect to its suitability for a particular use, including bathing water, inhabiting of marine organisms
Zinc	Zinc - heavy metal
Zooplankton	Pelagic animals in the water column. Most are grazers on phytoplankton.

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Note to the reader:
In this report the time for start of construction is artificially set to 1 October 2014 for the tunnel and 1 January 2015 for the bridge alternative. In the Danish EIA (VVM) and the German EIA (UVS/LBP) absolute year references are not used. Instead the time references are relative to start of construction works. In the VVM the same time reference is used for tunnel and bridge, i.e. year 0 corresponds to 2014/start of tunnel construction; year 1 corresponds to 2015/start of bridge construction etc. In the UVS/LBP individual time references are used for tunnel and bridge, i.e. for tunnel construction year 1 is equivalent to 2014 (construction starts 1 October in year 1) and for bridge construction year 1 is equivalent to 2015 (construction starts 1st January).

0 EXECUTIVE SUMMARY

The planned fixed link across Fehmarnbelt may affect the water quality and plankton populations during and after the construction phase.

During construction sediment spill will increase turbidity in the water and consequently may reduce the aesthetic quality of bathing water and influence the light available for photosynthesis affecting plankton production and biomass. Reduced light at seabed may reduce production of benthic plants which will affect oxygen concentration in bottom waters. If H_2S and toxic substances in dredged sediments are released during dredging and disposal, oxygen levels may be depressed and toxic effects in plankton organisms can be affected.

The construction work will inevitably increase sedimentation of spilled sediments that can lead to increased sedimentation of phytoplankton by flocculation and bury resting eggs of copepods and potentially affect recruitment of copepods.

The introduction of new structures in the marine environment of Fehmarnbelt may permanently change the hydrography in the area, e.g. by affecting vertical mixing and the water column stratification. Furthermore, bridge pylons and pillars will increase the area of hard surface thus constituting additional substrate for blue mussels, macroalgae and ephyra of jellyfish with subsequent effects on the water quality and the plankton communities.

The position of water quality, plankton and jellyfish in the Environmental Impact Assessment framework is shown in Table 0.1. The sub-factor components concern marine water quality, bathing water quality and phytoplankton, zooplankton and jellyfish biology.

Table 0.1 The environmental sub-factors and components assessed in this report.

Environmental factor	Environmental sub-factor	Environmental component
Water	Marine Water	Marine Water Quality Bathing Water Quality
Fauna, flora and biodiversity	Marine flora and fauna	Phytoplankton
		Zooplankton
		Jellyfish

The components of marine water and marine plankton include the subcomponents listed in Table 0.2. The impact assessment has been carried out on the scale of the components and/or sub-components.

In this report the time for start of construction is artificially set to 1 October 2014 for the tunnel and 1 January 2015 for the bridge alternative. In the Danish EIA (VVM) and the German EIA (UVS/LBP) absolute year references are not used, but instead the relative time references from start of construction works (year 0, year 1, etc.), i.e. year 0 corresponds to 2014; year 1 corresponds to 2015 etc.

Table 0.2 The sub-components included in water quality and marine plankton assessment and the potential environmental effect of changes

Environmental sub-component	Environmental effect of changes
Water Quality – including bathing water	
Nutrients	Affects eutrophication
Suspended solids	Affects turbidity / Secchi depth / bathing water
Light penetration	Affects light availability to benthic plants and aesthetic quality at beaches
Oxygen (near seabed)	Affects benthic organisms
Toxic substances	May affect plankton and benthic organisms
Plankton Biology	
Chlorophyll-a	High concentration can be a sign of eutrophication
Phytoplankton production, concentration and composition	Affects pelagic and benthic food webs
Bloom of cyanobacteria	Potential toxic, affects food webs
Zooplankton production and biomass	Important to planktivorous fish
Copepod resting eggs	May affect recruitment of copepods
Jellyfish abundance	Affect food webs (competes with fish larvae)

The specific objectives of the water quality and plankton biology impact assessment have been to:

- Predict the degree and severity of impact on water quality and plankton from temporal activities of dredging and sediment spill during construction of the Fehmarnbelt Fixed Link
- Predict the degree and severity of impact on water quality and plankton of permanent changes in the hydrography and by introduction of new structures during the operation phase of the Fehmarnbelt Fixed Link
- Assess the significance of the predicted impacts for water quality and plankton
- Compare the impacts on water quality and plankton of the bridge and the tunnel alternatives

Methods

The basis for the impact assessment is the water quality and plankton investigated during the baseline investigation. The basis for determining the range, duration and intensity of the pressures is the project description, modelled sediment spill and hydrography and available literature data. The impacts are predicted using hydrodynamic and ecological modelling, quantitative assessment and expert knowledge. The area assessed is shown in Figure 0.1 and encompasses the Fehmarnbelt, Rødsand Lagoon, and the adjacent waters; Great Belt in the West and Mecklenburg Bight and the western Baltic Sea in the East.

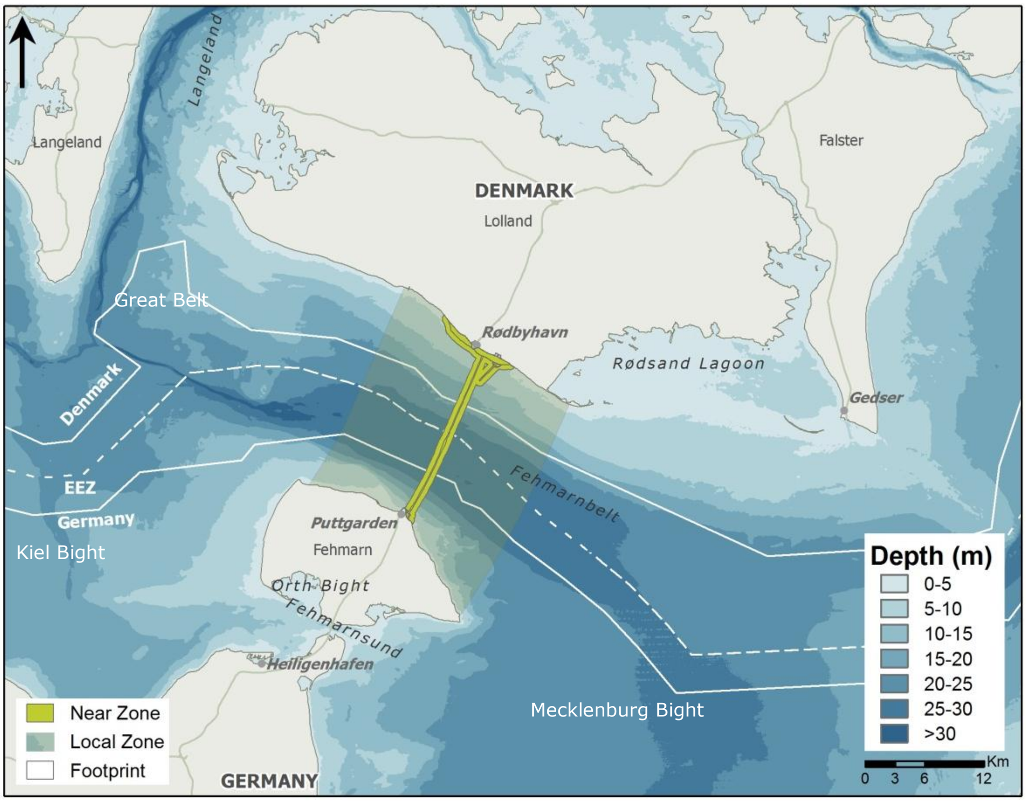


Figure 0.1 Assessment area and defined sub-regions for the tunnel and bridge alternative.

The assessment methodology relies extensively on dynamic models, including Hydrodynamic Models, Sediment Model and Water Quality Models. In general, most steps in the impact assessment are an integral part of models:

- Important pressures related to construction and operation period of tunnel and bridge (e.g. concentration of spilled sediment) are modelled dynamically in 3 dimensions; concentrations of spilled sediments are used to calculate light attenuation dynamically (i.e. dose-response between sediment concentrations and light attenuation - "Sensitivity"), which in turn affects the growth of phytoplankton (i.e. dose-response between light intensity and growth rate - "Sensitivity") and biomass (i.e. impact of dredging on phytoplankton - "Impairment"), benthic vegetation biomass (impact of dredging on seagrass and macroalgae - "Impairment") and indirectly, oxygen concentration at seabed (impact of dredging on water quality - "Impairment").
- Degrees of impairment are averaged over appropriate periods and the "Severity" of impairment is assessed using a 4-level criteria scale "Very high", High, "Medium", "Minor" defined by degree of deviation from baseline conditions after taking account of natural year-to-year variation. Impairments below "Minor" are considered as "Negligible").
- "Significance" of impairment is assessed by combining Degree or Severity of Impairment with area extension of impairment.

For potential impacts that cannot be modelled directly, i.e. when dose-response relationships are less well documented, appropriate model outputs are overlaid (time-step by time-step), to identify areas and duration where and when 2-to-several criteria is fulfilled. In the assessment of dredging-related impacts direct effects of suspended sediments on phytoplankton sedimentation and on zooplankton growth and survival such approaches are used.

Release of toxic substances and oxygen demand during dredging are assessed using Monte-Carlo analysis based distribution functions of dredging spill, of toxic concentration in sediments, of release rates and of current speed (i.e. dilution). Calculated concentrations are compared to EU Environmental Quality Standards (EQS), German standards and Danish Water Quality Criteria (VKK). For oxygen, calculated oxygen demand are subtracted from background concentrations and the resulting concentration compared to internationally accepted criteria for minor, high and very high levels of oxygen deficiency (4 mg O_2/l, 2 mg O_2/l, 1 mg O_2/l).

Main pressures

Six types of pressures are relevant for water quality and plankton biology. They include 4 temporary pressures related to construction works:

- Increased concentration of suspended sediments (spill from dredging operations) influencing light penetration into the water column (Secchi depth), that in turn affects primary production, phytoplankton biomass and composition and, zooplankton production. Besides, high concentration of suspended sediments can lead to increased (facilitated) sedimentation of phytoplankton by flocculation and high turbidity along beaches caused by high concentrations of suspended sediments constitutes an aesthetical nuisance for bathers.
- Increased sedimentation of suspended sediments can bury resting eggs of copepods and potentially affect recruitment of copepods affecting the composition of zooplankton community.
- Release of toxic substances during dredging operations potentially harming plankton organisms.
- Increase of oxygen demand caused by reduced substances released during dredging
- Discharge of pollutants under both temporary and permanent nature

Two relevant pressures are of permanent nature (operational phase). They include:

- Changes in hydrography, especially changes in vertical mixing across pycnocline affecting strength and duration of water column stratification. Changes in vertical mixing can affect risks for blooms of cyanobacteria and also change the oxygen ventilation of near-bed water.
- Bridge pylons and pillars will increase the area of hard substrate thus constituting additional substrate for blue mussels, macroalgae and ephyra of jellyfish. Mussels will remove part of phytoplankton passing between pillars, macroalgae growing below pycnocline can add to oxygen content in water passing pillars and higher abundance of ephyras may increase recruitment of jellyfish that is regarded as a nuisance in the western Baltic Sea.

Dredging volumes are about 10 times larger for the immersed tunnel than for the bridge alternative, which implies that pressures overall will be much larger. However, locally in vicinity of a dredger magnitude of some pressures such as increase in concentration of toxic substances and release of oxygen demand will be similar for the two link alternatives. Impacts of these pressures are assessed commonly for the two alternatives.

Toxic substances

Assumptions for calculating the risk for exceeding EQS for heavy metals and persistent organic pollutants are listed in Table 0.3.

Table 0.3 Overview of assumptions used in Monte-Carlo analysis of risks for exceeding EQS for heavy metals and POP's during dredging in Fehmarnbelt

Activity/process	Value (range)	Comments
Dredging intensity	5000 m^3/dredger/d	Fixed (FEHY 2012)
Sediment bulk density	1800 kg/m^3	Fixed (FEHY 2012)
Plume width	25 m	Fixed (very conservative)
Depth	15 m	Fixed (to primary pycnocline)
Sediment spill	0.3-5%	Variable: uniform probability
Tox conc in sediments	Measurement range	Variable: log-normal distribution
Tox release from sediments	Measurement range	Variable: log-normal distribution
Flow velocity	Measurement range	Variable: log-normal distribution

The predicted increase in concentration of a toxic substance (ΔTox) during dredging operations is calculated as:

$$\Delta Tox = \text{Dredging intensity} \times \text{Bulk density} \times \text{Spill (\%)} \times \text{Tox conc} \times \text{Tox release} / (\text{Plume width} \times \text{Depth} \times \text{Flow velocity}).$$

Monte-Carlo analyses were carried out using the assumptions in Table 0.4 and distribution functions for sediment spill, concentration and release rates of metals, and flow velocity, Figure 0.2). The analyses were performed for the metals Cd, Ni and Zn which showed the highest release rates and also for Benz(a)pyrene.

The analyses showed that the probability of dredging activity in the Fehmarnbelt should lead to toxic impacts on the plankton communities, is practically non-existing. For the three metals with the highest mobility (Cd, Ni, Zn) median values (50% percentiles) of predicted concentrations were between 20 and 500 times lower than the EU Environmental Quality Standards and between 5 and 50 times below the more restrictive provisional Danish standards (Figure 0.3, Table 0.4). For zinc, the predicted median concentrations were 50 times below $VKK_{general}$ and the max predicted concentration (99.9 percentile) was 3.5 times below VKK_{max} (Table 0.4).

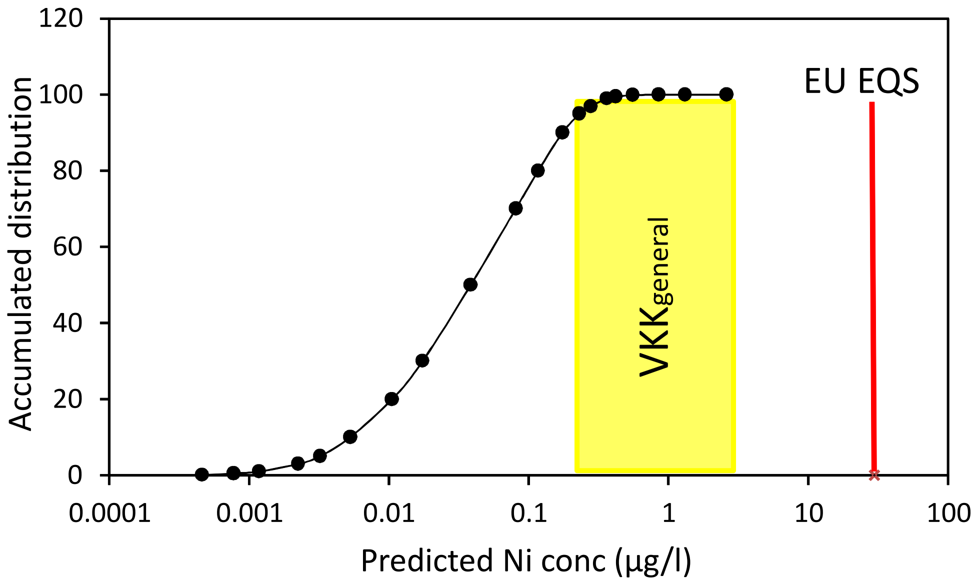


Figure 0.2 Distribution function for predicted concentration of Ni down-stream a dredger extracting 5000 m^3/d. Vertical red line denotes the EU EQS for Ni at 20 µg/l; yellow box the range in Danish provisional EQS ($VKK_{general}$) ranging 0.23-3.0 µg/l (added conc).

For persistent organic pollutants (PAH's, PCB's, DDT, TBT, HCB) and their degradation products risks for exceeding EQS and water quality criteria are even lower than for heavy metals. In a worst case estimate concentration of benz(a)pyrene (B(a)P - PAH compound) was estimated to 0.0003 µg/l which is much lower than the EQS at 0.1 µg/l.

It is therefore concluded that dredging activities will not lead to toxic concentrations of heavy metals or persistent organic pollutants.

Table 0.4 Predicted distributions of increases in concentrations of Cd, Ni and Zn (µg/l) for dredging activities in the Fehmarnbelt. EQS (annual, allowable concentration) are environmental quality standards that protect aquatic life. Predicted median concentrations (50 percentile) are compared to VKKgeneral and EQSannual; and 99.9 percentile are compared to VKKmax.

Percentile	**Cd**	**Ni**	**Zn**
1	0.00006	0.001	0.01
50	*0.0033*	*0.036*	*0.16*
95	0.0092	0.23	0.91
99	0.023	0.36	1.4
99.9	**0.029**	**0.55**	**2.55**
EQS (EU), ann avr.	*0.45*	*20*	
"EQS",VKK_{max} (DK)		**6.8**	**8.4**
"EQS", $VKK_{general}$ (DK)		*0.23-3*	

Oxygen demand

During dredging, part of accumulated oxygen demand in sediments will be released momentarily to the water column where reduced substances such as H_2S will be oxidized consuming part of the dissolved oxygen in water. Eventually, concentration of oxygen may decrease to below critical levels set to protect marine organisms.

Oxygen demand in sediments from the Fehmarnbelt was examined in laboratory experiments as part of the baseline study. Using distribution statistics for organic content, weight-specific oxygen demand, and density of sediments the median oxygen demand (50-percentile) was calculated at 23.25 kg O_2/day at a daily dredging intensity of 5000 m^3/dredger and, 25- and 75-percentiles at 17 and 45.25 kg O_2/day.

Worst-case scenarios of dredging impacts in the spill plume were quantified by combining a high release of oxygen demand (45.25 kg O_2/day) with low current speeds (25%-percentiles: 0.024 m/s below pycnocline; 0.049 m/s above pycnocline), and during periods (late summer) of low oxygen in bottom water (≈ 2.6 mg O_2/l). Four examples (3 worst-case and a central estimate) are shown in Table 0.5.

Table 0.5 Estimated reductions in oxygen concentration in bottom water caused by release of oxygen demand during dredging. Three examples from a hypothetical 18 m station shown; worst-case 1: high oxygen demand, low current speed, 3% spill distributed evenly over the water column; worst-case 2: as worst-case 1 but all spill occurs below pycnocline; median; worst-case for station located above pycnocline.

	Below pycnocline (18 m)			**Above pycnocline**
	worst-case 1.	**worst-case 2.**	**median**	**worst case**
Current speed (m/s)	0.028	0.028	0.063	0.025
Diss. oxygen (g/m^3)	2.6	2.6	7.5	7.5
Oxygen demand (kg/d)	45.25	45.25	23.25	45.25
Plume width (m)	25	25	25	25
Oxygen demand (kg/m/d)	2.51	15.08	1.29	7.54
Depth range (spill)	18	3	18	6
Daily oxygen flux (kg/m)	157	157	1021	405
Oxygen reduction	1.6%	9.6%	0.1%	1.9%

The most critical environment for release of oxygen demand is benthic habitats located just below the pycnocline at 15-18m; benthic organisms are sensitive to hypoxia, current speeds are low and, the height of bottom water (from seabed to pycnocline) is small implying that oxygen content is limited and additional oxygen demand may exhaust the oxygen content.

Worst-case scenario impacts on bottom water oxygen due to release of oxygen demand showed reductions between 1.6% and 9.6% from an already low concentration of 2.6 mg O_2/l. The highest reduction (worst-case 2) will occur if all oxygen demand (and sediment spill) takes place in a shallow (3 m) water column below the pycnocline. Such situations are highly unlikely.

A more likely situation is represented by the "median" case in Table 0-5; concentration of oxygen below pycnocline is close to saturation (as during October through May), current speeds and oxygen demand are at median levels, and sediment spill and oxygen demand are distributed all over the water column. In such situations a reduction in bottom water oxygen of ca. 0.1% can be expected. Overall, the impairment will be negligible both for the tunnel and the bridge alternative.

Assessment of impact of tunnel alternative

Permanent impacts

The submerged tunnel will lead to a loss of pelagic habitats due to land reclamation and protection reefs covering 355 ha of seabed in shallow areas. The loss of pelagic volume can be estimated to ca. 9,900,000 m^3. More than 90% of this volume loss is confined to shallow waters of low importance for plankton. Overall, permanent impacts of the tunnel alternative will be negligible and insignificant.

Temporary impacts

Spill of dredged sediments will affect transparency (Secchi depth) directly, and plankton and oxygen production indirectly.

Secchi depth and bathing water
In baseline conditions the Secchi depth varies spatially in the Fehmarnbelt from 7.0-7.5 m in the central part to less than 2m in the non-vegetated eastern part of the Rødsand Lagoon (Figure 0.3). Low Secchi depths in Rødsand are caused by regular resuspension of the relative fine sediment. In the western part of Rødsand Lagoon Secchi depth is larger because the dense population of *Zostera* supresses resuspension. Scattered areas along the Lolland coast and west of Fehmarn Secchi depth was higher than in the central part of Fehmarnbelt due to filtration by the large population of blue mussels. A high filtration pressure reduces phytoplankton and chlorophyll-a, and thereby reduces the absorption of light.

As a result of dredging in 2014-2015 the Secchi depth will be reduced by up to app. 45% along the Lolland coast in the local area on both side of the alignment and up to 40% in the Rødsand Lagoon owing to sediment spill and spread and resuspension of sediments originating from dredging works.

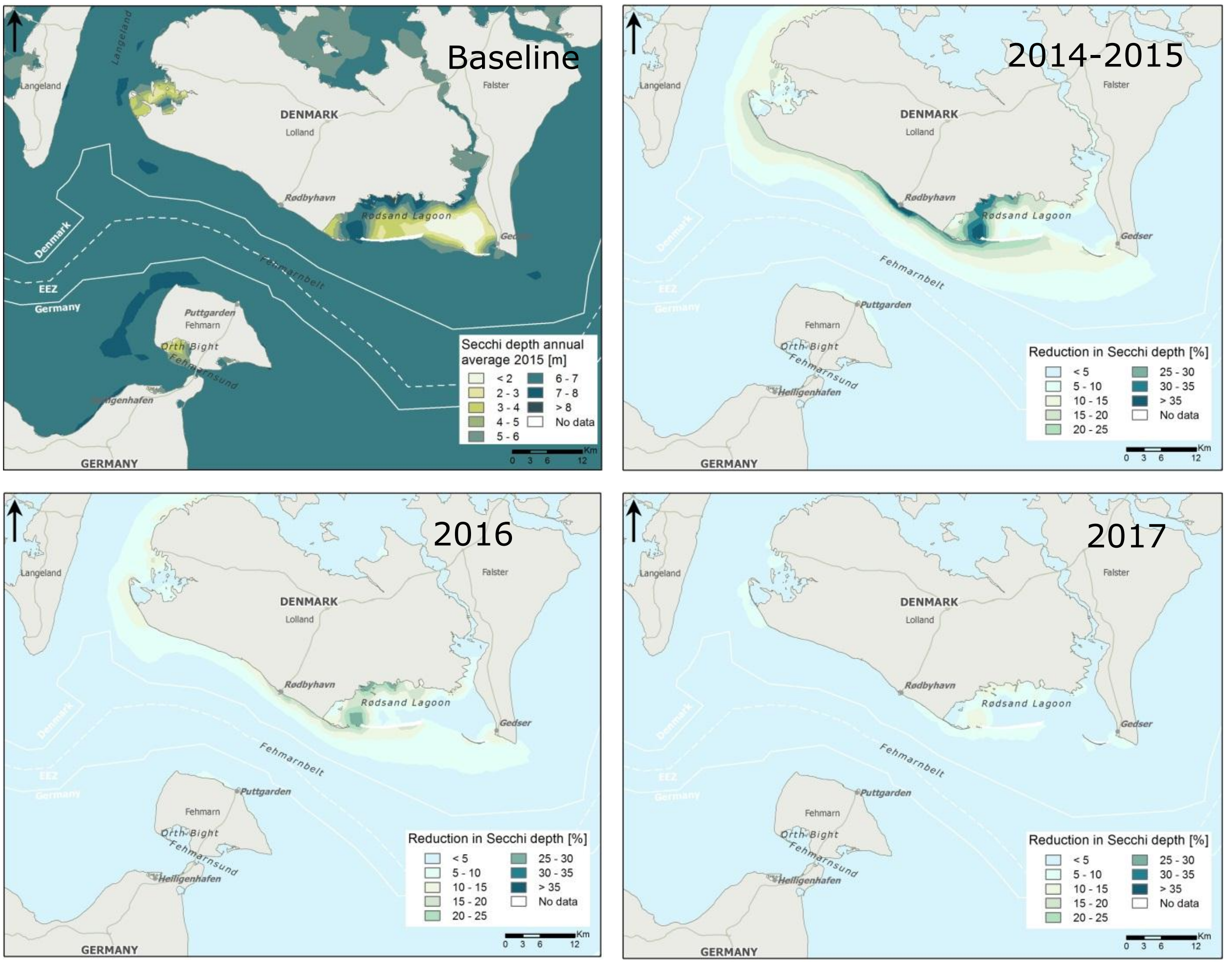


Figure 0.3 *Modelled Secchi depth under baseline conditions (upper left) and %-reduction in Secchi depth in 2014-2015 (dredging works most intense), in 2016 and 2017. Based on average Secchi depths from October 2014 to December 2015, and yearly averages in 2016 and 2017.*

In 2016 the effect on Secchi depth of sediment spill will decrease along Lolland with smaller areas affected. In the Rødsand Lagoon, local reductions of up to app. 32 are observed. In 2017 only the Rødsand Lagoon is affected (maximum reduction locally at 15%) and in 2018 baseline conditions of Secchi depths will be fully restored (not shown).

The reductions in Secchi depth correspond to Minor, Medium, and High degree of impairment in 2014-2015 and 2016 and Minor degree of impairment in 2017 (Table 0.6).

In German waters only Minor reductions (up to 15%) in Secchi depth are predicted in 2014-2015 and 2016. The reductions occur in deep waters outside Puttgarden in 2014-2015 and only west of Puttgarden 2016 (Figure 0.3).

In German waters, reductions in the yearly averaged Secchi depths correspond to Minor degree of impairment in small areas in 2014-2015 (Table 0.6).

Table 0.6 Degree of impairment on Secchi depth (areas in ha) caused by suspended sediments for the tunnel alternative.

	Total	**DK national + EEZ**	**DE national**	**DE EEZ**
2014-15				
Very high	0	0	0	0
High	1975	1975	0	0
Medium	5952	5952	0	0
Minor	35858	35745	113	0
Total	**43785**	**43672**	**113**	**0**
2016				
Very high	0	0	0	0
High	7	7	0	0
Medium	1926	1926	0	0
Minor	15874	15874	0	0
Total	**17807**	**17807**	**0**	**0**
2017				
Very high	0	0	0	0
High	0	0	0	0
Medium	0	0	0	0
Minor	2022	2022	0	0
Total	**2022**	**2022**	**0**	**0**

There are 16 designated beaches along the Fehmarnbelt coasts, 10 on Lolland and 6 on Fehmarn (Figure 0.4). The bathing water quality based on bacteriological status complies with the Bathing Water Directive for all beaches. Although it cannot be assessed in detail, it is not likely that dredging works will affect the bacteriological status of beaches. But it is more likely that reduction in transparency in bathing waters might affect the behaviour of bathers and their choice of beaches.

When construction work is initiated the two beaches near Rødbyhavn, "Lalandia" and "Rødbyhavn at Søpavillon" will be closed because of land reclamation. The remaining 14 beaches will be subject to potential impact from dredging works.

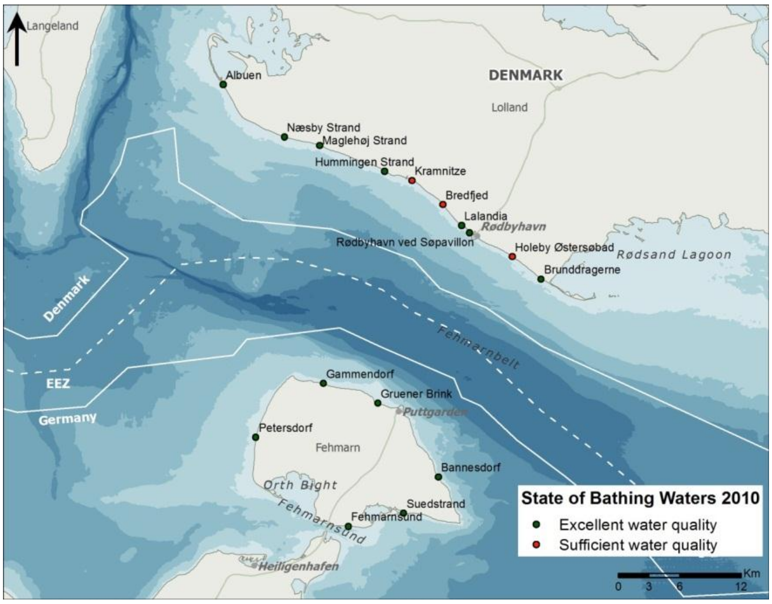


Figure 0.4 *Location of 16 designated beaches with obligatory assessment of bathing water quality and, their compliance with the bathing water Directive. Blue indicates that the water quality is compliant with the guide values of the Directive or excellent water quality for 2010. Green indicates that the water quality is compliant with the mandatory values of the Directive or sufficient water quality for 2010.*

Figure 0.5 shows two examples of temporal variation in Secchi depth for baseline and tunnel-scenario during 2015. A general feature is that reduction in Secchi depth will increase towards the work-areas outside Rødby Harbour, where sediments are spilled. Reductions in Secchi depths in the bathing season 2015 will vary between 16% and 48% at Lolland beaches, while predicted reductions at German beaches will not exceed 1% (Table 0.7).

Based on experiences from New Zealand of bathers' perception of reductions in water transparency and duration in transparency, critical levels of Secchi depth were defined and applied to modelled Secchi depth on beaches in Fehmarnbelt. One beach on Lolland (Bredfjed) will be "minor" affected due to reduction in Secchi depth during the bathing season 2015, when dredging works will be most intense. In 2016 and later, reduction of Secchi depth on Danish beaches varied between 0.6 and 2%, and did not exceed 0.6% on German beaches. Overall, impairment of the aesthetical values of beaches by the submerged tunnel will be insignificant.

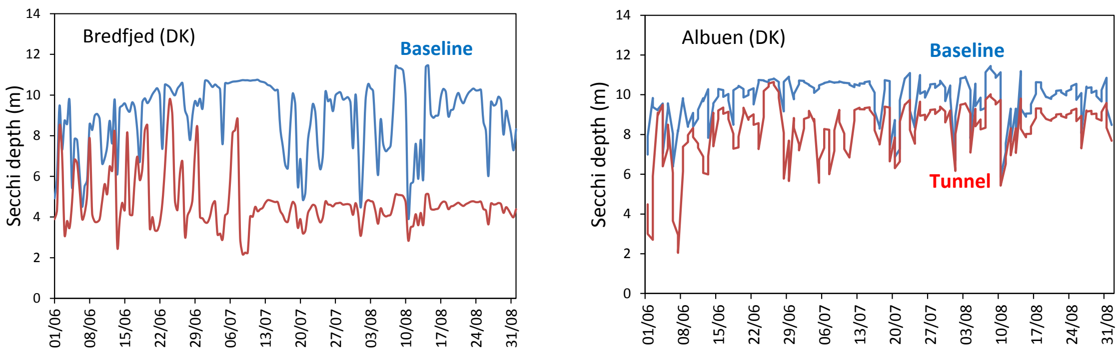


Figure 0.5 *Temporal variation during bathing season (1 June-31 August) in Secchi depth at Bredfjed and Albuen beaches on Lolland in 2015. Baseline and tunnel scenarios shown.*

Table 0.7 Predicted status of 14 beaches (based of water clarity) in 2015 and 2016. Values in brackets are reductions (%) in Secchi depth compared to baseline

Beaches	**2015**		**2016**
Albuen (DK)	negligible	(-16%)	negligible
Næsby Strand (DK)	negligible	(-17%)	negligible
Maglehøj Strand (DK)	negligible	(-20%)	negligible
Hummingen Strand (DK)	negligible	(-18%)	negligible
Kramnitze (DK)	negligible	(-27%)	negligible
Bredfjed (DK)	*minor*	(-48%)	negligible
Holeby (Hyldtofte) Østersøbad (DK)	negligible	(-16%)	negligible
Brunddragerne (DK)	negligible	(-16%)	negligible
Petersdorf (DE)	negligible	(-0.2%)	negligible
Gammendorf (DE)	negligible	(-0.4%)	negligible
Gruener Brink (DE)	negligible	(-0.9%)	negligible
Bannesdorf (DE)	negligible	(-0.7%)	negligible
Suedstrand (DE)	negligible	(-0.5%)	negligible
Fehmarnsund (DE)	negligible	(-0.2%)	negligible

Indirect effect on oxygen

Availability of dissolved oxygen at the sediment-water interface is important for the benthic fauna. Under baseline condition concentration of oxygen in near-bed layer during summer and early autumn varied spatially from super-saturation (>8 mg O_2/l) in shallow areas with benthic vegetation (e.g. Rødsand Lagoon) to 2 mgO_2/l in the deep parts of Mecklenburg Bight (Figure 0.6).

As an indirect result of dredging works in 2015 average concentration of near-bed oxygen will be reduced by up to 10% in Rødsand Lagoon. Outside Rødsand Lagoon and along Lolland coast, reductions in oxygen decrease with depth and also decrease westwards along the Lolland coast (Figure 0-6). The reduction is due to suppression of benthic primary production caused by shading from suspended sediments.

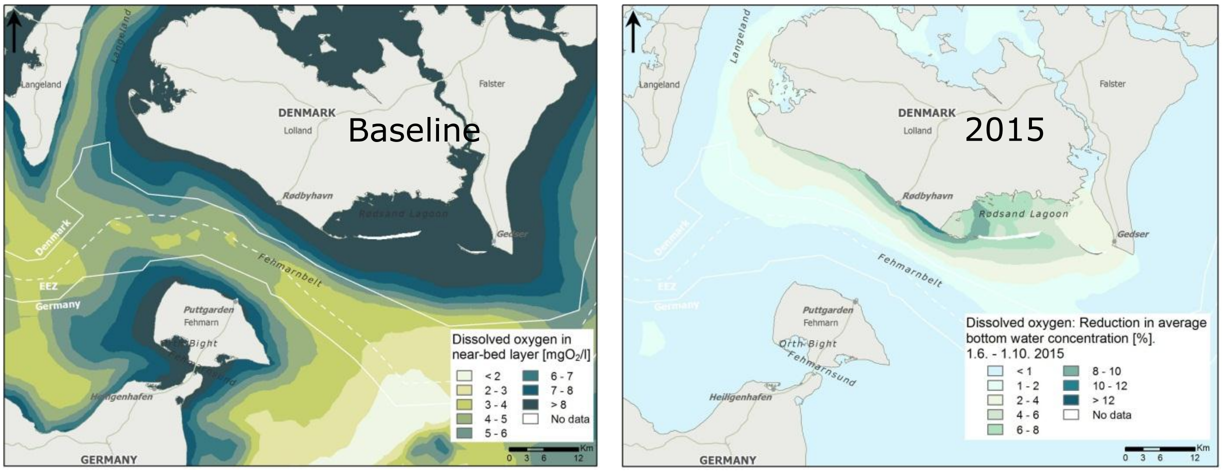


Figure 0.6 Modelled concentration of dissolved oxygen in near-bed layer under baseline conditions (left) and %-reduction in oxygen in 2015 (right), when dredging works will be most intense. Based on average oxygen concentration from 1 June to 1 October.

In areas with the largest reduction in oxygen concentration such as in Rødsand Lagoon and along Lolland, the concentration of oxygen does not fall below 6 mgO_2/l (not shown). Hence, using a critical level of 5.7 mg O_2/l for benthic fauna, reduction in oxygen levels caused by dredging will not constitute an additional pressure on benthos and therefore impairment of indirect oxygen reductions are considered to be insignificant.

Phytoplankton and chlorophyll-a

Chlorophyll-a is an important light-harvesting pigment that occurs in all algae and therefore chlorophyll-a is a much used surrogate measure of plankton algal biomass. In the FEMA model chlorophyll-a is a derived variable depending on phytoplankton biomass and the internal concentrations of nitrogen and phosphorus in phytoplankton. Chlorophyll-a therefore only loosely tracks phytoplankton biomass, but over the growth season there is a very strong correlation between biomass and chlorophyll-a.

Under baseline conditions the concentration of chlorophyll-a during the main production period (March through November) varied in the Fehmarnbelt, from 1.6-2 µg chlorophyll-a/l in the central part, to less than 0.5 µg/l in the Rødsand Lagoon (Figure 0.7). Along the Lolland coast concentration of chlorophyll-al was slightly lower than in the central part of Fehmarnbelt due higher influence of Baltic Sea water (which has a lower chlorophyll-a concentration), but also due to filtration of the population of blue mussels. The highest concentrations were found southeast of Fehmarn caused by influence from the more nutrient-rich Mecklenburg Bight.

During 2015 concentration of chlorophyll-a will be reduced by up to 8-10% in Rødsand Lagoon, while reductions will be lower along the Lolland coast (max reduction 3-4%, Figure 0.7). Reductions in chlorophyll-a are much lower than impacts on the sedentary eelgrass and macroalgae (FEMA 2013a), because plankton is continuously replenished by advective transports mainly from the western Baltic Sea. The water with reduced plankton concentration is advected westwards with minor increases in chlorophyll-a towards the Great Belt. In 2016 and 2017 reductions in chlorophyll-a will gradually decrease (Figure 0.7) and baseline conditions are restored in 2018.

Based on long-term monitoring data a deviation in chlorophyll-a larger than 5% from baseline conditions was considered to reflect a minor impairment, a deviation larger than 10% a medium impairment etc. The modelled reductions in chlorophyll-a correspond to a low and negligible degree of impairment, because reductions in waters of special importance for plankton (depths > 6m) are below 5% in all years during construction (Figure 0.7).

High concentration of suspended sediments (> 10 g/m^3) can lead to increased sedimentation of phytoplankton provided that phytoplankton cells are "sticky" (primarily diatoms) and they occur in high concentrations (> 300 mg/m^3). Such situations only occur during the spring bloom in the Fehmarnbelt.

The criteria for aggregation between phytoplankton and sediment spill and subsequent sedimentation are met along Lolland coast for a 6-7 day period in late March. Assuming that all phytoplankton biomass in these areas aggregate with suspended sediments and settle ca. 14 tons organic carbon will be taken out of the water column and added to the seabed. Under baseline condition and no sediment spill 8.8 tons organic carbon will sediment in "aggregation" area, but summed over an entire year differences in sedimentation is very small. For the entire assessment area the difference in accumulated sedimentation is much below 0.01%.

Overall, the impact on phytoplankton is therefore considered to be insignificant.

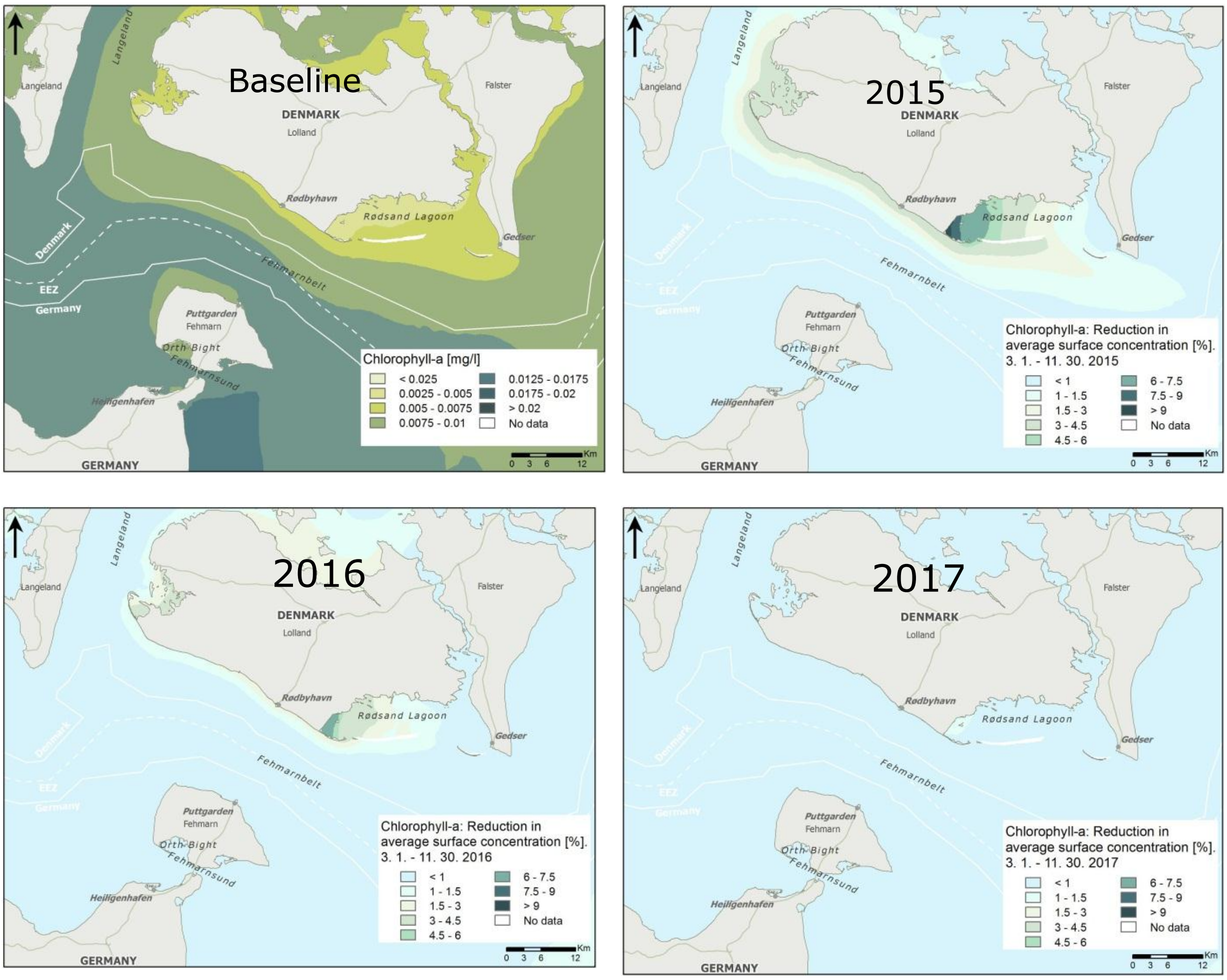


Figure 0.7 Modelled concentration of chlorophyll-a under baseline conditions (upper left) and %-reduction in chlorophyll-a in 2015, when dredging works for tunnel will be most intense. Based on average concentrations 1 March-30 November.

Zooplankton

Production and biomass of zooplankton depend amongst other on availability of food (primarily phytoplankton). Thus, reduction in food concentration mediated through shading from suspended sediments can lead to reduction in growth and biomass in zooplankton, i.e. an indirect effect.

Baseline condition the biomass of zooplankton varies 10-fold within the model area, lowest in Rødsand Lagoon and highest west of Fehmarn (Figure 0.8). The indirect effect of suspended solids on zooplankton was very low in 2015 where sediment spill was highest, not exceeding 1% reduction in average biomass (Figure 0.8). Reductions larger than 0.1% were confined to Rødsand Lagoon, along Lolland coast and Hyllekrog. Significance of impairment due to indirect effects on zooplankton of suspended sediments is insignificant, because reductions in biomass in all areas are below 1% of the baseline condition, and summed over the entire model area reductions are below 0.1%.

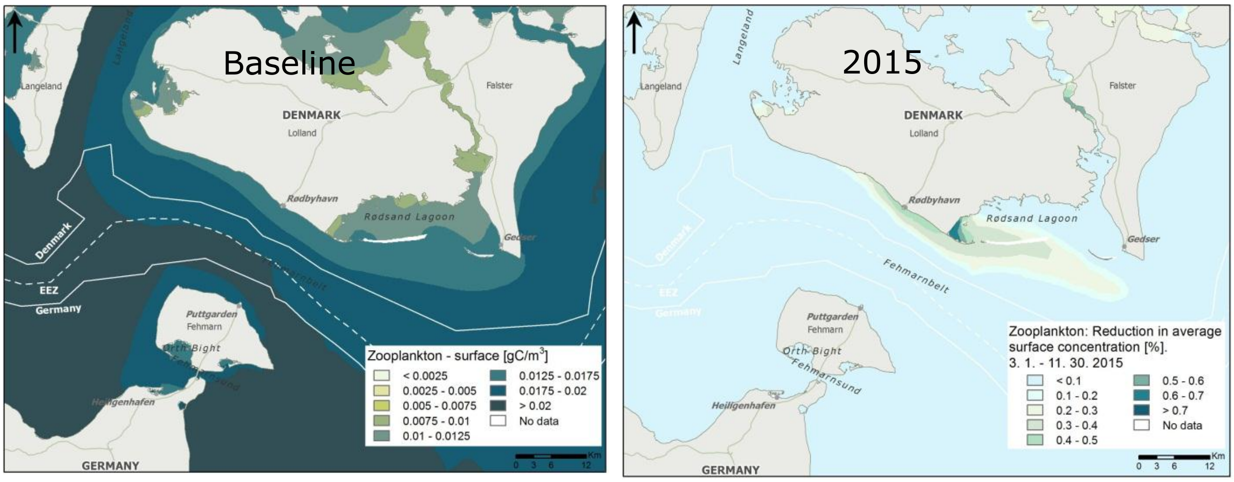


Figure 0.8 Modelled biomass of zooplankton under baseline conditions (left) and %-reduction in zooplankton in 2015 (right), when dredging works for tunnel will be most intense.

Reduction in behaviour, feeding activity and rate of egg production can be affected in some zooplankton species at suspended sediment concentrations above 10-20 mg/l, while copepods that dominate the biomass in Fehmarnbelt are much less sensitive (50-100 mg/l). Recognising that the water column concentration of additional suspended sediments in general is low (< 2 mg/l) in Fehmarnbelt, except in coastal waters along Lolland and in Rødsand the direct impact of suspended solids on zooplankton will be very low.

Recruitment of zooplankton (especially copepods) can be impaired if resting eggs in sediments are covered with 20-40 mm sediment for extended periods. Resting eggs produced in late autumn that settle in the tunnel trench will likely not hatch because of burial under several cm of fine sand. Likewise, permanent burial of resting eggs will take place in the western part of Rødsand lagoon. The total area affected is 760,000 m^2 (Rødsand: 600,000 m^2, tunnel trench: 160,000 m^2). Compared to the total area of assessment (402,282 ha) the affected area is very low and insignificant at 0.02%. In addition, when resting eggs are produced in autumn the biomass of zooplankton is very low in Rødsand, indicating a very low production of resting eggs. Also, given the large exchange with the adjacent areas minor "deficits" in recruitment will be compensated by imports from the Great Belt and the Western Baltic Sea.

Three impacts related to dredging affect zooplankton; direct burial of eggs, direct and indirect effects of suspended sediment on growth. All impacts will be very low and collectively they are assessed to be insignificant.

Assessment of impact of bridge alternative

Temporary impacts

Dredging works for the bridge is much less extensive than for the tunnel, approximately one tenth in terms of sediment spill. Impacts related to dredging works, i.e. extra suspended sediment and sedimentation are much lower than for the tunnel solution. In effect, impairments on water quality and plankton will be very small in terms of degrees and area extension and accordingly, impairments are negligible.

Permanent impacts

Implementation of the bridge solution will lead to a loss of pelagic habitats due to land reclamation (36 ha), piers and pylons (20 ha). The volume lost to land reclamation (ca. 1,000,000 m^3) is confined to waters of general importance for plankton, while volume loss due to piers and pylons (455,000 m^3) almost exclusively is located in waters of special importance for plankton (depth > 6m). Compared to the volume of Fehmarnbelt and the entire volume of assessment area the loss is insignificant.

Detailed predictions of permanent change in hydrographic regime including temperature, salinity, current speed, wave climate were carried out using two different numerical models (MIKE 3 and GETM, see 3.9.2). In general, changes in water quality (Secchi dish depth, Chlorophyll-a) and plankton sub-factors were small (FEHY 2013c) and insignificant with negligible impacts.

Change in hydrography

Hydrographic regime and especially the intensity of vertical mixing have strong influences on water quality and plankton. Long-lasting density stratification is a prerequisite for development of oxygen deficiency in bottom waters because exchange of oxygen with atmosphere is prevented. Besides preventing oxygen deficiency in bottom water, intensity of vertical mixing, structures the composition of plankton communities and increases coupling between benthic grazers and phytoplankton.

Two issues related to the bridge structure were noticeable; the bridge structures will lead to increased vertical mixing of water passing piers and pylons. East of the alignment in the main stem of Fehmarnbelt, the density stratification (difference between bottom and surface waters) will be reduced by up to 0.20 kg/m^3 especially during summer, while density stratification will increase in the Mecklenburg Bight (up to 0.12-0.16 kg/m^3), see Figure 0.9.

Increased strength of stratification especially during summer will favour cyanobacteria compared to other phytoplankton groups, but it is uncertain to what extent a small increase of 0.12-0.20 kg/m^3 in density difference between surface and bottom water (≈ 0.12-0.2 psu) will affect the risk for cyanobacteria blooms. Other factors may be equally important as indicated by the rather poor linkages between meteorological conditions during summer and concentration of cyanobacteria in the Fehmarnbelt area.

The increase in vertical mixing has a positive effect on oxygen in bottom water, with local increases between 0.1-0.2 mg O_2/l covering an area of ca. 150-200 km^2 east of the alignment (Figure 0.10). Interestingly, bottom water in Mecklenburg Bight is not likely to be affected negatively by the stronger stratification, as bottom water oxygen is either unaffected or slightly increased. Therefore, the advection of oxygen-enriched bottom water east of the alignment more than counteracts the increase in stratification.

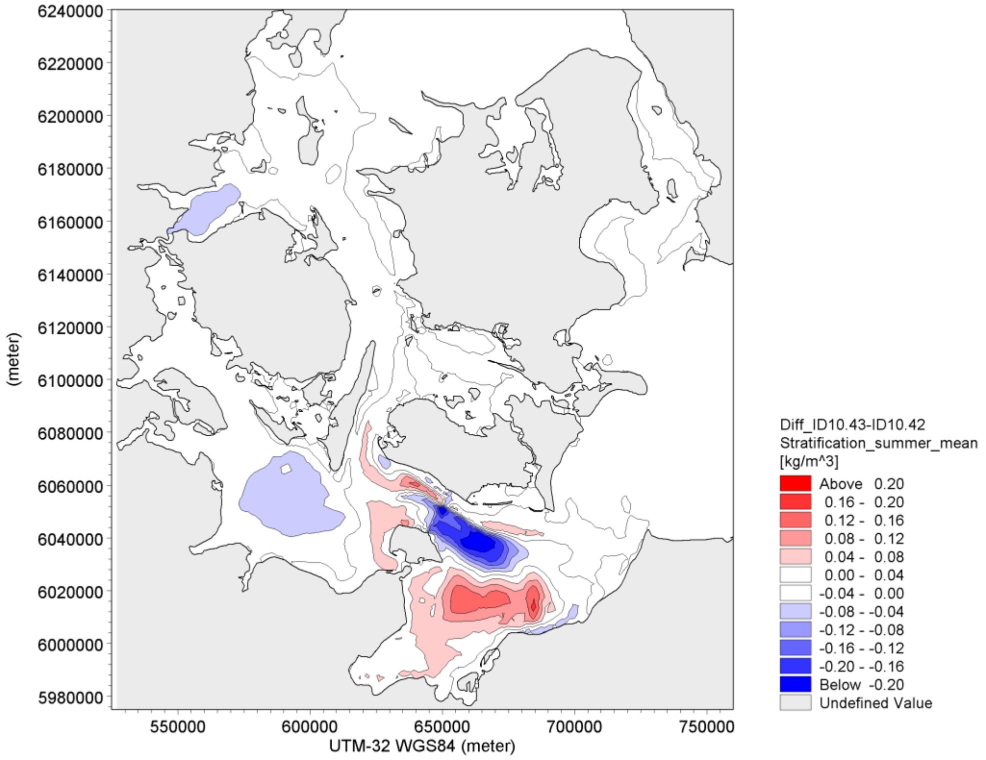


Figure 0.9 Predicted change in stratification during summer using MIKE 3 (upper) local model for "bridge+ferry" case.

In spite of the modest increase in oxygen under average (model) conditions, the down-mixing of oxygen can provide an important supply to the benthic communities during periods with critical low oxygen levels as in 2010.

Because the supply of oxygen is permanent the effect is considered significant (positive).

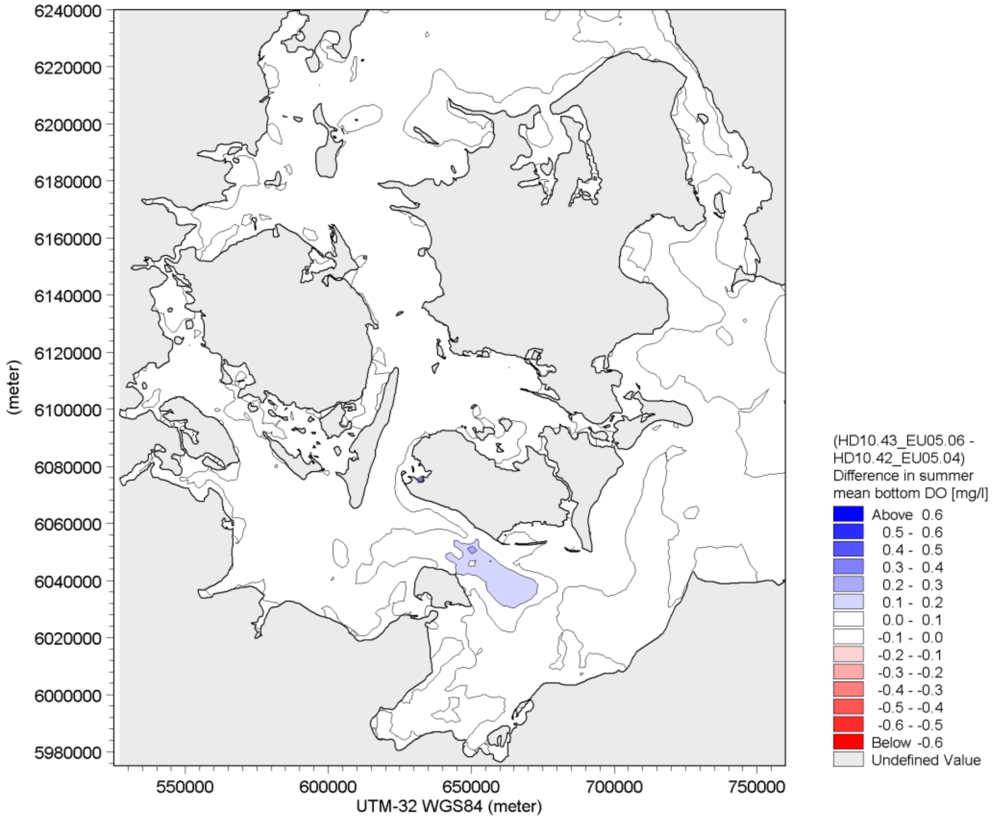


Figure 0.10 Predicted change in bottom water oxygen during summer using MIKE 3 local model for "bridge+ferry" case, from FEHY (2013b).

Additional solid substrate

Bridge piers, pylons and scour protection will increase the area of solid substrate and thereby favour populations of epibenthic invertebrates provided they are substrate limited. Blue mussels will populate the solid substrate and filter phytoplankton advected between pylons and piers. Based on expected abundance and size distribution (based on other bridge projects and wind farms), literature information on filtration capacity in length classes, and average current speed blue mussels theoretically will be able to filter between 0.6% and 1.5% of the phytoplankton biomass passing between bridge pylons.

Mussels on pylons will egest feces that will settle to the seabed and increase oxygen demand in sediments. The area affected below pycnocline is about 4 times the area occupied by the pylons. The increased oxygen demand will not affect oxygen concentration in bottom water.

Other issues of potential larger importance include effects on jellyfish.

Polyps of *Aurelia aurita* will populate the additional hard substrate with a minimum abundance of 20,000 individuals per m^{-2} above the pycnocline and a 10 times lower abundance below the pycnocline.

The additional area of solid substrate from 3 m below MSL and to 20 m is 254,000 m^2. This area has to be compared to the existing area of solid substrate suitable for polyps. Besides stones and perennial macroalgae, shells of blue mussels – living or dead - constitute such substrate. Blue mussels dominate the benthic biomass along Lolland coast and around Fehmarn in three communities: *Mytilus* community, *Bathyporeia* community and *Gammarus* community in an area totalling 1,200,000 ha.

Theoretically, popyps cannot establish on shells of young mussels (i.e. mussels with a shell length less than 35 mm) because young mussels continuously clean their shells with the foot). Hence, shells of mussels with shell length less than 35 mm cannot be considered as an available substrate for *Aurelia* polyps.

In calculating the area of available solid substrate it is assumed that only one of the two shells will be exposed and available to settlement. The total shell area representing living mussels were calculated based on size-abundance data from stations located within the three benthic communities dominated by blue mussels.

Table 0.8 shows the calculated area of shell substrate with the depth range 3-20m where mussels occur.

Table 0.8 Calculated solid substrate composed of shells of blue mussels larger than 35 mm. Only one shell from each individual is included.

Depth range	Shell area m^2/m^2	Community area m^2	Solid Substrate ha
3-14 m	0.12	120 $*10^7$	14,400
14-20 m	0.005	160 $*10^7$	800
total 3-20m			15,200

The calculated area of solid substrate in terms of mussel shells at 15,200 ha is 600 times larger than the solid substrate of the bridge structures and accordingly, the additional recruitment of jellyfish caused by the bridge structures will be insignificant.

Growth of macroalgae on bridge piers, pylons and scour protection below pycnocline can add to oxygen content in bottom water passing the solid structures populated with macroalgae and thereby increase water quality during late summer, when bottom water reaches the seasonal oxygen minimum. The effect on oxygen content was estimated from:

- the additional area of hard substrate in the depth range 14-20m (24.4 ha), where algae was found in baseline investigations
- depth distribution of algal biomass below pycnocline, in the depth range 14-20m as observed in baseline investigations
- modelled specific net growth and oxygen production rates in depth interval 14-20 m averaged from 1 July through September.

The increase in oxygen concentration in bottom water related to algal production was compared to the daily modelled flux of bottom water (25-th and 75-th percentiles) averaged from 1 July through September and a typical minimum concentration (2.5 g O_2/m^3) in late summer.

The maximum oxygen production was estimated to 290 kg O_2/d, not accounting for shelf-shading in the vertically "growing" macroalgae. The calculated increase in oxygen concentration was very small compared to the advective oxygen transport, ranging between 0.03% and 0.06% of baseline transport. Overall, the positive effect on oxygen content in bottom water must be considered as insignificant.

Comparison of bridge and tunnel alternative

Based on the individual assessments of water quality and plankton of the two link alternatives the bridge is the preferred alternative, based on much less impact on transparency of water during 2-3 years of construction, and a slight but positive effect of the bridge on oxygen in bottom waters east of the bridge (Table 0.9).

Table 0.9 Results for the comparison between alternatives (++ preferred, + slightly preferred, 0 no difference).

	Tunnel	Bridge
Suspended sediments		++
Sedimentation	0	0
Toxic substances	0	0
Solid substrate	0	0
Lost habitats	0	0
Hydrographical regime	0	+
Total	0	++

1 INTRODUCTION

1.1 Environmental theme and components assessed

The planned fixed link across Fehmarnbelt may affect the water quality and plankton populations during the construction phase and operation phases.

During the construction phase sediment spill may cause increased turbidity in the water and consequently reduce the aesthetic quality of bathing water and influence the light available in the water column for photosynthesis affecting plankton production and biomass. If toxic substances are found in the dredging area they may be released during dredging operations, which potentially can harm the plankton organisms.

The construction work may also cause an increased sedimentation of suspended matter that can lead to increased sedimentation of phytoplankton by flocculation and bury resting eggs of copepods and potentially affect recruitment of copepods.

The introduction of new structures in the marine environment of Fehmarnbelt may permanently change the hydrography in the area, e.g. by affecting the water column stratification. Furthermore, bridge pylons and pillars will increase the area of hard surface thus constituting additional substrate for blue mussels, macroalgae and ephyra of jellyfish with subsequent effects on the water quality and the plankton communities.

The overall objective of the Fehmarnbelt Fixed Link water quality and plankton biology impact assessment is to carry out detailed analyses of permanent and temporary impacts arising from the construction and operation activities and from the new structures of the Fehmarnbelt Fixed Link.

The position of water quality, plankton and jellyfish in the EIA framework is shown in Table 1.1. It appears that the sub-factor components concern marine water quality, bathing water quality and phytoplankton, mesozooplankton (later called zooplankton) and jellyfish biology.

The components of marine water and marine plankton include the subcomponents given in Table 1.2. The impact assessment has been carried out on the scale of the components and/or sub-components.

Table 1.1 The environmental sub-factors and components assessed in this report.

Environmental factor	Environmental sub-factor	Environmental component
Water	Marine Water	Marine Water Quality Bathing Water Quality
Fauna, flora and biodiversity	Marine flora and fauna	Phytoplankton Zooplankton Jellyfish

Table 1.2 The sub-components included in water quality and marine plankton assessment and the potential environmental effect of changes

Environmental sub-component	Environmental effect of changes
Water Quality – including bathing water	
Nutrients	Affects eutrophication
Suspended solids	Affects turbidity / Secchi depth / bathing water
Light penetration	Affects light availability to benthic plants and aesthetic quality at beaches
Oxygen (near seabed)	Affects benthic organisms
Toxic substances	May affect plankton
Plankton Biology	
Chlorophyll-a	High concentration can be sign of eutrophication
Phytoplankton production, concentration and composition	Affects pelagic and benthic food webs
Bloom of cyanobacteria	Potential toxic, affects food webs
Zooplankton production and biomass	Important to planktivorous fish
Copepod resting eggs	May affect recruitment of copepods
Jellyfish abundance	Affect food webs (competes with fish larvae)

The specific objectives of the water quality and plankton biology impact assessment have been to:

- Predict the degree and severity of impact on water quality and plankton from temporal activities of dredging and sediment spill during construction of the Fehmarnbelt Fixed Link
- Predict the degree and severity of impact on water quality and plankton of permanent changes in the hydrography and by introduction of new structures during the operation phase of the Fehmarnbelt Fixed Link
- Assess the significance of the predicted impacts for water quality and plankton, including
  - increase in harmful substances,
  - change in Secchi depth and nutrient concentrations,
  - change in primary production, phytoplankton biomass and increase in cyanobacterial blooms
  - change in zooplankton biomass (i.e. potential food for plankton eating fish), zooplankton "diversity" (shifts in group composition) and change in recruitment from copepod resting eggs,
  - increased production in ephyras (larval stage) of jellyfish due to introduction of hard substrate

- Compare the impacts on water quality and plankton of the bridge and the tunnel alternatives

The basis for the impact assessment is the water quality and plankton investigated during the baseline investigation. The basis for determining the range, duration and intensity of the pressures is the project description, modelled sediment spill and hydrography and available literature data. The impacts are predicted using hydro-dynamic and ecological modelling, quantitative assessment and expert knowledge.

In this report the time for start of construction is artificially set to 1 October 2014 for the tunnel and 1 January 2015 for the bridge alternative. In the Danish EIA (VVM) and the German EIA (UVS/LBP) absolute year references are not used, but instead the relative time references from start of construction works (year 0, year 1, etc.), i.e. year 0 corresponds to 2014; year 1 corresponds to 2015 etc.

1.2 Water and plankton of Fehmarnbelt

The water quality and the plankton biology have been studied during a two year baseline investigation in 2009-2010 to document the baseline conditions in Fehmarnbelt and adjacent areas (Figure 3.1) based on data collected *in situ* and relevant historical data (FEMA-FEHY 2013). Some of the major findings of the baseline investigations were:

The main and overall variation in water quality and plankton communities was caused by the seasonality in environmental parameters, especially light, nutrients, and temperature, and spatial variation in plankton biomasses and production were modest in the investigation area. At the westernmost stations where high saline water enters the area from Kattegat and Skagerrak, the plankton species composition was slightly differing at the easternmost stations which are influenced by less saline water flowing into the area from the Baltic Sea.

Nutrient concentrations peaked in winter, but decreased in early spring due to the diatom spring bloom, when the chlorophyll-a concentrations, a proxy for phytoplankton biomass, peaked and the Secchi depths were lowest. After the spring bloom particularly dissolved inorganic nitrogen became limiting for phytoplankton growth throughout the entire investigation area. The zooplankton biomass increased in early summer and reached the highest biomasses in summer concomitant with high levels of primary production, approx. 1000-1200 mg C m^{-2} d^{-1}, indicating a high turnover during the warmer summer months and that primary production primarily was based on regenerated nutrients, used by the dominant small phytoplankton flagellates. In autumn larger phytoplankton species became abundant.

Lion's mane jellyfish, moon jellyfish and the invasive jellyfish *Mnemiopsis leidyi* dominated the gelatinous plankton community. Whereas lion's mane jellyfish was primarily located below the pycnocline, the moon jellyfish was abundant above the pycnocline. *M. leidyi* showed no preference in vertical distribution.

Particularly in autumn 2010 periods of severe oxygen deficiency were detected in the bottom water, while 2009 had a higher frequency of stormy winds causing better vertical mixing of oxygen rich water into the bottom layer.

Regarding the bathing water quality, 13 out of the 16 official beaches in the area had 'excellent water quality' and the remaining three beaches had 'sufficient' water quality.

1.2.1 *Importance*

The importance categories of the water quality and the plankton biology have been defined by the functional value of the environmental components (FEMA-FEHY 2013). Since the biological components, phytoplankton, zooplankton and jellyfish, as well as the environmental component water quality, are indirectly protected by the Water Framework Directive, i.e. they may not be deteriorated, and none of the plankton species are adopted on any "Red Lists", a two-level scale of importance is appropriate for these components. Table 1.3 shows the criteria defined to determine the importance levels for the environmental components.

Table 1.3 Importance levels for water quality and plankton of the Fehmarnbelt area

Environ-mental Sub-factor	Environmental component	Importance level	Criteria
Seawater	Water quality	Special	Areas with designated bathing water
		General	All other areas
Marine Plank-ton	Phyto- and zooplankton Zooplankton Jellyfish	Special	Areas with sufficient depth-integrated primary production and high depth-integrated bio-mass of phyto- and zooplank-ton to maintain planktonic food webs
		General	All other areas

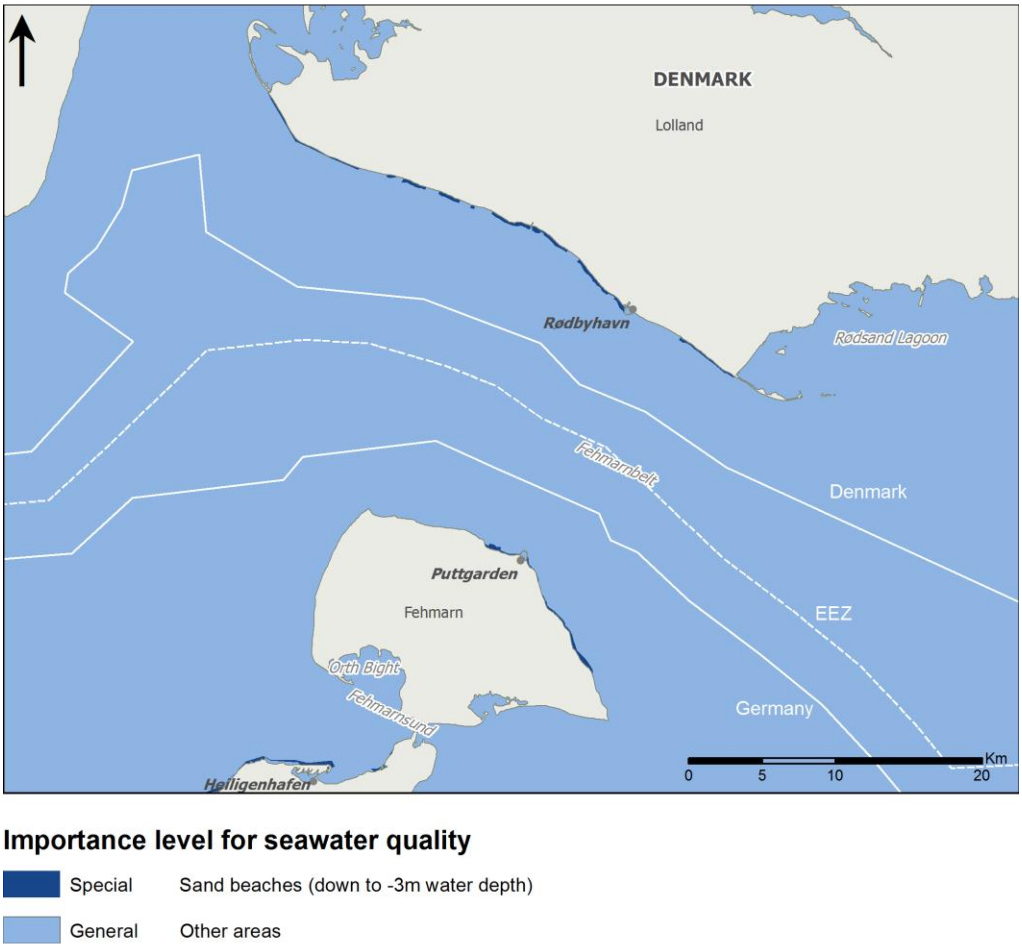


Figure 1.1 *Importance level indicated for water quality in the Fehmarnbelt and adjacent areas. Areas of special importance are designated (according to Bathing Water Directive) bathing waters down to -3m water depth. All other areas are of general importance.*

For the environmental component water quality, areas of special importance are only the bathing waters designated according to the Bathing water Directive; as indicated in Figure 1.1. All other areas are assigned as having general importance. The most important ecosystem services related to plankton in the Fehmarnbelt are the level of primary production and the biomass of phyto- and zooplankton that together are of high importance for production of planktivorous fish such as fish larvae and herring, and for production of blue mussels that again constitute the prime food for eider birds. The depth-integrated primary production and plankton (phyto- and zooplankton) biomass increase with water depth and areas of special importance have been delineated by a 6 m depth contour (Figure 1.2). At water depths larger than 6 m the water column production (above the pycnocline) is double as high as the production at water depths less than 6 m. Likewise, depth-integrated biomass of zooplankton is 3-5 times higher at depths larger than 6 m than in shallow waters. Consequently, only the areas with a depth below 6 m have been assigned having general importance in Figure 1.2.

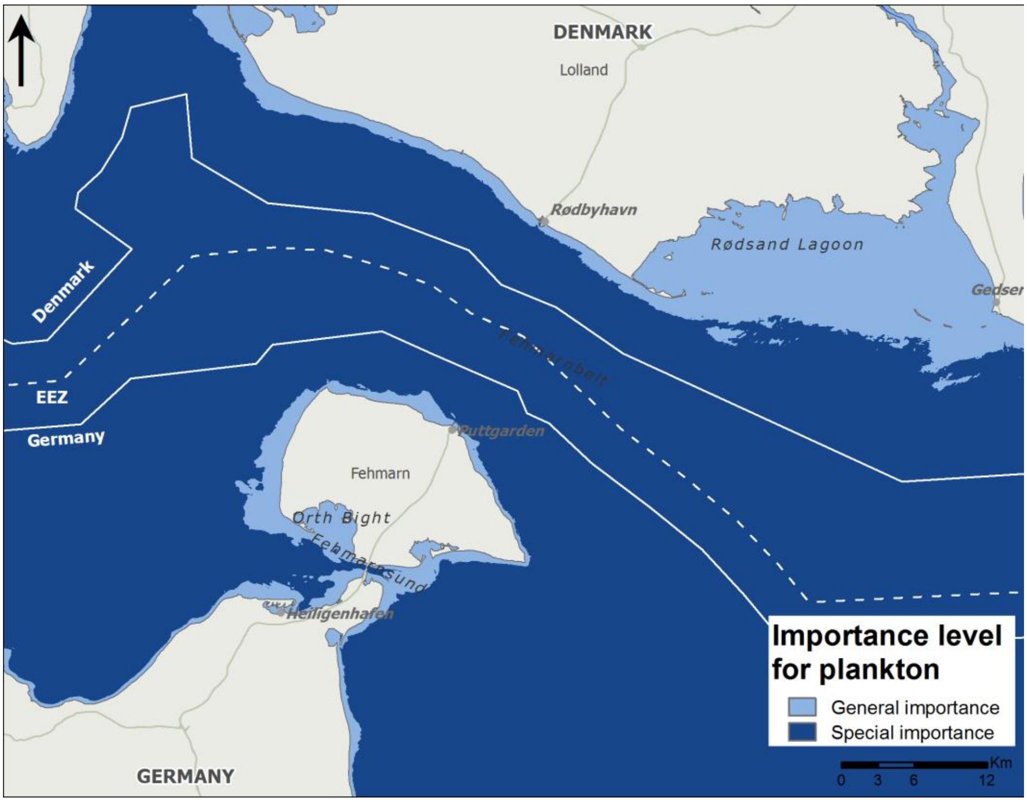


Figure 1.2 Importance level for plankton in the Fehmarnbelt and adjacent areas. Areas of special importance are of primary functional value to the other trophic levels of the marine ecosystem as they offer the highest depth-integrated plankton production and biomass *(= areas with water depth >6m). All other areas are of general importance.*

In the Water Framework Directive (WFD) classification of water quality is based on the deviation of the present conditions from the reference conditions representing the conditions prior to significant human influence. For phytoplankton excessive blooms and high biomass are classical examples of unwanted responses most often resulting from increased nutrients inputs to the sea, but changes in the hydrographical regime could potentially also affect the nutrient availability.

Zooplankton serves as food resource for planktivorous fish (bottom up control) and subsequently also for higher trophic levels (predatory fish and birds). Top down, zooplankton has a grazing function on phytoplankton.

2 THE FEHMARNBELT FIXED LINK PROJECT

2.1 General description of the project

The Impact assessment is undertaken for two fixed link alternatives:

- Immersed tunnel E-ME (August 2011)
- Cable Stayed Bridge Variant 2 B-EE (October 2010)

In the following the two alternatives is described.

2.1.1 The Immersed Tunnel (E-ME August 2011)

The alignment for the immersed tunnel passes east of Puttgarden, crosses the Fehmarnbelt in a soft curve and reaches Lolland east of Rødbyhavn as shown in Figure 2.1 along with near-by NATURA2000 sites.

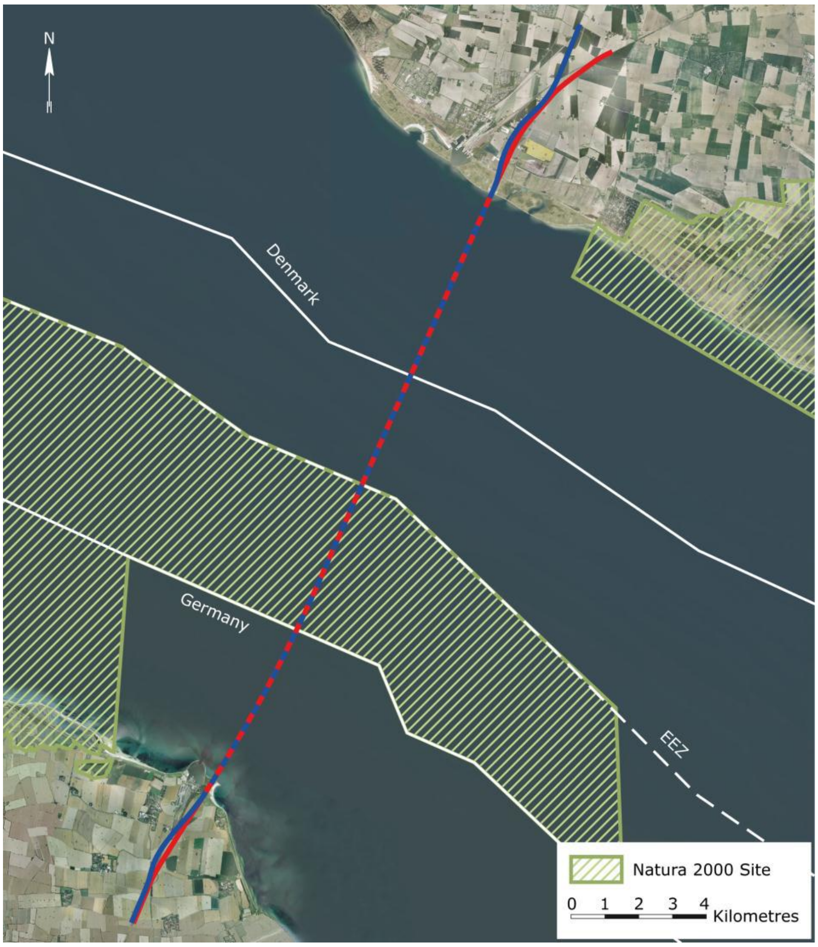


Figure 2.1 Conceptual design alignment for immersed tunnel E-ME (August 2011)

Tunnel trench

The immersed tunnel is constructed by placing tunnel elements in a trench dredged in the seabed, see Figure 2.2. The proposed methodology for trench dredging comprises mechanical dredging using Backhoe Dredgers (BHD) up to 25m and Grab Dredgers (GD) in deeper waters. A Trailing Suction Hopper Dredger (TSHD) will be used to rip the clay before dredging with GD. The material will be loaded into barges and transported to the near-shore reclamation areas where the soil will be unloaded from the barges by small BHDs. A volume of approx. 14.5 mio. m^3 sediment is handled.

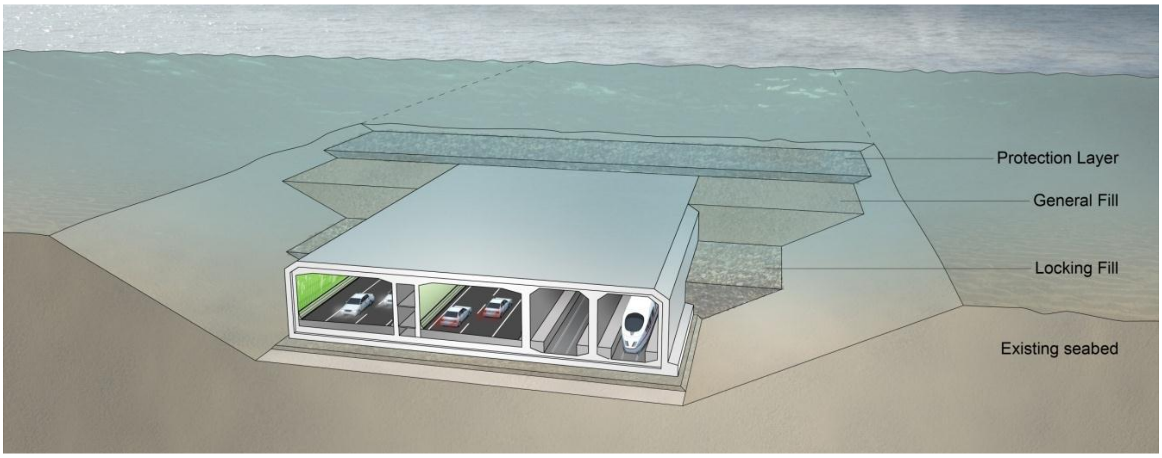


Figure 2.2 Cross section of dredged trench with tunnel element and backfilling

A bedding layer of gravel forms the foundation for the elements. The element is initially kept in place by placing locking fill followed by general fill, while on top there is a stone layer protecting against damage from grounded ships or dragging anchors. The protection layer and the top of the structure are below the existing seabed level except near the shore. At these locations, the seabed is locally raised to incorporate the protection layer over a distance of approximately 500-700m from the proposed coastline. Here the protection layer is thinner and made from concrete and a rock layer.

Tunnel elements

There are two types of tunnel elements: standard elements and special elements. There are 79 standard elements, see Figure 2.3. Each standard element is approximately 217 m long, 42m wide and 9m tall. Special elements are located approximately every 1.8 km providing additional space for technical installations and maintenance access. There are 10 special elements. Each special element is approximately 46m long, 45m wide and 13m tall.

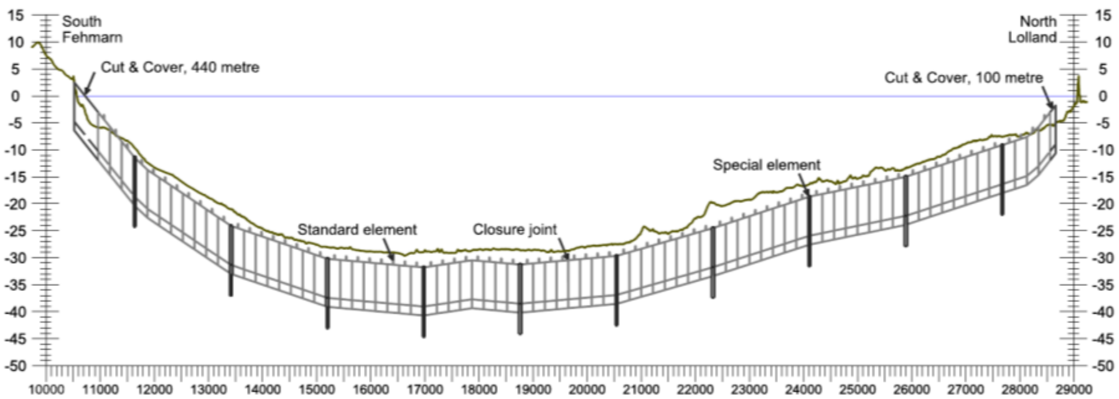


Figure 2.3 Vertical tunnel alignment showing depth below sea level

The cut and cover tunnel section beyond the light screens is approximately 440m long on Lolland and 100m long on Fehmarn. The foundation, walls, and roof are constructed from cast in-situ reinforced concrete.

Tunnel drainage

The tunnel drainage system will remove rainwater and water used for cleaning the tunnel. Rainwater entering the tunnel will be limited by drainage systems on the approach ramps. Firefighting water can be collected and contained by the system for subsequent handling. A series of pumping stations and sump tanks will transport the water from the tunnel to the portals where it will be treated as required by environmental regulations before being discharged into the Fehmarnbelt.

Reclamation areas

Reclamation areas are planned along both the German and Danish coastlines to accommodate the dredged material from the excavation of the tunnel trench. The size of the reclamation area on the German coastline has been minimized. Two larger reclamations are planned on the Danish coastline. Before the reclamation takes place, containment dikes are to be constructed some 500m out from the coastline.

The landfall of the immersed tunnel passes through the shoreline reclamation areas on both the Danish and German sides

Fehmarn reclamation areas

The proposed reclamation at the Fehmarn coast does not extend towards north beyond the existing ferry harbour at Puttgarden. The extent of the Fehmarn reclamation is shown in Figure 2.4. The reclamation area is designed as an extension of the existing terrain with the natural hill turning into a plateau behind a coastal protection dike 3.5m high. The shape of the dike is designed to accommodate a new beach close to the settlement of Marienleuchte.

Figure 2.4 Reclamation area at Fehmarn

The reclaimed land behind the dike will be landscaped to create an enclosed pasture and grassland habitat. New public paths will be provided through this area leading to a vantage point at the top of the hill, offering views towards the coastline and the sea.

The Fehmarn tunnel portal is located behind the existing coastline. The portal building on Fehmarn houses a limited number of facilities associated with essential equipment for operation and maintenance of the tunnel and is situated below ground level west of the tunnel.

A new dual carriageway is to be constructed on Fehmarn for approximately 3.5km south of the tunnel portal. This new highway rises out of the tunnel and passes onto an embankment next to the existing harbour railway. The remainder of the route of the highway is approximately at level. A new electrified twin track railway is to be constructed on Fehmarn for approximately 3.5km south of the tunnel portal. A lay-by is provided on both sides of the proposed highway for use by German customs officials.

Lolland reclamation area

There are two reclamation areas on Lolland, located either side of the existing harbour. The reclamation areas extend approximately 3.7km east and 3.4km west of the harbour and project approximately 500m beyond the existing coastline into the Fehmarnbelt. The proposed reclamation areas at the Lolland coast do not extend beyond the existing ferry harbour at Rødbyhavn.

The sea dike along the existing coastline will be retained or reconstructed, if temporarily removed. A new dike to a level of +3m protects the reclamation areas against the sea. To the eastern end of the reclamation, this dike rises as a till cliff to a level of +7m. Two new beaches will be established within the reclamations. There will also be a lagoon with two openings towards Fehmarnbelt, and revetments at the openings. In its final form the reclamation area will appear as three types of landscapes: recreation area, wetland, and grassland - each with different natural features and use.

The Lolland tunnel portal is located within the reclamation area and contained within protective dikes, see Figure 2.5. The main control centre for the operation and maintenance of the Fehmarnbelt Fixed Link tunnel is housed in a building located over the Danish portal. The areas at the top of the perimeter wall, and above the portal building itself, are covered with large stones as part of the landscape design. A path is provided on the sea-side of the proposed dike to serve as recreation access within the reclamation area.

Figure 2.5 Tunnel portal area at Lolland

A new dual carriageway is to be constructed on Lolland for approximately 4.5km north of the tunnel portal. This new motorway rises out of the tunnel and passes onto an embankment. The remainder of the route of the motorway is approximately at level. A new electrified twin track railway is to be constructed on Lolland for approximately 4.5km north of the tunnel portal. A lay-by is provided in each direction off the landside highway on the approach to the tunnel for use by Danish customs officials. A facility for motorway toll collection will be provided on the Danish landside.

Marine construction works
The temporary works comprises the construction of two temporary work harbours, the dredging of the portal area and the construction of the containment dikes. For the harbour on Lolland an access channel is also provided. These harbours will be integrated into the planned reclamation areas and upon completion of the tunnel construction works, they will be dismantled/removed and backfilled.

Production site
The current design envisages the tunnel element production site to be located in the Lolland east area in Denmark. Figure 2.6 shows one production facility consisting of two production lines. For the construction of the standard tunnel elements for the Fehmarn tunnel four facilities with in total eight production lines are anticipated.

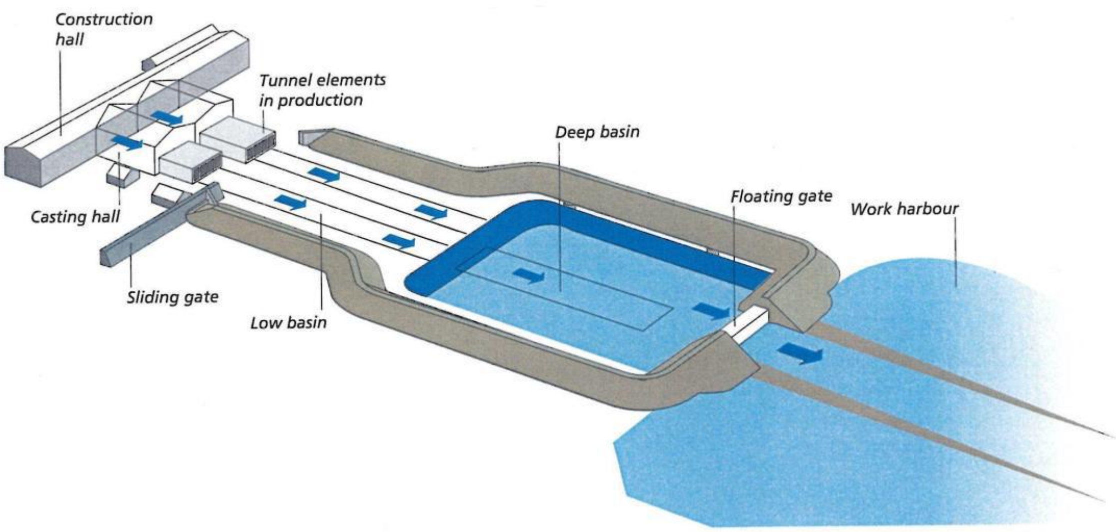


Figure 2.6 Production facility with two production lines

In the construction hall, which is located behind the casting and curing hall, the reinforcement is handled and put together to a complete reinforcement cage for one tunnel segment. The casting of the concrete for the segments is taking place at a fixed location in the casting and curing hall. After the concrete of the segments is cast and hardened enough the formwork is taken down and the segment is pushed forward to make space for the next segment to be cast. This process continues until one complete tunnel element is cast. After that, the tunnel element is pushed into the launching basin. The launching basin consists of an upper basin, which is located at ground level and a deep basin where the tunnel elements can float. In the upper basin the marine outfitting for the subsequent towing and immersion of the element takes place. When the element is outfitted, the sliding gate and floating gate are closed and sea water is pumped into the launching basin until the elements are floating. When the elements are floating they are transferred from the low basin to the deep basin. Finally the water level is lowered to normal sea level, the floating gate opened and the element towed to sea. The proposed lay-out of the production site is shown in Figure 2.7.

Dredging of approx. 4 mio. m^3 soil is required to create sufficient depth for temporary harbours, access channels and production site basins.

Figure 2.7 Proposed lay-out of the production site east of Rødbyhavn

2.1.2 The Cable Stayed Bridge

The alignment for the marine section passes east of Puttgarden harbour, crosses the belt in a soft S-curve and reaches Lolland east of Rødbyhavn, see Figure 2.8.

Bridge concept

The main bridge is a twin cable stayed bridge with three pylons and two main spans of 724m each. The superstructure of the cable stayed bridge consists of a double deck girder with the dual carriageway road traffic running on the upper deck and the dual track railway traffic running on the lower deck. The pylons have a height of 272m above sea level and are V-shaped in transverse direction. The main bridge girders are made up of 20m long sections with a weight of 500 to 600t. The standard approach bridge girders are 200m long and their weight is estimated to ~ 8,000t.

Caissons provide the foundation for the pylons and piers of the bridge. Caissons are prefabricated placed 4m below the seabed. If necessary, soils are improved with 15m long bored concrete piles. The caissons in their final positions end 4m above sea level. Prefabricated pier shafts are placed on top of the approach bridge caissons. The pylons are cast in situ on top of the pylon caissons. Protection Works are prefabricated and installed around the pylons and around two piers on both sides of the pylons. These works protrudes above the water surface. The main bridge is connected to the coasts by two approach bridges. The southern approach bridge is

5,748m long and consists of 29 spans and 28 piers. The northern approach bridge is 9,412m long and has 47 spans and 46 piers.

Figure 2.8 Main bridge part of the cable stayed bridge

Land works
A peninsula is constructed both at Fehmarn and at Lolland to use the shallow waters east of the ferry harbours breakwater to shorten the Fixed Link Bridge between its abutments. The peninsulas consist partly of a quarry run bund and partly of dredged material and are protected towards the sea by revetments of armour stones.

Fehmarn
The peninsula on Fehmarn is approximately 580m long, measured from the coastline, see Figure 2.9. The gallery structure on Fehmarn is 320m long and enables a separation of the road and railway alignments. A 400m long ramp viaduct bridge connects the road from the end of the gallery section to the motorway embankment. The embankments for the motorway are 490m long. The motorway passes over the existing railway tracks to Puttgarden Harbour on a bridge. The profile of the railway and motorway then descend to the existing terrain surface.

Lolland
The peninsula on Lolland is approximately 480m long, measured from the coastline. The gallery structure on Lolland is 320m long. The existing railway tracks to Rødbyhavn will be decommissioned, so no overpass will be required. The viaduct bridge for the road is 400m, the embankments for the motorway are 465m long and for railway 680m long. The profile of the railway and motorway descend to the natural terrain surface.

Figure 2.9 Proposed peninsula at Fehmarn east of Puttgarden

Drainage on main and approach bridges
On the approach bridges the roadway deck is furnished with gullies leading the drain water down to combined oil separators and sand traps located inside the pier head before discharge into the sea.

On the main bridge the roadway deck is furnished with gullies with sand traps. The drain water passes an oil separator before it is discharged into the sea through the railway deck.

Marine construction work
The marine works comprises soil improvement with bored concrete piles, excavation for and the placing of backfill around caissons, grouting as well as scour protection. The marine works also include the placing of crushed stone filling below and inside the Protection Works at the main bridge.

Soil improvement will be required for the foundations for the main bridge and for most of the foundations for the Fehmarn approach bridge. A steel pile or reinforcement cage could be placed in the bored holes and thereafter filled with concrete.

The dredging works are one of the most important construction operations with respect to the environment, due to the spill of fine sediments. It is recommended that a grab hopper dredger with a hydraulic grab be employed to excavate for the caissons both for practical reasons and because such a dredger minimises the sediment spill. If the dredged soil cannot be backfilled, it must be relocated or disposed of.

Production sites
The temporary works comprises the construction of two temporary work harbours with access channels. A work yard will be established in the immediate vicinity of the harbours, with facilities such as concrete mixing plant, stockpile of materials, storage of equipment, preassembly areas, work shops, offices and labour camps.

The proposed lay-out of the production site is shown in Figure 2.10.

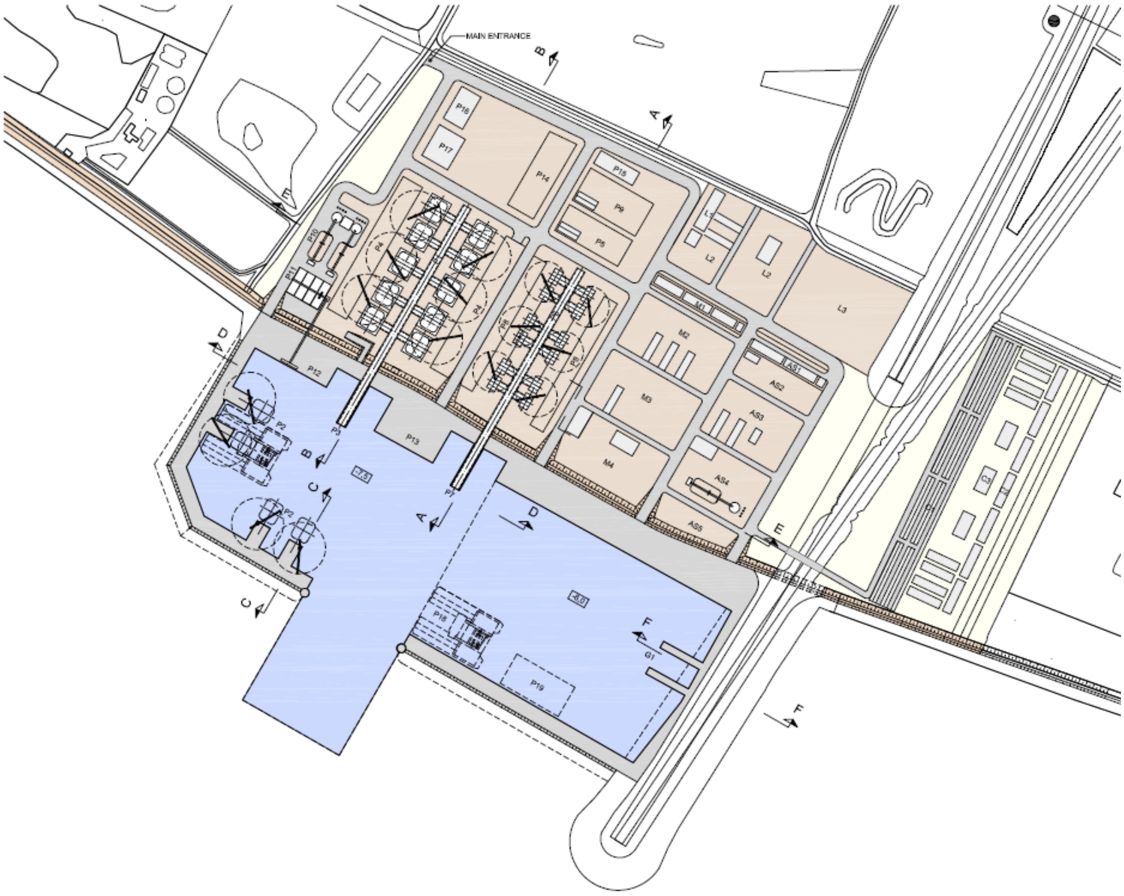


Figure 2.10 Proposed lay-out of the production site at Lolland east of Rødbyhavn

2.2 *Important project pressures*

The potential pressures from the Fehmarnbelt Fixed Link project affecting water quality (WQ; including bathing water quality) and plankton biology are listed in Table 2.1. The pressures and the components affected are in accordance with the issues identified in the Scoping Report (Femern and LBV-SH-Lübech 2010).

Table 2.1 List of pressures related to the Fixed Link project that potentially can affect the different components of seawater quality, bathing water quality or plankton biology. Pressures are assigned to construction phase (C), related to structures (S) or operation phase (O). Relevancy evaluation are given in the main text.WQ = water quality.

Pressure		**Potentially Impacted (Sub-) Components**
C	Suspended sediment by sediment spill due to dredging/disposal activities	WQ: Secchi depth and bathing water Plankton: primary production, chlorophyll-a, phytoplankton composition, zooplankton consumption and production
C	Sedimentation of spilled sediment	Plankton: phytoplankton reduced biomass by excess sedimentation zooplankton: production and biomass
C	Toxic substances released from spilled sediment	WQ: concentration of contaminants Plankton: toxic effects
C	Nutrients released from spilled sediment	WQ: concentration of nutrients Plankton: primary production, chlorophyll-a, phytoplankton composition,
C	Release of oxygen consuming substances from spilled sediment	WQ: concentration of oxygen
S	Loss of habitat	All plankton sub-components
S	Additional solid substrate	Jellyfish (by increased recruitment) Additional oxygen production by macroalgae (by) Reduction in phytoplankton biomass caused by blue mussels (by)
S	Changes in hydrographical regime (due to permanent structures or cessation of ferries)	WQ: Secchi depth, Oxygen Plankton: Chl-a and Cyanobacterial bloom
C/O	Toxic substances released by wastewater, drainage and other releases	WQ: concentration of contaminants Plankton: toxic effects
C/O	Nutrients released by wastewater, drainage and other releases	WQ: concentration of nutrients Plankton: primary production, chlorophyll-a, phytoplankton composition, zooplankton consumption and production

The relevancy of the pressures is evaluated below. Most of the relevant pressures are of provisional, of indirect nature and related to dredging activities (for bridge, tunnel, work harbours, access channel). Recovery time for seawater quality and plankton biology is not an issue for the provisional impacts. Because these components are highly dynamic with short generation times and in addition, and because of continuous exchange with the adjacent waters (the western Baltic and the Great Belt), changes in the components are diluted and quickly "washed away" and conditions will re-establish very fast after stop of the direct pressure.

Permanent effects on water quality and plankton are only relevant for the bridge alternative which may cause permanent changes in the hydrographical regime. The continuous and extensive displacement of the water implies that all pressures potentially may cause impacts outside the Fehmarnbelt.

2.2.1 *Suspended sediment from sediment spill*

Spill of sediment during dredging and disposal lead to increase of suspended solids in the water column, i.e. increased turbidity. The increase can be divided into two: a) increased turbidity due to sediment plumes and b) increased turbidity due to re-suspension of settled spill sediment. Potential effects on water quality and plankton ecology are due to direct impacts and indirect impacts.

Direct impacts

High concentrations of suspended solids reduce transparency of water which may result in reduced aesthetic bathing water quality, if the turbid water reaches the beaches during the bathing season.

Zooplankton may also be directly affected by high concentrations. The Fehmarnbelt zooplankton is dominated by copepods, cladocerans, meroplankton (benthic invertebrate larvae), rotifers and ciliates (FEMA-FEHY 2013). As several of the taxonomic groups are non-selective filter-feeders, high concentration of suspended solids interfere with the feeding process leading to lower intake of food (i.e. phytoplankton) and a reduced growth.

Indirect impacts

Intuitively, and based on numerous EIA studies of dredging works, the most important temporary impacts of dredging activities relate to shading effects of spilled sediments and how this effect is transmitted in the planktonic and benthic ecosystem. Production of phytoplankton is limited by nutrient availability and light, and if penetration of light into the water column is reduced primary production will be reduced at larger depths, where light intensity is low and production light-limited.

Reduction in primary production will also lead to reduction in the total biomass of phytoplankton (Sherk et al. 1976). Naturally occurring high turbidity in estuaries is known to suppress pelagic primary production (Lara-Lara et al. 1990) with the turbid Western Scheldt as a much cited example (e.g. Underwood and Kromkamp 1999).

Different phytoplankton groups differ in their requirement to light intensity for optimal growth. Therefore, changes in light availability in the water column may induce shifts in composition of the phytoplankton community, by favouring the groups with the lowest requirements or those that can escape light limitation by aggregating at sea surface.

For zooplankton a potentially important indirect impact of suspended matter is mediated through changes in food availability. Other potential impacts related to spilled sediments are facilitated sedimentation of phytoplankton, meaning that phytoplankton cells can be trapped during flocculation of sediment particles and thus be "lost" from the pelagic environment by sedimentation. A sketch of these impacts is shown in Figure 2.11.

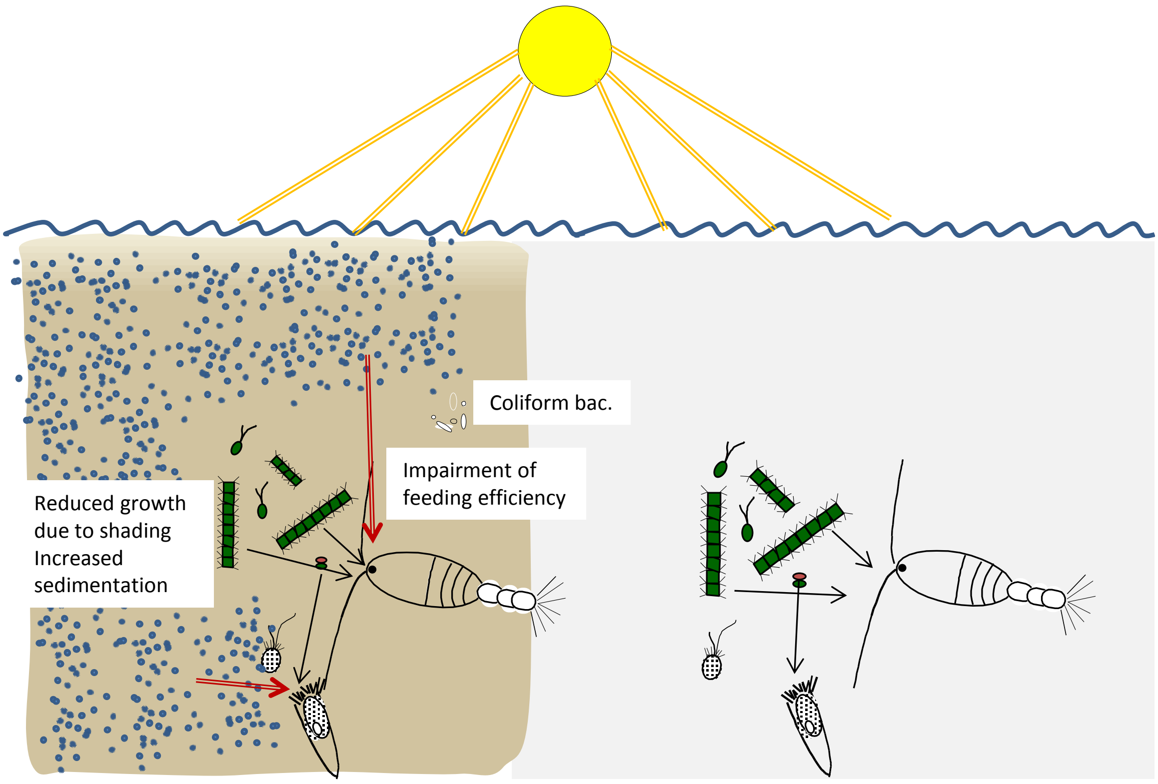


Figure 2.11 Conceptual diagram of the most important indirect effects related to dredging activities (left part of figure). The blue dots illustrate sediment particles.

2.2.2 Sedimentation of spilled sediment

Suspended solids from spilled sediments will undergo more or less regular sedimentation and resuspension events until its final deposit at seabed where shear stress is so low that sediments cannot be eroded.

During periods of high concentrations of phytoplankton (e.g. blooms) concomitant high concentrations of suspended matter may increase sedimentation of phytoplankton by flocculation. The effect is poorly understood but seems to be 'regulated' by a varying degree of 'stickiness' of phytoplankton cells.

Several copepod species produce 'resting' eggs during unfavourable conditions or as part of their normal life-cycle. These eggs sediment to seabed and there is a risk of lower rate of hatching if they are deeply burrowed in the sediment, e.g. caused by deposited sediment spill.

2.2.3 Toxic substances released from spilled sediment

Concentration of toxic substances in sediments to be dredged in the Fehmarnbelt was below national and international guidelines for contaminated sediments and accordingly, is was concluded that spread of dredged sediment would not constitute a threat to benthic organisms (FEHY 2013d). However, depending on actual concentrations, amount of sediment spilled and the character of sediments, toxic substances may be released to the water during dredging and if the concentrations exceed environmental quality standards (EQS) for seawater, effects on the plankton is to be expected.

2.2.4 Nutrients released from spilled sediment

Release of nutrients from dredged sediment could potentially stimulate algal growth but the pressure was assessed and evaluated as insignificant (FEHY 2013d). Based on elutriation studies using surface sediment from the alignment corridor, a daily release of 0.5 kg inorganic nitrogen (DIN) and 2 kg phosphate (DIP/PO_4) can be expected. Compared to local inputs from land, atmosphere and especially the advection of nutrients from the neighbouring waters, the maximum enrichment that dredging can cause is 0.1% for DIP and much lower for DIN. The pressure is therefore not discussed further in this report.

2.2.5 *Oxygen consuming substances released from spilled sediment*

During dredging reduced substances accumulated in sediments (e.g. H_2S) may be released into the water column leading to reduction in oxygen levels in water when these substances are oxidized. Based on "shake-bottle" experiments with surface and subsurface sediments collected at 18 stations along Fixed link alignment a daily oxygen consumption rate of 93 kg O_2 (range: 68-181 kg O_2/d) was calculated for a spill rate of 3% and a daily dredging intensity of 5000 m^3/dredger (FEHY 2013d). Critical impacts may occur if both current speeds and oxygen levels are low (e.g. during summer) in bottom water (below pycnocline). Above the pycnocline current speed is larger and effects of additional oxygen demand will easier be eliminated by oxygen exchange with the atmosphere.

2.2.6 *Loss of habitat*

Loss of pelagic volume due to occupation by bridge piers and pylons, land reclamation areas and protections reefs will reduce the total primary production and plankton biomass in the EIA assessment area.

In contrast to plankton, water quality parameters are always area- or volume-specific and loss will not be relevant for the water quality.

Besides occupation of volume, the bridge structure will reduce insolation on sea surface in the vicinity of the structure due to shading and thus potentially affect primary production locally. This impact cannot be quantified without detailed knowledge of surface characteristics (especially reflective properties) of piers, pylons and underside of roadway. This pressure is likely to be very small (<< 0.01% of total production in assessment area) and will not be considered further.

2.2.7 *Additional solid substrate*

The bridge solution will increase the area of hard substrate potentially promoting sessile stages of plankton organisms. Among these is the polyp stage of scyphozoan medusa such as the moon jellyfish that regularly show up in mass occurrences. Such phenomena are often explained by successful settlement of tiny planula larvae on suitable hard substrate and its development into conspicuous polyp colonies. There are also theories in scientific literature that western Baltic population of *Aurelia aurita* may be limited by availability of hard substrate, and introduction of additional hard substrate by bridge pylons and piers may indirectly promote mass occurrence of jellyfish in the area.

Other issues are settlement and growth of filter-feeding blue mussels on piers and pylons. They can reduce concentration of phytoplankton passing through piers with advected water.

2.2.8 *Changes in hydrographical regime*

For the Fixed Link solution particularly piers and pylons will function as local areas with increased friction to flow. As the water column in the deeper parts of Fehmarnbelt is stratified with brackish surface water (typically 12-15 psu) and denser and more saline bottom water (typically 20-26 psu) any friction to flow will tend to increase mixing between surface and deeper waters around piers potentially improving oxygen levels in bottom water during summer. Expenditure to increased mixing will reduce the momentum in advected water, which may lead to reduced water exchange in lagoons and bays and secondary, thereby affect risks for cyano-bacteria blooms in surface waters and reduced oxygen levels during critical periods during summer.

Intense ship traffic is known to increase vertical mixing of the water column and, increase resuspension of sediments especially when manoeuvring in harbours (Lindholm et al. 2001). Termination of the ferry traffic between Rødby Harbour and Puttgarden after establishment of the Fixed Link theoretically would decrease the vertical mixing. Based on hydrodynamic modelling and expert judgements (FEHY 2013b) concluded that termination of ferry traffic would not affect the vertical and horizontal exchanges in Fehmarnbelt. In addition to external nutrient loads and in-solation, the temporal and spatial variation in water quality and plankton are largely under hydrodynamic control (Filardo and Dunstan 1985) and accordingly, if hydro-dynamic conditions are unaffected by termination of ferry traffic changes in water quality and plankton are unlikely. Therefore, effects of ferry termination on water quality and plankton are negligible and are not discussed further.

2.2.9 Wastewater, drainage and other releases

Due to land reclamation on Lolland the outfall from the Rødbyhavn sewer treatment plant needs to be relocated to a new position. This position is not identified yet but it is assumed that relocation will not cause deterioration of bathing water quality and the general water quality. During operation additional wastewater will be pro-duced from facilities for servicing tunnel and bridge infrastructure. It is assumed that the existing criteria to regulate sewer outlet will be met, eventually by increas-ing the capacities of local sewer treatment plant if required.

Drainage water will be discharged partly via the existing water courses, but for both alternatives an additional outlet at each coastline is planned. The magnitude of pressure on water quality is primarily under stochastic control depending on amount of drainage water discharged per precipitation event, the number of events and, the distance to the outlet area.

For the tunnel solution run-off water from highway and railroad will be collected and treated before discharge and, for the bridge solution run-off from approach bridges (railroad and highway) and from the lanes of the main bridge and from peninsulas will be collected and treated. Discharge from landscaped areas and from embankments (tunnel solution) and from railroad on the main bridge will not be collected and treated.

The total discharge will not exceed 1 m^3/s on average and the specific outlets will be positioned off-shore to ensure sufficient mixing and dilution with Fehmarnbelt water. The effluents have been assessed to having no effect on the salinity and general hydrography close to the source point or on larger scales, taken into ac-count the normal variation in salinity in the affected areas (9–25 psu) and the effi-cient flushing (FEHY 2013b). Likewise, given the limited discharge of drainage wa-ter, concentration of nutrients and toxic substances should be extremely high to trigger toxic or eutrophication effects in vicinity of discharge.

Thus, the magnitude of pressure of drainage is negligible and will not be considered further in this report.

3 DATA AND METHODS

3.1 Other Fehmarnbelt Fixed Link assessment studies

The report draws upon water quality and plankton data collected during the 2 year baseline study (reported in FEMA-FEHY 2013); the baseline and assessment study on sediment chemistry (FEHY 2013d); the assessment of the predicted sediment spill from the tunnel and bridge alternative respectively (FEHY 2013a), and the impact assessment of the hydrography of the Fehmarnbelt area (FEHY 2013b).

3.2 Areas of Assessment

The area of assessment (Figure 3.1) for water quality and plankton ecology include the Western Baltic area, which encompasses Fehmarnbelt, the southern Great Belt, the Kiel Bight, southern Little Belt, and the Mecklenburg Bight because these areas include (1) the footprint of the proposed dredging, (2) the potential extent of sediment plume dispersion caused by dredging for the bridge and tunnel scenarios and, (3) the designated bathing waters along Lolland and Fehmarn coasts.

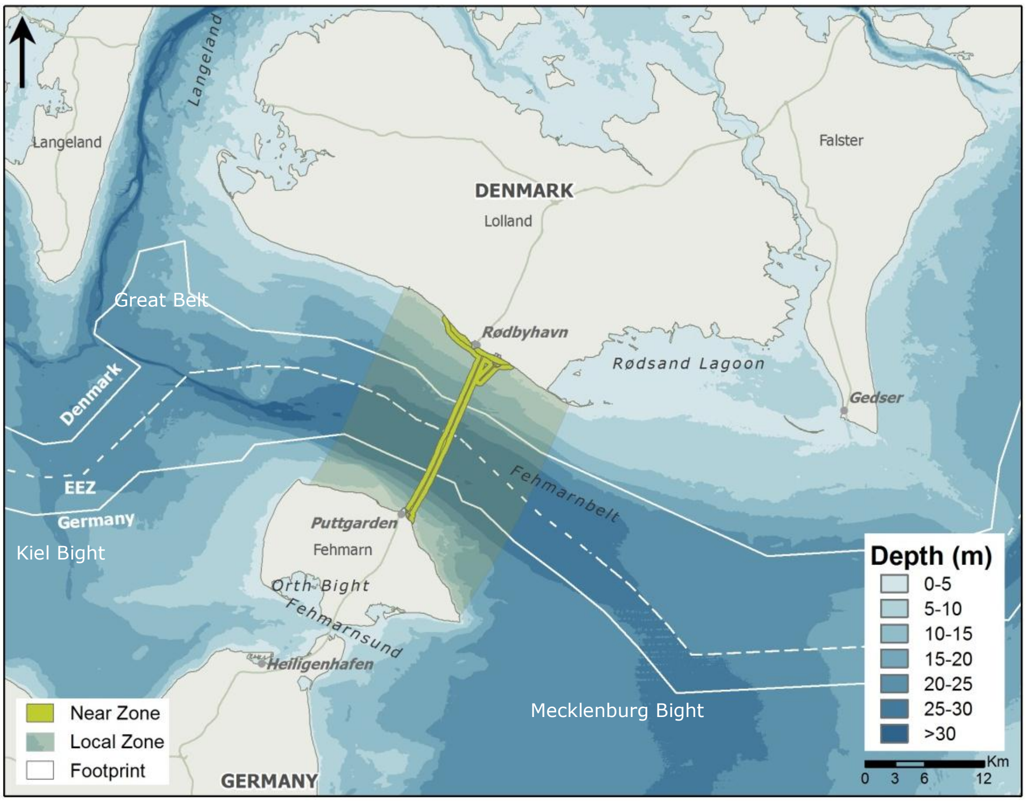


Figure 3.1 Assessment area and defined sub-regions for the tunnel and bridge alternative. Only near -zone differs slightly between the link alternatives.

Besides the assessment area the predicted impacts is related to administrative zones (national, coastal, EEZ) and to zones around the project (near and local zones if considered relevant. The location of the zones with respect to the EIA assessment area is shown in Figure 3.1.

There are no transboundary impacts and thus no assessment is carried out on a transboundary scale.

3.3 The Assessment Methodology

To ensure a uniform and transparent basis for the EIA, a general impact assessment methodology for the assessment of predictable impacts of the Fixed Link Project on the environmental factors (see box 3.1) has been prepared. The methodology is defined by the impact forecast methods described in the scoping report (Femern and LBV-SH-Lübeck 2010, section 6.4.2). In order to give more guidance and thereby support comparability, the forecast method has been further specified.

As the impact assessments cover a wide range of environs (terrestrial and marine) and environmental factors, the general methodology is further specified and in some cases modified for the assessment of the individual environmental factors (e.g. the optimal analyses for migrating birds and relatively stationary marine bottom fauna are not identical). These necessary modifications are explained in Section 3.2.2. The specification of methods and tools used in the present report are given in the following sections of Chapter 3.

3.3.1 Overview of terminology

To assist reading the background report as documentation for the German UVS/LPB and the Danish VVM, the Danish and German terms are given in the columns to the right.

Term	Explanation	Term DK	Term DE
Environmental factors	The environmental factors are defined in the EU EIA Directive (EU 1985) and comprise: Human beings, Fauna and flora, Soil, Water, Air, Climate, Landscape, Material assets and cultural heritage. In the sections below only the term environmental factor is used; covering all levels (factors, sub-factors, etc.; see below). The relevant level depends on the analysis.	Miljøforhold/-faktor	Schutzgut
Sub-factors	As the Fixed Link Project covers both terrestrial and marine sections, each environmental factor has been divided into three sub-factor: Marine areas, Lolland and Fehmarn (e.g. Marine waters, Water on Lolland, and Water on Fehmarn)	Sub-faktor	Teil-Schutzgut
Components and sub-components	To assess the impacts on the sub-factors, a number of components and sub-components are identified. Examples of components are e.g. Surface waters on Fehmarn, Groundwater on Fehmarn; both belonging to the sub-factor Water on Fehmarn. The sub-components are the specific indicators selected as best suitable for assessing the impacts of the Project. They may represent different characteristics of the environmental system; from specific species to biological communities or specific themes (e.g. trawl fishery, marine tourism).	Component/sub-komponent	Komponente
Construction phase	The period when the Project is constructed; including permanent and provisional structures. The construction is planned for 6½ years.	Anlægsfase	Bauphase
Structures	Constructions that are either a permanent elements	Anlæg	Anlage

Term	Explanation	Term DK	Term DE
	of the Project (e.g. bridge pillar for bridge alternative and land reclamation at Lolland for tunnel alternative), or provisional structures such as work harbours and the tunnel trench.		
Operation phase	The period from end of construction phase until decommissioning.	Driftsfase	Betriebsphase
Permanent	Pressure and impacts lasting for the life time of the Project (until decommissioning).	Permanent	Permanent
Provisional (temporary)	Pressure and impacts predicted to be recovered within the life time of the project. The recovery time is assessed as precise as possible and is in addition related to Project phases.	Midlertidig	Temporär
Pressures	A pressure is understood as all influences deriving from the Fixed Link Project; both influences deriving from Project activities and influences originating from interactions between the environmental factors. The type of the pressure describes its relation to construction, structures or operation.	Belastning	Wirkfaktoren
Magnitude of pressure	The magnitude of pressure is described by the intensity, duration and range of the pressure. Different methods may be used to arrive at the magnitude; dependent on the type of pressure and the environmental factor to be assessed.	Belastnings-størrelse	Wirkintensität
Footprint	The footprint of the Project comprises the areas occupied by structures. It comprises two types of footprint; the permanent footprint deriving from permanent confiscation of areas to structures, land reclamation etc., and provisional footprint which are areas recovered after decommissioning of provisional structures. The recovery may be due to natural processes or Project aided re-establishment of the area.	Arealinddragelse	Flächeninan-spruchnahme
Assessment criteria and Grading	Assessment criteria are applied to grade the components of the assessment schemes. Grading is done according to a four grade scale: very high, high, medium, minor or a two grade scale: special, general. In some cases grading is not doable. Grading of magnitude of pressure and sensitivity is method dependent. Grading of importance and impairment is as far as possible done for all factors.	Vurderings-kriterier og graduering	Bewertungs-kriterien und Ein-stufung
Importance	The importance is defined as the functional values to the natural environment and the landscape.	Betydning	Bedeutung
Sensitivity	The sensitivity describes the environmental factors capability to resist a pressure. Dependent on the subject assessed, the description of the sensitivity may involve intolerance, recovery and importance.	Følsomhed/ Sårbarhed	Empfindlichkeit
Impacts	The impacts of the Project are the effects on the environmental factors. Impacts are divided into Loss and Impairment.	Virkninger	Auswirkung
Loss	Loss of environmental factors is caused by permanent and provisional loss of area due to the footprint of the Project; meaning that loss may be permanent or provisional. The degree of loss is described by the intensity, the duration and if feasible, the range.	Tab af areal	Flächenverlust

Term	*Explanation*	*Term DK*	*Term DE*
Severity of loss	Severity of loss expresses the consequences of occupation of land (seabed). It is analysed by combining magnitude of the Project's footprint with importance of the environmental factor lost due to the footprint.	Omfang af tab	Schwere der Auswirkungen bei Flächenverlust
Impairment	An impairment is a change in the function of an environmental factor.	Forringelse	Funktionsbeeinträchtigung
Degree of impairment	The degree of impairments is assessed by combining magnitude of pressure and sensitivity. Different methods may be used to arrive at the degree. The degree of impairment is described by the intensity, the duration and if feasible, the range.	Omfang/grad af forringelser	Schwere der Funktionsbeeinträchtigung
Severity of impairment	Severity of impairment expresses the consequences of the Project taking the importance of the environmental factor into consideration; i.e. by combining the degree impairment with importance.	Virkningens væsentlighed	Erheblichkeit
Significance	The significance is the concluding evaluation of the impacts from the Project on the environmental factors and the ecosystem. It is an expert judgment based on the results of all analyses.		

It should be noted that in the sections below only the term environmental factor is used; covering all levels of the receptors of the pressures of the Project (factors, sub-factors, component, sub-components). The relevant level depends on the analysis and will be explained in the following methodology sections (section 3.2.3 and onwards).

3.3.2 The Impact Assessment Scheme

The overall goal of the assessment is to arrive at the severity of impact where impact is divided into two parts; loss and impairment (see explanation above). As stated in the scoping report, the path to arrive at the severity is different for loss and impairments. For assessment of the *severity of loss* the footprint of the project (the areas occupied) and the *importance* of the environmental factors are taken into consideration. On the other hand, the assessment of severity of impairment comprises two steps; first the *degree of impairment* considering the magnitude of pressure and the sensitivity. Subsequently the severity is assessed by combining the degree of impairment and the importance of the environmental factor. The assessment schemes are shown in Figure 3.2 - Figure 3.4. More details on the concepts and steps of the schemes are given below. As mentioned above, modification are required for some environmental factors and the exact assessment process and the tools applied vary dependent on both the type of pressure and the environmental factor analysed. As far as possible the impacts are assessed quantitatively; accompanied by a qualitative argumentation.

3.3.3 Assessment Tools

For the impact assessment the assessment matrices described in the scoping report have been key tools. Two sets of matrices are defined; one for the assessment of loss and one for assessment of impairment.

The matrices applied for assessments of severity of loss and degree of impairment are given in the scoping report (Table 6.4 and Table 6.5) and are shown below in Table 3.1 and Table 3.2, respectively.

Table 3.1 The matrix used for assessment of the severity of loss. The magnitude of pressure = the footprint of the Project is always considered to be very high.

Magnitude of the predicted pressure (footprint)	**Importance of the environmental factors**			
	Very high	High	Medium	Minor
Very High	**Very High**	**High**	**Medium**	**Minor**

The approach and thus the tools applied for assessment of the degree of impairment varies with the environmental factor and the pressure. For each assessment the most optimal state-of-the-art tools have been applied, involving e.g. deterministic and statistical models as well as GIS based analyses. In cases where direct analysis of causal-relationship is not feasible, the matrix based approach has been applied using one of the matrices in Table 3.2 (Table 6.5 of the scoping report) combining the grades of magnitude of pressure and grades of sensitivity. This method gives a direct grading of the degree of impairment. Using other tools to arrive at the degree of impairment, the results are subsequently graded using the impairment criteria. The specific tools applied are described in the following sections of Chapter 3.

Table 3.2 The matrices used for the matrix based assessment of the degree of impairment with two and four grade scaling, respectively

Magnitude of the predicted pressure	**Sensitivity of the environmental factors**			
	Very high	High	Medium	Minor
Very high	**General loss of function, must be substantiated for specific instances**			
High	**Very High**	**High**	**High**	**Medium**
Medium	**High**	**High**	**Medium**	**Low**
Low	**Medium**	**Medium**	**Low**	**Low**

Magnitude of the predicted pressure	**Sensitivity of the environmental factors**	
	Special	General
Very high	**General loss of function, must be substantiated for specific instances**	
High	**Very High**	**High**
Medium	**High**	**Medium**
Low	**Medium**	**Low**

To reach severity of impairment one additional matrix has been prepared, as this was not included in the scoping report. This matrix is shown in Table 3.3.

Table 3.3 The matrix used for assessment of the severity of impairment

Degree of impairment	Importance of the environmental factors			
	Very high	High	Medium	Minor
Very High	**Very High**	**High**	**Medium**	**Minor**
High	**High**	**High**	**Medium**	**Minor**
Medium	**Medium**	**Medium**	**Medium**	**Minor**
Low	**Minor**	**Minor**	**Minor**	**Negligible**

Degree of impairment	Importance of the environmental factors	
	Special	General
Very high	**Very High**	**Medium**
High	**High**	**Medium**
Medium	**Medium**	**Medium**
Low	**Minor**	**Minor**

3.3.4 *Assessment Criteria and Grading*

For the environmental assessment two sets of key criteria have been defined: Importance criteria and the Impairment criteria. The importance criteria is applied for grading the importance of an environmental factor, and the impairment criteria form the basis for grading of the impairments caused by the project. The criteria have been discussed with the authorities during the preparation of the EIA.

The impairment criteria integrate pressure, sensitivity and effect. For the impact assessment using the matrix approach, individual criteria are furthermore defined for pressures and sensitivity. The criteria were defined as part of the impact analyses (severity of loss and degree of impairment). Specific assessment criteria are developed for land and marine areas and for each environmental factor. The specific criteria applied in the present impact assessment are described in the following sections of Chapter 3 and as part of the description of the impact assessment.

The purpose of the assessment criteria is to grade according to the defined grading scales. The defined grading scales have four (very; high, Medium; minor) or two (special; general) grades. Grading of magnitude of pressure and sensitivity is method dependent, while grading of importance and impairment is as far as possible done for all factors.

3.3.5 *Identifying and quantifying the pressures from the Project*

The pressures deriving from the Project are comprehensively analysed in the scoping report; including determination of the pressures which are important to the individual environmental sub-factors (Femern and LBV SH Lübeck 2010, chapter 4 and 7). For the assessments the magnitude of the pressures is estimated.

The magnitudes of the pressures are characterised by their type, intensity, duration and range. The *type* distinguishes between pressures induced during construction, pressures from the physical structures (footprints) and pressures during operation. The pressures during construction and from provisional structures have varying duration while pressures from staying physical structure (e.g. bridge piers) and from the operation phase are permanent. Distinctions are also made between direct and indirect pressures where direct pressures are those imposed directly by the Project activities on the environmental factors while the indirect pressures are the consequences of those impacts on other environmental factors and thus express the interactions between the environmental factors.

The *intensity* evaluates the force of the pressure and is as far as possible estimated quantitatively. The *duration* determines the time span of the pressure. It is stated as relevant for the given pressure and environmental factor. Some pressures (like footprint) are permanent and do not have a finite duration. Some pressures occur in events of different duration. The *range* of the pressure defines the spatial extent. Outside of the range, the pressure is regarded as non-existing or negligible.

The magnitude of pressure is described by pressure indicators. The indicators are based on the modes of action on the environmental factor in order to achieve most optimal descriptions of pressure for the individual factors; e.g. mm deposited sediment within a certain period. As far as possible the magnitude is worked out quantitatively. The method of quantification depends on the pressure (spill from dredging, noise, vibration, etc.) and on the environmental factor to be assessed (calling for different aggregations of intensity, duration and range).

3.3.6 Importance of the Environmental Factors

The importance of the environmental factor is assessed for each environmental sub-factor. Some sub-factors are assessed as one unity, but in most cases the importance assessment has been broken down into components and/or sub-components to conduct a proper environmental impact assessment. Considerations about standing stocks and spatial distribution are important for some sub-factors such as birds and are in these cases incorporate in the assessment.

The assessment is based on *importance criteria* defined by the functional value of the environmental sub-factor and the legal status given by EU directives, national laws, etc. the criteria applied for the environmental sub-factor(s) treated in the present report are given in a later section.

The importance criteria are grading the importance into two or four grades (see section 3.2.4). The two grade scale is used when the four grade scale is not applicable. In a few cases such as climate, grading does not make sense. As far as possible the spatial distribution of the importance classes is shown on maps.

3.3.7 Sensitivity

The optimal way to describe the sensitivity to a certain pressure varies between the environmental factors. To assess the sensitivity more issues may be taken into consideration such as the intolerance to the pressure and the capability to recover after impairment or a provisional loss. When deterministic models are used to assess the impairments, the sensitivity is an integrated functionality of the model.

3.3.8 Severity of loss

Severity of loss is assessed by combining information on magnitude of footprint, i.e. the areas occupied by the Project with the importance of the environmental factor (Figure 3.2. Loss of area is always considered to be a very high magnitude of pressure and therefore the grading of the severity of loss is determined by the importance (see Table 3.1).

The loss is estimated as hectares of lost area. As far as possible the spatial distribution of the importance classes is shown on maps.

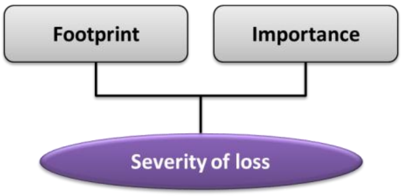


Figure 3.2 The assessment scheme for severity of loss

3.3.9 Degree of impairment

The degree of impairment is assessed based on the magnitude of pressure (involving intensity, duration and range) and the sensitivity of the given environmental factor (Figure 3.3). In worst case, the impairment may be so intensive that the function of the environmental factor is lost. It is then considered as loss like loss due to structures, etc.

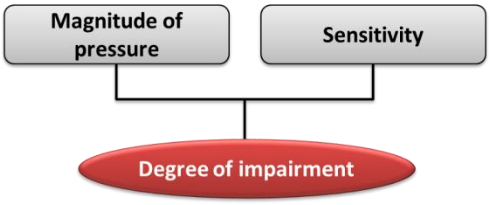


Figure 3.3 The assessment scheme for degree of impairment

As far as possible the degree is worked out quantitatively. As mentioned earlier the method of quantification depends on the environmental factor and the pressure to be assessed, and of the state-of-the-art tools available for the assessment.

No matter how the analyses of the impairment are conducted, the goal is to grade the degree of impairment using one of the defined grading scales (two or four grades). Deviations occur when it is not possible to grade the degree of impairment. The spatial distribution of the different grades of the degree of impairment is shown on maps.

3.3.10 Severity of Impairment

Severity of impairment is assessed from the grading's of degree of impairment and of importance of the environmental factor (Figure 3.4) using the matrix in Table 3.3. If it is not possible to grade degree of impairment and/or importance an assessment is given based on expert judgment.

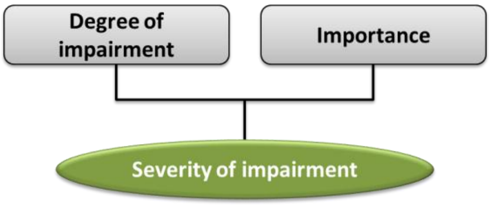


Figure 3.4 The assessment scheme for severity of impairment

In the UVS and the VVM, the results of the assessment of severity of impairment support the significance assessment. The UVS and VVM do not present the results as such.

3.3.11 Range of impacts

Besides illustrating the impacts on maps, the extent of the marine impacts is assessed by quantifying the areas impacted in predefined zones. The zones are shown in Figure 3.5. In addition the size of the impacted areas located in the German national waters and the German EEZ zone, respectively, as well as in the Danish national plus EEZ waters (no differentiation) are calculated. If relevant the area of transboundary impacts are also estimated.

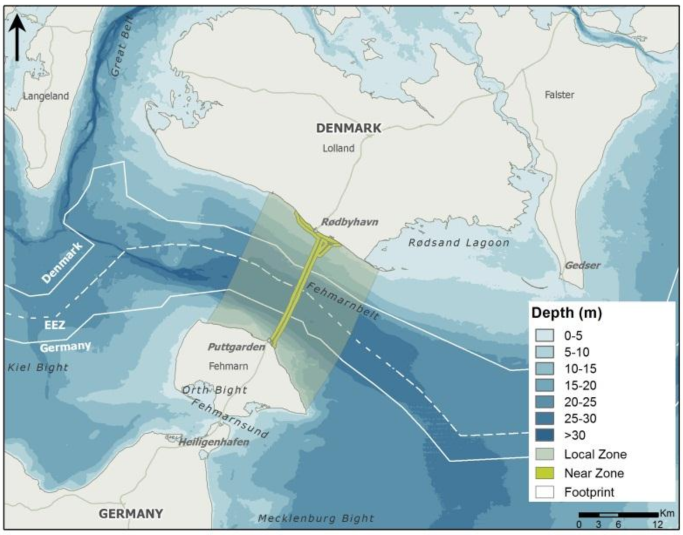


Figure 3.5 The assessment zones applied for description of the spatial distribution of the impacts. The near zone illustrated is valid for the tunnel alternative. It comprises the footprint and a surrounding 500 m band. The local zone is identical for the two alternatives. The eastern and western borders are approximately 10 km from the centre of the alignment.

3.3.12 Duration of impacts

Duration of impacts (provisional loss and impairments) is assessed based on recovery time (restitution time). The recovery time is given as precise as possible; stating the expected time frame from conclusion of the pressure until pre-project conditions is restored. The recovery is also related to the phases of the project using Table 3.4 as a framework.

Table 3.4 Framework applied to relate recovery of environmental factors to the consecutive phases of the Project

Impact recovered within:	In wording
Construction phase+	recovered within 2 year after end of construction
Operation phase A	recovered within 10 years after end of construction
Operation phase B	recovered within 24 years after end of construction
Operation phase C	recovery takes longer or is permanent

It should be noted that in the background reports, the construction phase has been indicated by exact years (very late 2014-2020 (tunnel) and early 2014-2020 (bridge). As the results are generic and not dependent on the periodization of the construction phase, the years are in the VVM and the UVS indicated as calendar year 0, year 1, etc. This means that the construction of the tunnel starts in Year 0 (only some initial activities) and the bridge construction commence in year 1.

3.3.13 *Significance*

The impact assessment is finalised with an overall assessment stating the significance of the predicted impacts. This assessment of significance is based on expert judgement. The reasoning for the conclusion on the significance is explained. Aspects such as degree and severity of impairment/severity of loss, recovery time and the importance of the environmental factor are taken into consideration.

3.3.14 *Comparison of environmental impacts from project alternatives*

Femern A/S will prepare a final recommendation of the project alternative, which from a technical, financial and environmental point of view can meet the goal of a Fehmarnbelt Fixed Link from Denmark to Germany. As an important input to the background for this recommendation, the consortia have been requested to compare the two alternatives, immersed tunnel and cable-stayed bridge, with the aim to identify the alternative having the least environmental impacts on the environment. The bored tunnel alternative is discussed in a separate report. In order to make the comparison as uniform as possible the ranking is done using a ranking system comprising the ranks: 0 meaning that it is not possible to rank the alternatives, + meaning that the alternative compared to the other alternative has a minor environmental advantage and ++ meaning that the alternative has a noticeable advantage. The ranking is made for the environmental factor or sub-factor included in the individual report (e.g. for the marine area: hydrography, benthic fauna, birds, etc.). To support the overall assessment similar analyses are sometimes made for individual pressures or components/subcomponents. It should be noticed that the ranking addresses only the differences/similarities between the two alternatives and not the degree of impacts.

3.3.15 *Cumulative impacts*

The aim of the assessment of cumulative impacts is to evaluate the extent of the environmental impact of the project in terms of intensity and geographic extent compared with the other projects in the area and the vulnerability of the area. The assessment of the cumulative conditions does not only take into account existing conditions, but also land use and activities associated with existing utilized and unutilized permits or approved plans for projects in the pipe.

When more projects within the same region affect the same environmental conditions at the same time, they are defined to have cumulative impacts. A project is relevant to include, if the project meets one or more of the following requirements:

- The project and its impacts are within the same geographical area as the fixed link
- The project affects some of the same or related environmental conditions as the fixed link
- The project results in new environmental impacts during the period from the environmental baseline studies for the fixed link were completed, which thus not is included in the baseline description
- The project has permanent impacts in its operation phase interfering with impacts from the fixed link

Based on the criteria above the following projects at sea are considered relevant to include in the assessment of cumulative impacts on different environmental conditions. All of them are offshore wind farms:

Project	Placement	Present Phase	Possible interactions
Arkona-Becken Südost	North East of Rügen	Construction	Sediment spill, habitat displacement, collision risk, barrier effect
EnBW Windpark Baltic 2	South east off Kriegers Flak	Construction	Sediment spill, habitat displacement, collision risk, , barrier effect
Wikinger	North East of Rügen	Construction	Sediment spill, habitat displacement, collision risk, , barrier effect
Kriegers Flak II	Kriegers Flak	Construction	Sediment spill, habitat displacement, collision risk, barrier effect
GEOFReE	Lübeck Bay	Construction	Sediment spill, habitat displacement, collision risk
Rødsand II	In front of Lolland's southern coast	Operation	Coastal morphology, collision risk, barrier risk

Rødsand II is included, as this project went into operation while the baseline investigations for the Fixed Link were conducted, for which reason in principle a cumulative impact cannot be excluded.

On land, the following projects are considered relevant to include:

Project	Placement	Phase	Possible cumulative impact
Extension of railway	Orehoved to Holeby	Construction	Area loss, noise and dust
		Operation	Landscape, barrier effect
Construction of emergency lane	Guldborgsund to Rødbyhavn	Construction	Area loss, noise and dust
		Operation	Landscape, barrier effect
Extension of railway	Puttgarden to Lübeck	Construction	Area loss, noise and dust
		Operation	Landscape, barrier effect
Upgrading of road to highway	Oldenburg to Puttgarden	Construction	Area loss, noise and dust
		Operation	Landscape, barrier effect

The increased traffic and resultant environmental impacts are taken into account for the environmental assessment of the fixed link in the operational phase and is thus not included in the cumulative impacts. In the event that one or more of the included projects are delayed, the environmental impact will be less than the environmental assessment shows.

For each environmental subject it has been considered if cumulative impact with the projects above is relevant.

3.3.16 *Impacts related to climate change*

The following themes are addressed in the EIA for the fixed link across Fehmarnbelt:

- Assessment of the project impact on the climate, defined with the emission of greenhouse gasses (GHG) during construction and operation
- Assessment of expected climate change impact on the project
- Assessment of the expected climate changes impact on the baseline conditions
- Assessment of cumulative effect between expected climate changes and possible project impacts on the environment
- Assessment of climate change impacts on nature which have to be compensated and on the compensated nature.

Changes in the global climate can be driven by natural variability and as a response to anthropogenic forcing. The most important anthropogenic force is proposed to be the emission of greenhouse gases, and hence an increasing of the concentration of greenhouse gases in the atmosphere.

Even though the lack of regulations on this issue has made the process of incorporating the climate change into the EIA difficult, Femern A/S has defined the following framework for assessment of importance of climate change to the environmental assessments made:

- The importance of climate change is considered in relation to possible impacts caused by the permanent physical structures and by the operation of the fixed link.
- The assessment of project related impacts on the marine hydrodynamics, including the water flow through the Fehmarnbelt and thus the water exchange of the Baltic Sea, is based on numerical model simulations, for baseline and the project case, combined with general model results for the Baltic Sea and climate change.
- Possible consequences of climate change for water birds are analysed through climatic niche models. A large-scale statistical modelling approach is applied using available data on the climatic and environmental factors determining the non-breeding distributions at sea of the relevant waterbirds in Northern European waters.
- The possible implications of climate change for marine benthic flora and fauna, fish, marine mammals, terrestrial and freshwater flora and fauna, coastal morphology and surface and ground water are addressed in a more qualitative manner based on literature and the outcome of the hydrodynamic and ecological modelling.
- Concerning human beings, soil (apart from coastal morphology), air, landscape, material assets and the cultural heritage, the implications of climate changes for the project related impacts are considered less relevant and are therefore not specifically addressed in the EIA.

The specific issues have been addressed in the relevant background reports.

3.3.17 *How to handle mitigation and compensation issues*

A significant part of the purpose of an EIA is to optimize the environmental aspects of the project applied for, within the legal, technical and economic framework. The optimization occurs even before the environmental assessment has been finalized and the project, which forms the basis for the present environmental assessment, is improved environmentally compared to the original design. The environmental impacts, which are assessed in the final environmental assessment, are therefore the residual environmental impacts that have already been substantially reduced.

Similarly, a statement of the compensation measures that will be needed to compensate for the loss and degradation of nature that cannot be averted shall be prepared. Compensating measures shall not be described in the impact assessment of the individual components and are therefore not treated in the background reports, but will be clarified in the Danish EIA and the German LBP (Landschaftspflegerischer Begleitplan), respectively.

In the background reports, the most important remediation measures which are included in the final project and are of relevance to the assessed subject are mentioned. In addition additional proposals that are simple to implement are presented.

3.4 *Deviations from the general assessment scheme*

The assessment methodology for water quality and plankton biology relies extensively on dynamic models, including Hydrodynamic Models (HD), Sediment Model (SM) and Water Quality (WQ) Models. In general, most steps in the impact assessment are an integral part of numerical models:

- Important pressures related to construction and operation period of tunnel and bridge (e.g. concentration of spilled sediment) are modelled dynamically in 3 dimensions; concentrations of spilled sediments are used to calculate light attenuation dynamically (i.e. dose-response between sediment concentrations and light attenuation - "Sensitivity"), which in turn affects the growth of phytoplankton (i.e. dose-response between light intensity and growth rate - "Sensitivity") and biomass (i.e. impact of dredging on phytoplankton - "Impairment"), benthic vegetation biomass (impact of dredging on seagrass and macroalgae - "Impairment") and indirectly, oxygen concentration at seabed (impact of dredging on water quality - "Impairment").
- Degrees of impairment are averaged over appropriate periods and the "Severity" of impairment is assessed using a 4-level criteria scale "Very high", High, "Medium", "Minor" defined by degree of deviation from baseline conditions after taking account of natural year-to-year variation. Impairments below "Minor" are considered as "Negligible").
- "Significance" of impairment is assessed by combining Degree or Severity of Impairment with area extension of impairment.

For potential impacts that cannot be modelled directly, i.e. when dose-response relationships are less well documented, appropriate model outputs are overlaid (time-step by time-step), to identify areas and duration where and when 2-to-several criteria is fulfilled. In the assessment of dredging-related impacts direct effects of suspended sediments on phytoplankton sedimentation and on zooplankton growth and survival such approaches are used.

Release of toxic substances and oxygen demand during dredging are assessed using Monte-Carlo analysis based distribution functions of dredging spill, of toxic concentration in sediments, of release rates and of current speed (i.e. dilution). Calculated concentrations are compared to EU Environmental Quality Standards (EQS) and Danish Water Quality Criteria (VKK). For oxygen, calculated oxygen demand are subtracted from background concentrations and the resulting concentration compared to internationally accepted criteria for minor, medium and high levels of oxygen deficiency (5.7 mg O_2/l, 4 mg O_2/l, 2.5 mg O_2/l). The impairment scale is further refined by taking account of duration of oxygen deficiency.

3.5 Magnitude of pressure

Table 3.5 shows an overview of the pressures, pressure indicators and methods used to assess the magnitude of pressure.

Table 3.5 Summary of methods used for assessment of magnitude of pressure.

Pressure	Component/ sub-component impacted	Pressure indicator	Methods
Suspended sediment	WQ: Secchi depth, inorganic nutrients, oxygen concentration Phytoplankton: chl a and biomass, species composition	Modelled reduction of light in water column	Light reduction was modelled based on sediment spill data from FEHY (2013b) Oxygen reduction based on oxygen demand estimated in (FEHY 2013d)
	Phytoplankton: facilitated sedimentation	Running average of suspended sediment conc.	Calculated based on modelled sediment spill data from FEHY (2013a)
	Zooplankton	Average suspended sediment concentrations the	Calculated based on modelled sediment spill data from FEHY (2013a)
Sedimentation	Zooplankton, resting eggs	Permanent deposition after end of construction	Extracted from modelled sediment spill data from FEHY (2013a)
Toxic substances	Water quality All biological components	Toxic substance concentration dissolved in the water	Data from sediment chemistry report (FEHY 2013d) and feasibility study (COWI-Lahmeyer 1998). Measured current speed 2010, release rates from literature.
Lost habitat	Plankton	Replaced water volume	Volume calculated from technical drawings and bathymetric maps. Plankton production and biomasses from FEMA baseline scenario

Pressure	Component/ sub-component impacted	Pressure indicator	Methods
Solid substrate	Plankton affected by blue mussels on piers Polyps of jellyfish	Area (m^2) of new solid substrate	Areas calculated from technical drawings
Hydrodynamic regime	General water quality and plankton	Modelled changes in stratification and oxygen concentrations at bottom	Modelled changes in hydrography (FEHY 2013b)

3.5.1 *Suspended sediment*

Light reduction due to suspended sediment from sediment spill was modelled using the FEMA model (see section 3.9 below) using modelled spill data from (FEHY 2013a).

Concentrations of spilled sediment and size distribution of particles in spill affect light attenuation, and thereby, the transparency of water and Secchi depth. Suspended solids differ in their optical properties, where the organic content, size distribution and shape of particles are important for the mass-specific light attenuation. The attenuation of light is the combined effects of two processes in the water column, namely the scattering of light and absorption of light. The scatter of light scales to cross-sectional area of particles (living and dead, inorganic), while the mass-specific scatter (b*) including a diffraction effect can be described by:

$$b^* = \frac{3}{D\rho_P}$$

where D is the diameter of a (spherical) particle and ρ_P is the density of the particle (see Annex D). Besides area, surface properties of particles such as their refractive index are important for the mass-specific scatter. The optical properties of fine sediments from the Fehmarnbelt were estimated in lab experiments (described in detail in Appendix D) and calculated for the four particle size classes modelled in spill scenarios (Table 3.6).

Table 3.6 Optical properties of different sized particles; mass-specific absorption (a – m2/g) and scatter (b – m2/g) coefficients and corresponding mass-specific attenuation coefficients (Kd – m2/g).

Sediment fraction Diameter (mm)	Reference (incl. attenuation due to dissolved substances) a	b	Specific Kd (m^2/g)
0.064	0.0278	0.354	0.057
0.028	0.0278	0.756	0.078
0.010	0.0278	1.814	0.117
0.0065	0.0278	2.714	0.142

Based on the same experimental data, two sets of absorption and scatter coefficients were developed (including and excluding absorption from dissolved substances originating from sediment pore water). In a test using the FEMA model for the tunnel dredging scenario (October 2014 – December 2015) the set of coefficients that included absorption from dissolved substances gave slightly (1-10%)

higher reductions in Secchi depth in areas affected by suspended sediment compared to a test using coefficients not considering dissolved substances.

In all FEMA modelling, the "+dissolved substances" set of coefficients was therefore used implying that the estimated impacts on water quality, plankton (and benthic vegetation) will be slightly conservative.

The light attenuation coefficient in the FEMA model is described by the Kirk formula:

$$Kd = [(aw+aal+ass+adoc+adc)^2 + c*(aw+aal+ass+adoc+adc)*(bal+bss+bdc)]^{0.5},$$

where aw, aal, ass, adoc, adc represent the absorption due to water itself, phytoplankton, dissolved organic matter and detritus, respectively, and bal, bss, bdc represent scatter caused by phytoplankton, suspended solids and detritus. The constant c was fixed at 0.256.

The combined effect of background constituents in water affecting attenuation and additional constituents from sediment spill was modelled dynamically as Kd in every model grid. Secchi depth was calculated as an integral measure of light attenuation according to:

$$\text{Secchi depth (m)} = 1.9/Kd$$

3.5.2 Hydrographic regime

Structure-related impact on the hydrographical regime by focussing on changes in vertical mixing (and strength of density stratification;) and in turn oxygen conditions in bottom waters was quantified using hydrodynamic and water quality modelling (FEHY 2013b) within the assessment area.

3.6 Sensitivity

Impact on water quality and planktonic components such as phyto- and zooplankton varies with type and magnitude of pressure, but also with the sensitivity of biological component. Sensitivity is assessed using documentation from the literature on the relationship between different pressures and the various water quality and plankton components. When dynamic models are used in impact assessments, sensitivity is implemented in model descriptions as "dose-response" relationships.

3.6.1 Suspended sediment

The sensitivity of phytoplankton production to light reduction is described by formulations between light and productivity embedded in the numerical FEMA model, see section 3.9 below.

Sensitivity of the composition of phytoplankton to light reduction is assessed using theoretical arguments on the interaction of light, temperature and nutrients on various groups of phytoplankton (see Chapter 5).

The sedimentation rates of phytoplankton cells increase through aggregation with other suspended particles (also called 'flocculation'). The probability and extent of aggregations are assessed based on literature data on critical phytoplankton concentrations for natural aggregation, the aggregated settling velocity and removal of plankton by artificial induced flocculation (described in detail in Chapter 5).

The efficiency of filter-feeding by zooplankton may decrease if high concentration of suspended sediments is present in the water column. The risk for and extent of impacts are evaluated by combining "dose-response" relationships from literature with modelled concentration of suspended sediments (see Chapter 5).

3.6.2 Sedimentation

If resting eggs of zooplankton are deeply buried under sedimented material they may lose viability and cannot contribute to recruitment of zooplankton stock after winter. Impacts are assessed by coupling the predicted sediment accumulation with literature information on critical burial depth and, taking account of advection of zooplankton from adjacent areas to replenish Fehmarnbelt stocks.

3.6.3 Toxic substances

Release of toxic substances is due to the pressure "suspended sediments", but toxic substances itself constitute a pressure for all biological components in the Fehmarnbelt. The EU Commission has set Environmental Quality Standards (EQS) for List 1 substances and Guidance values (water quality criteria) for List 2 substances. Besides, Denmark and other EU member states may have set national standards for List 2 substances. Generally, after initial dilution EQS must not be exceeded for any of the List 1 prioritised substances and for List 2 substances emissions should be minimised to ensure that water quality criteria are not exceeded.

3.6.4 Solid substrate

Effects of additional solid substrate on water quality, phytoplankton and jellyfish are assessed based on the area of new substrate, the settlement and fouling by macroalgae, mussels and ephyra stages (based on field studies carried out in the Fehmarnbelt and in nearby comparable areas), and how fouling organisms affect oxygen concentration in bottom water, biomass of phytoplankton by mussel filtration and jellyfish by increasing recruitment.

3.6.5 Hydrographic regime

Sensitivity towards changes in the hydrographical regime is implemented in the FEHY local model description (FEHY 2013b).

3.7 Assessment criteria

The principles used to define criteria for the assessment of impairments on water quality and plankton are listed in Table 3.7 and Table 3.8.

Table 3.7 Assessment criteria for water quality in the Fehmarnbelt area. Regional includes the Western Baltic Sea (see Figure 3.1).

Pressure	Criteria for degrees of impairment	Duration	Range	Degree
Suspended sediment (construction-related)	High to very high reduction in Secchi depth	Temporary	Regional (also outside local zone)	Very high
	Medium to high reduction in Secchi depth	Temporary	Regional (also outside local zone)	High
	Minor to medium reduction in Secchi depth	Temporary	Regional (also outside local zone)	Medium
	Only minor reduction in Secchi depth	Temporary	Regional (also outside local zone)	Minor
Toxic substances (construction-related)	Concentration must not exceed Environmental Quality Standards (EQS) or national water quality criteria	Temporary	Local	Case-by-case related
Oxygen consuming substances	Reduction in oxygen concentration should not lead to impacts on benthic organisms	Temporary	Regional (also outside local zone)	Case-by-case related
Hydrographical regime (structure-related)	Change in oxygen concentration should not lead to impacts on benthic organisms (increase in DO is not regarded as a negative impact)	Permanent	Regional (also outside local zone)	Case-by-case related
Oxygen production in bottom water (structure-related)	Increase in oxygen concentration in bottom water is not regarded as a negative impact	Premanent	Regional (also outside local zone)	Case-by-case related

Table 3.8 Assessment criteria for plankton in the Fehmarnbelt area.

Pressure	Criteria for degrees of impairment	Duration	Range	Degree
Suspended sediment (construction-related)	High to very high reduction in phytoplankton (Chl-a) due to reduction in light	Temporary	Regional (also outside local zone)	Very high
	Medium to high reduction in phytoplankton (Chl-a)	Temporary	Regional (also outside local zone)	High
	Minor to medium reduction in phytoplankton (Chl-a)	Temporary	Regional (also outside local zone)	Medium
	Only minor reduction in phytoplankton (Chl-a)	Temporary	Regional (also outside local zone)	Minor
Sedimentation (construction-related)	Quantitative/qualitative assessment based on available data	Temporary	Regional (also outside local zone)	Case-by-case related
Hydrographical regime (structure- related)	Quantitative/qualitative assessment based on available data	Permanent	Regional (also outside local zone)	Case-by-case related
Solid substrate (structure-related)	Jellyfish and plankton Quantitative/qualitative assessment based on available data	Permanent	Local	Case-by-case related

Key principles for criteria covering water quality and plankton ecology are outlined below:

- Numeric changes in key indicators that reflect the habitat requirements for resident plants and animals in the Fehmarnbelt including salinity, temperature, dissolved oxygen, water transparency, and primary production. Plants and animals are adapted to rather broad ranges in physical and biological variables, but variations outside the normal range can increase the "pressure" on biota affecting the viability of populations.

- Phytoplankton production is one of the highest ranked "ecosystem services" characteristic for coastal and offshore ecosystems fuelling the planktonic and benthic grazing food webs (Fletcher et al. 2011). Reduction in phytoplankton below the "natural" year-to-year variation can have immediate effects on the dominant mussel population along Lolland and around Fehmarn (and cascading effects on population of wintering eiders) and on zooplankton (important food for planktivorous fish).

- All physical, chemical and biological elements in the water column will vary from year-to-year driven by variation in land use (run-off), climate and meteorology. Populations of plants and animals in the Fehmarnbelt are adapted to such variability, but large deviations outside the natural range may affect populations. Plankton organisms have short generation times and except for vertical migration among copepods and some phytoplankton plankton organisms are transported passively with water. Therefore and in contrast to sessile benthic organisms, plankton organisms will not be exposed to sudden changes in salinity (and temperature), except for vertically migrating plankters.

- The natural variability is quantified based on: 20-years monitoring data from 2-3 stations in the Fehmarnbelt and, data from the Baltic Sea basins. Data analysed include temperature, salinity, different indices of water column stratification, Secchi depth, chlorophyll and bottom water oxygen.

Besides the criteria for water quality and plankton listed above a formal and overarching group of criteria for water quality and plankton biology is set by the Water Framework Directive (WFD; 2000/60/EC) and its sister (underlying) directives: (1) Dangerous Substances Directives (DSD; 76/464/EEC), (2) Shellfish Water Directive (SWD; 79/923/EEC) and, (3) Bathing Water Directive (BWD; 76/160/EEC).

Water Framework Directive (WFD) and Marine Strategy Framework Directive (MSFD)

One of the objectives of WFD is to 'prevent deterioration of the status of all bodies of surface water'. Although not explicitly said in WFD it is implied that not a single supporting physical-chemical and biological quality element are permitted to deteriorate to the ecological status dictated by the worst affected quality element (EEC2005). In effect, neither physical nor chemical quality elements such as concentration of nutrients and chlorophyll-a are permitted to increase and, Secchi depth and oxygen concentration should not decrease outside the normal range for extended periods as result of Fehmarnbelt construction works and of the permanent structures.

Toxic substances

The Dangerous Substances Directive (76/464/EEC) and its 'daughter' directives regulate discharges that are liable to contain dangerous substances and that go to inland, coastal and territorial surface waters. Dangerous substances are toxic substances that pose the greatest threat to the environment and human health.

The directive specifies two lists of Dangerous Substances. List I covers those which are most toxic, persistent, and which are able to accumulate in the environment. List II covers substances that are toxic, but less serious.

The directive requires that pollution by List I substances is eliminated and that pollution by List II substances is minimised. To do this, all discharges that are liable to contain dangerous substances must be reviewed by relevant authorities. The directive also specifies the requirements for environmental monitoring.

For dangerous substances, EU has set Community-wide 'guidance' (minimum) standards (EU 2008) which have to be met as part of the objective of achieving 'good chemical status' of surface waters, and pollution by the dangerous substances is defined as exceedance of these Environmental Quality Standards (EQSs). EQS defines a concentration in surface waters below which the substance will not have a polluting effect or cause harm to plants and animals, and if the concentration in the water is less than the EQS then pollution is supposed to be eliminated. In setting an EQS, detailed data on biological toxicity and the aquatic ecosystem need to be taken into account. Therefore, EQSs are likely to differ from region to region and from water type to water type and accordingly, member states are allowed to modify the EU-set EQS. EQS shall be peer-reviewed and can be changed if new information becomes available.

Bathing water quality

The Bathing Water Directive (BWD; 76/160/EEC) set uniform EU standards for meeting acceptable (good and excellent) bathing water quality based on presence of microbiological organisms presenting a risk to bathers' health. The Directive does also pay specific attention to the risks of proliferation of cyanobacteria and health risk from accumulation of macro-algae and/or phytoplankton. In the previous directive other environmental factors were also considered. They included turbidity of water, pH and aesthetic factors such as abnormal change in colour, visible film on the water and odour due to mineral oils. In general, the Directive states that visible impacts should be inspected and in case of posing a health risk to bathers they must be informed.

The greatest risk associated with the Fixed Link project for affecting the bathing water quality (*sensu* increasing concentration of coliform bacteria) along Lolland coast is related to eventually higher loads from the Rødbyhavn treatment plant or increased risk for a larger proportion of treated water ending at beaches following relocation of outlet. At present it is assumed that relocation of outlet will not reduce bathing water quality.

As bathers' choice of beaches is known to depend on turbidity, the risk of increased turbidity due to dredging, are assessed. To our knowledge only few studies (from New Zealand) do directly address bather's reaction to turbidity and this study is used as reference for the assessment.

Shellfish water quality

Because the main environmental requirements in the Shellfish Water Directive is covered by Dangerous Substances Directive and the Bathing Water Directive and production of mussels does not take place, specific assessment according to the Shellfish Water Directive will not be carried out.

The Shellfish Water Directive (DWD; 79/923/EEC) sets EU standards to water quality to ensure that growth conditions are suitable for shellfish (i.e. well-oxygenated waters) and especially that shellfish do not accumulate substances that are harmful to consumers. In addition to List 1 substances and several substances in List 2 (see 3.5.1 above), coliform bacteria and algal toxins are also included.

Neither Germany nor Denmark has designated production areas for shellfish in the vicinity of the project (see map for DK: www.foedevarestyrelsen.dk/ Kontrol/ Sadan_kontrollerer_vi/ Muslingeovervaagning/ Danmark/ Kort_over_produktions-omraaer/ Documents/ 1d5ad7c685d442c6b9ea71bc27ea3a5cSydsjælland5.pdf, for Germany: www.wrrl-mv.de/doku/2004/sg_fisch.pdf), meaning that neither bottom trawling nor line production of mussels and oysters are allowed. Besides, to our knowledge neither production nor trawling takes places in the adjacent Danish production areas 175 (at Nakskov) and 184 (east of Falster) (Wittrup Seafood A/S, pers. com). The closest German production areas are located in Flensborg Fjord and Eckernförder Bucht.

Degree of impairment has been defined for impact of the pressure suspended sediment on the water quality sub-components Secchi depth (incl. bathing water), oxygen concentration in bottom water and, the plankton subcomponent chlorophyll-a.

For all other pressures the impairment is very low and assessed to be lower than lower limit of a possible minor category of the degree of impairment.

In this chapter the grades for degree of impairment are described for the mentioned pressures and secondly, the methods for assessing the degree of impairment are described.

The detailed criteria applied for the water quality and plankton components are described below.

Toxic substances

For toxic substances the national standards has been used to assess the degree of impairment.

For marine waters Germany has adapted the EU-published EQS (Hillenbrand et al. 2006), while Denmark additionally has set provisional standards for List II substances and in several cases has set lower standards than EU (Naturstyrelsen 2010). Although EU's EQS for annual average and Danish "general" VKK cannot be compared directly because of different averaging periods (EU – annual average; Danish – average over periods of discharge), it is obvious that Danish standards in several cases are lower than corresponding EU standards, especially for Nickel (Table 3.9). Neither EU (including Germany) nor Denmark has set EQS's for PCB's.

Table 3.9 EU's Environmental Quality Standards and National Danish standards for Priority Substances and other toxic substances analyzed in Fehmarnbelt sediments. [a] "added concentration" denote the additional concentration related to discharge

Substance (µg/l)	EU/GER		Danish (draft)	
	Annual avr.	Max allowable	$VKK_{General}$	VKK_{Max}
As	-	-	0.11[a]	1.1[a]
Ag	-	-	0.2[a]	1.2[a]
Cd	0.2	0.45	*EU*	*EU*
Cr_{III}	-	-	3.4	124
Cr_{IV}	-	-	3.4	17
Hg	0.05	0.07	*EU*	*EU*
Ni	20[b]	-	0.23[a]-3	6.8
Pb	7.2	-	0.34	2.8
Zn	-	-	7.8[a]	8.4[a]
TBT	0.0002[c]	0.0015[c]	*EU*[c]	*EU*[c]
Total DDT	0.025	-	0.002	-
p,p-DDT	0.01	-	*EU*	-
HCB	0.01	0.05	*EU*	*EU*
Napthalene	1.4	-	*EU*	-
Acenaphthalene	-	-	0.39	3.8
Acenapthylene	-	-	0.13	3.6
Anthracene	0.1	0.4	*EU*	*EU*
Dibenz(a,h)anthracene	-	-	0.00014	0.018
Benz(a)anthracene	-	-	0.0012	0.018
Benzo(a)pyrene	0.05	0.1	*EU*	*EU*
Benzo(b)fluoroanthene	0.03	-	*EU*	-
Benzo(k)fluoroanthene	0.03	-	*EU*	-
Benzo(g,h,i)perylene	0.002	-	*EU*	-
Chrysene	-	-	0.0014	0.014
Phenanthrene	-	-	1.3	4.1
Pyrene	-	-	0.0017	0.023

[a] added concentration; [b] was changed in 2006 from a previous value (1.7 µg/l); [c] draft values from EU, GER, DK)

Secchi depth

Secchi depth in the Femernbelt varies within and between years driven by variation in meteorology, run-off, nutrient inputs and Secchi depth variation in adjacent waters, especially in the western Baltic Sea. Between-year variation in Secchi depth in historical data (1984-1997) expressed by standard deviation (SD) is comparable to the between-year difference in baseline data from 2009 and 2010 at 0.3-1.2m (Table 3.10). Scaled differences of baseline data and scaled SD (= coefficient of variation, CV) of historical data were almost similar at 12% and 9% and therefore, a 10% variation from an annual average can be seen as a natural year-to-year variation.

Table 3.10 Yearly average and Standard Deviations (±SD) of Secchi depth (in meters) in the Fehmarnbelt assessment area. Data from FEMA-FYHY (2013).

Sub Area	1984-1997	2009	2010	I2009-2010I
Darss Sill area	6.5 ± 0.7	7.6 (±0.2)	7.2 (±0.6)	0.4
Mecklenburg Bight		7.5 (±0.4)	6.7 (±0.3)	0.8
Fehmarnbelt	6.2 ± 0.8	7.8 (±0.6)	6.6 (±0.4)	1.2
Great Belt		7.2 (±0.3)	6.9 (±0.3)	0.3

A deviation (reduction) of 10% (half of the "natural" yearly variation) from the average Secchi depth is used to discriminate between negligible and minor impairment in the Fehmarnbelt for impacts related to construction phase. Boundaries for increasing degrees of impairments are -20%, -30%, and -50% reductions in Secchi depth (Table 3.11).

Table 3.11 Grades of degree of impairment used in the assessment of impacts on Secchi depth due to suspended sediment in the Fehmarnbelt (construction phase - short-term) and permanent impacts in the Fehmarnbelt and in the Baltic Sea resulting from structures (bridge alternative). Short-term criteria are used to evaluate impairments on a yearly basis during the construction period.

WQ component		Criteria				
		Very high	High	Medium	Small	Negligible
Secchi	Short-term	> 50%	30-50%	30-20%	20-10%	< 10%
Depth	Permanent	> 20%	4-20%	2-4%	1-2%	< 1%

Another set of criteria was developed for permanent impacts resulting from structures. Besides Fehmarnbelt these criteria do also apply in the Baltic Sea. Lower boundary (negligible) at 1% and higher class boundaries were set using statistics on Baltic Sea data.

Oxygen
The Fixed Link project can affect oxygen concentration in bottom waters by four different mechanisms:

- Construction-related by release of reduced substances from sediments during dredging
- Construction-related and mediated through reduction in benthic primary production and oxygen production caused by shading from sediment spill
- Structure-related by increase of vertical mixing and thereby increase transports of oxygen to bottom water
- Structure-related increase in oxygen production below pycnocline by macroalgae populating piers, pylons and scour protection

Deep waters
The deep parts of Fehmarnbelt experience regular hypoxia during late summer and early autumn, primarily due to a combination of oxygen uptake in sediments (Rasmussen et al. 2003b) and, a long residence time and because the advected bottom water from the Southern Kattegat and the Great Belt already have low oxygen concentration when entering the Fermarnbelt (Rasmussen et al. 2003a). The latest recorded event occurred in 2010 when intrusion of hypoxic and anoxic water bottom could be followed from the Great Belt in late July and reaching the Darss Sill

area in early September (FEMA-FEHY 2013). Therefore, benthic invertebrates living in deep water habitats must be adapted to regular hypoxia events of week-to-weeks duration. Therefore, the criteria delevoped for project-related reductions in oxygen concentration will be based on statistical reduction in concentration, rather than sensitivity of the benthic organisms.

At deeper water in Fehmarnbelt below the pycnocline benthos are exposed to hypoxia on a yearly basis, but anoxia as recorded in 2010 is rare (Figure 3.6). The critical period occurs in early autumn when the oxygen level reaches an average saturation of 40% (≈ 2.6 mg O_2/L). It is assumed, that the deep water communities are comparatively tolerant to hypoxia and, reductions by 0.15, 0.3 and 0.6 mg O_2/l are set as boundaries for minor, medium, high and very high degree of impairment (Table 3.12). If oxygen demand from dredged sediments exceeds concentration and flux of oxygen the deficit can be represented as "negative" oxygen concentration, i.e. an insufficient oxidation of H_2S.

Table 3.12 Reductions in dissolved oxygen to assess degree of impairment for oxygen concentration in near-bed water below pycnocline in the Fehmarnbelt. Unit for oxygen reductions (DO-red) is mg O_2/l

Very high	**High**	**Medium**	**Minor**	**Negligible**
DO-red > 1.5	1.5 > DO-red > 0.6	0.6> DO-red > 0.3	0.3> DO-red > 0.15	0.15 > DO-red

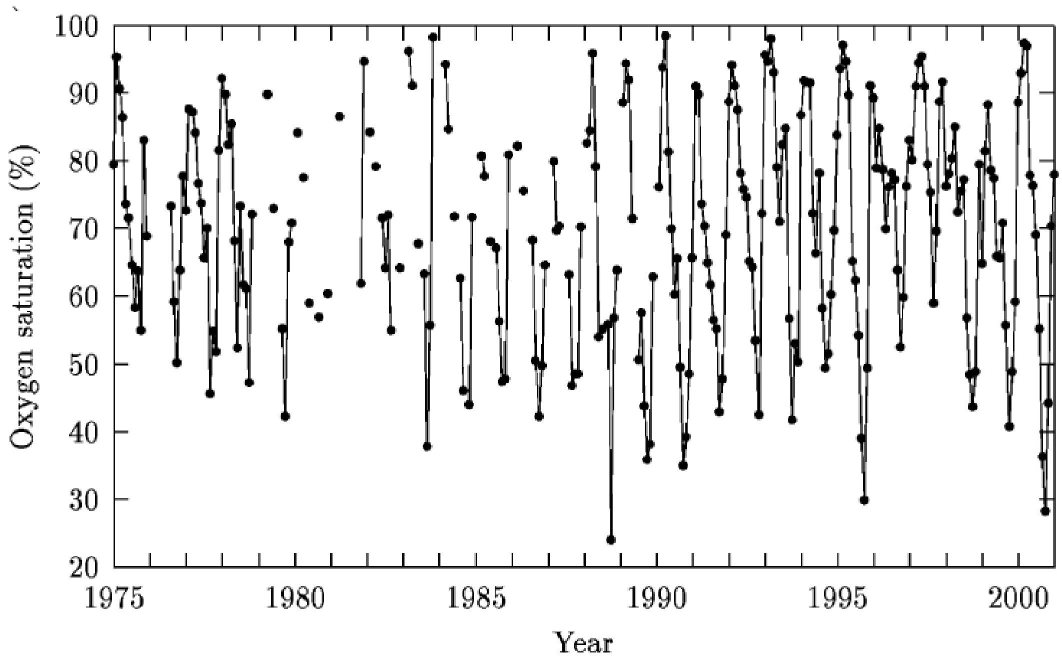


Figure 3.6 Oxygen saturation (%) in bottom water (20-40 m) at the entrance to the Baltic Sea (i.e. west of Darss Sill – Great Belt and Fehmarn Belt). Within a year minimum concentrations are reached in late August to October. From Rasmussen et al. (2003b).

Shallow waters

In contrast to deep waters, benthic communities living above the pycnocline, i.e. in shallow waters, will rarely experience hypoxic conditions and are presumably less tolerant to hypoxia than the communities occurring at large depth in the Fehmarnbelt. Although not a strict proof, the baseline investigation of the benthic fauna of Fehmarnbelt showed that the percentage oxygen-sensitive taxa such as crustaceans to total species number was markedly higher in communities located above pycnocline (at 27 ± 4%), than below pycnocline (at 19 ± 3%) (FEMA 2013b). This supports that benthic communities may be shaped according to local variation in oxygen availability, and that shallow water communities probably are not adapted to hypoxic conditions Therefore, oxygen criteria for shallow water organisms primarily build on information of species-related sensitivity to hypoxia.

Benthic vegetation and especially different macrofauna species have different "critical levels" of hypoxia which they can physiologically survive. In general, molluscs and polychaetes are the most tolerant, followed by echinoderms, crustaceans and then fish (Diaz and Rosenberg 1995, Modig and Olafsson 1998). Temperature does influence tolerance (decreases with increasing temperature due to higher oxygen demand in organisms and due to decreasing oxygen saturation in water with increasing temperature) affecting the seasonal sensitivity, but species-specific differences are by far more important. The "traditional" percived critical oxygen concentration of 2 mg/L causing significant effects on benthic communities (e.g. Pihl 1994), a mass mortality when 0.5 mg O_2/L is reached, and an oxygen concentration of 4 mg/L causing behavioural changes such as avoidance in fish (Pihl et al. 1991, Gray et at. 2002, Diaz and Rosenberg, 1995) may not be adequate to describe the onset of impacts in all except the most commonly laboratory held organisms or organisms subject large natural variability in environmental conditions.

In a recent comparative study based on data from more than 40 publications the sensitivity of different benthic taxonomic groups to reduced oxygen concentrations and the influence of duration of sub-oxic concentrations were quantified in terms of medians and 90-percentiles for lethal and sublethal effects and, the influence of exposure time on lethality (LT_{50}) was calculated (Vaquer-Suyer and Duarte 2008). One set of oxygen criteria for shallow waters have been developed using the information in (Vaquer-Suyer and Duarte 2008).

The scale of degree of impairment build on the threshold concentrations calculated by Vaquer-Suyer and Duarte (2008) is:

- Concentrations above 5.7 mg O_2/L will protect 90% of species within the most sensitive taxonomic group (crustaceans) against lethal effects. Given the variable environmental conditions, highly sensitive species within the crustacean group probably cannot maintain a population in the Fehmarnbelt due to the flucturing salinity. As crustanceans are the most sensitive of the taxonomic groups found in the Fehmarnbelt, a minimum value of 5.7 mg O_2/L will protect all species in the Fehmarnbelt.

- 4 mg O_2/L allows most species living in habitats regularly exposed to hypoxia to maintain viable populations. Vaquer-Suyer and Duarte (2008) calculated a median sublethal effect concentration at 4.41 mg O_2/L for fish, a median sublethal effect concentration at 3.21 mg O_2/L for crustaceans, and a 90-percentile lethal concentration 3.43 mg O_2/L for bivalves (protecting 90% of species).

- A boundary concentration 2.5 mg O_2/L (approx equivalent to 2 ml O_2/L) will protect 90% of fish species against lethal effects resulting from short exposures, 2 and 2.83 mg O_2/L are the median and 90th-percentile for sublethal effects in molluscs

Table 3.13 depicts a matrix to assess the degree of impairment for oxygen in near-bed waters in shallow areas without a pycnocline (i.e. water depths less than 14-18m).

Table 3.13 Matrix to assess degree of impairment for oxygen concentration in bottom water above pycnocline in the Fehmarnbelt. Unit for concentration of dissolved oxygen (DO) is mg O_2/l.

Duration (days)	DO < 2.5	2.5 < DO < 4	4 < DO < 5.7	5.7 <DO < saturation
10-∞	Very high	Very high	High	Negligible
3-10	High	High	Medium	Negligible
1-3	Medium	Medium	Minor	Negligible
< 1	Medium	Minor	Negligible	Negligible

Bathing water
The Bathing Water Directive mainly considers bacteriological quality of bathing waters but also the risk for accumulation of algae including blooms of cyanobacteria. Due to lack of information of changes in loads of coliform bacteria we cannot assess the risk for affecting bathing water quality by construction-related increase in bacterial concentration.

Another measure of quality of bathing waters and directly perceived by bathers is the clarity of water. Clarity or the inverse measure, turbidity was included as a quality parameter in the previous Bathing Water Directive. Given alternatives bathers will always prefer beaches where the water is clear and the bottom visible. Clear water is intuitively associated with water not being polluted. In microtidal and comparable wind protected waters such as the Fehmarnbelt bathers are used to clear-water beaches and it is questionable if they will accept major reductions in clarity of water.

Canadian authorities suggested an acceptable limit of clarity equivalent to a Secchi depth of 1.2m (Environment Canada 1972). In three well-documented studies Smith and co-workers analysed New Zealand bathers (interview on beaches) and water quality experts' perception of acceptable water quality for bathing. They found that a clarity level of 1.2m which is equivalent to a Secchi depth of 1.5m on average was required by bathers before they found waters suitable for bathing. Also, if 90% of people on beaches should accept bathing then a Secchi depth of approximately 2.75 m was required (Smith et al. 1991, Smith DG and Davies-Colley 1992, Smith et al. 1995). Based on poor statistics at high water clarities (few and conflicting responses) the authors concluded that water with Secchi depths larger than 2.75-3.0m will not increase bathers' choice of beaches. Including the possibility that bathers in the Fehmarnbelt region may be more critical than their colleges in New Zealand a value of 5m has been defined to delineate a negligible degree of impairment.

Table 3.14 below depicts a matrix to assess the degree of impairment for bathing water. In lack of Danish, German or European studies the New Zealand findings was adopted but also considered the duration, i.e. the % of beach days (1 June – 31 August) that Secchi depth is below 1.5, 2.75 and 5m.

Table 3.14 Matrix to assess degree of impairment for water clarity (equivalent Secchi depth) along beaches in Fehmarnbelt. Duration refers to % of time during bathing season (1 June – 31 August) Secchi depth boundaries was exceeded.

Duration (%)	Secchi depth (m) <1.5	1.5-2.75	2.75-5	>5
100	Very High	High	Medium	Negligible
50	High	Medium	Minor	Negligible
25	Medium	Minor	Negligible	Negligible
5	Minor	Negligible	Negligible	Negligible

Phytoplankton
Phytoplankton biomass is a lump measure of the sum of individual biomass belonging to multiple species (> 100). All phototrophic phytoplankton contain the pigment chlorophyll-a and concentration of chlorophyll-a has been used for decades as an easy and robust measure of phytoplankton biomass. Long time series of chlorophyll-a measurements are available for the Fehmarnbelt and adjacent waters.

The significance of impairment of phytoplankton biomass (and production) is evaluated based on the importance for the ecosystem services in the Fehmarnbelt. The Fehmarnbelt supports a large population of the common mussel that in turn constitutes the basis for an internationally important population of eiders that feed on mussels during the winter. Hence, besides supporting zooplankton, phytoplankton production indirectly supports the large eider population.

Natural variation in chlorophyll-a
Concentration of chlorophyll-a varies between years driven by variation in meteorology, local run-off and nutrient inputs and variation in adjacent waters, especially variation in the western Baltic Sea.

Chlorophyll-a in the Fehmarnbelt area during summer has been stable during the past 20 years, but with slightly different concentrations between subareas. West of the alignment (St 12 and St 46; both are long-term monitoring stations under the HelCom program, see Figure 3.7) summer (May through August) concentration varied ± 0.5µg chlorophyll-a/l around long term averages of 2.03 µg chlorophyll-a/l (St 12) and 1.64 µg/l (St 46) (see Figure 3.8). Although neither of the two stations can be regarded as fully representative of the conditions in the central Fehmarnbelt, it is reasonable to assume that a comparable year-to-year variation (in percentage of the long-term average) exist in the alignment area.

March through October constitutes the main growth season for phytoplankton in the Fehmarnbelt (FEMA-FEHY 2013), and this period is used to quantify impacts on phytoplankton. This leaves some uncertainty regarding between-year variation based on summer values and values averaged over the main growth season. In the Great Belt (Station P00, see Figure 3.10) the between-year variation in chlorophyll-a (expressed as coefficient of variation, CV: SD/average) was similar irrespective of avering periods (March-October: 17.6%; April-September: 17.0%; May-August: 16.2%). Therefore, we assume that the natural variation in summer values will be representative of the natural variation in chloroplyll-a averaged over the entire growth season in Fehmernbelt.

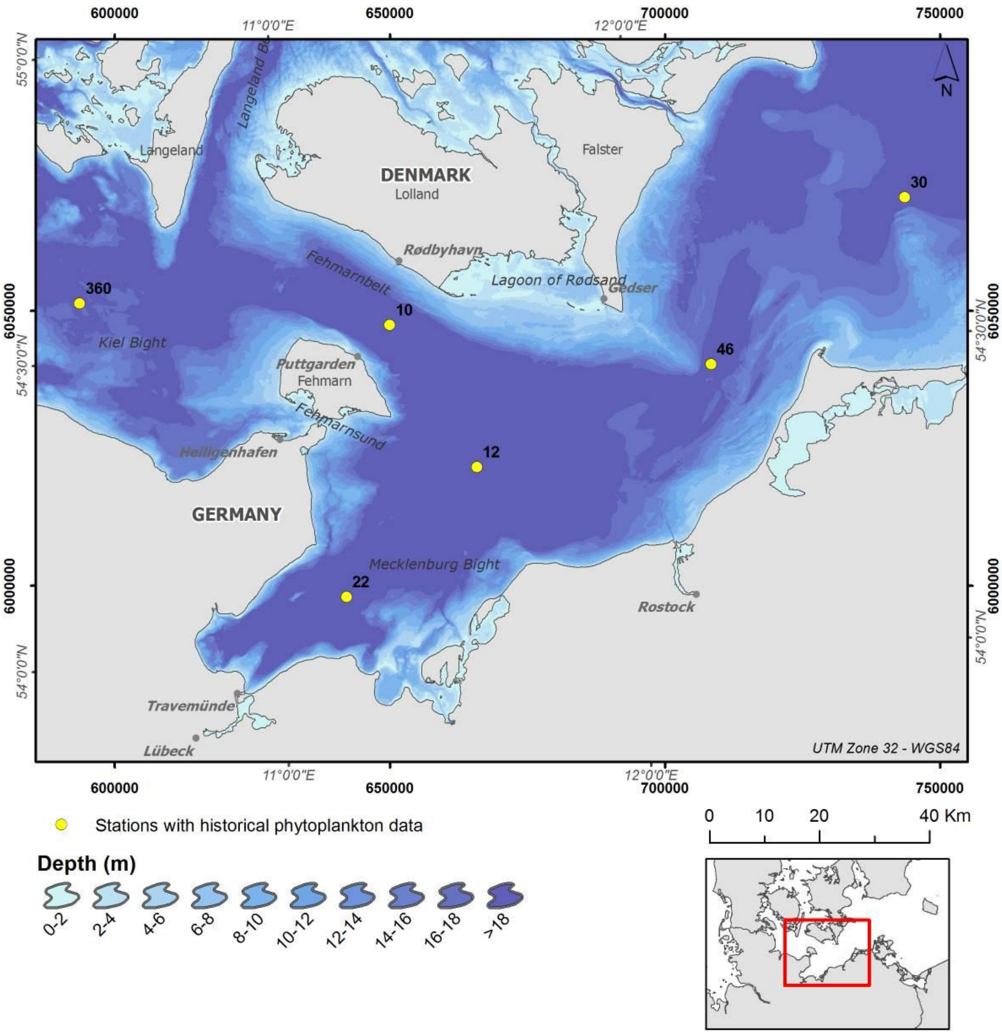
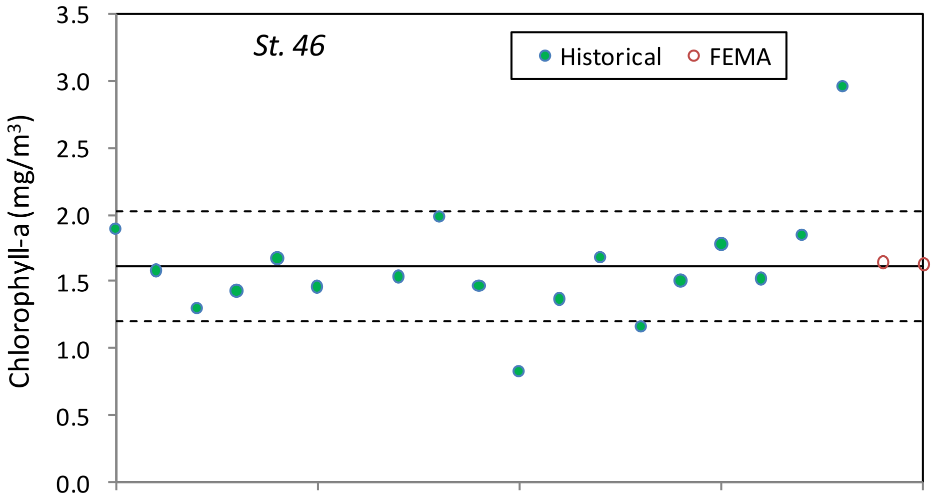


Figure 3.7 Geographical position of long-term phytoplankton and chlorophyll-a stations (360, 10, 22, 12, 46, 30).

Considering the rather large variation in summer concentration during the past 20 years (Figure 3.8) it seems reasonable to assume that a deviation in chlorophyll-a of 5% (e.g. caused by dredging) hardly will be important for the higher trophic levels. Hence, in the modelling of degree of impairment a deviation of ± 5% from the baseline will be regarded as negligible.



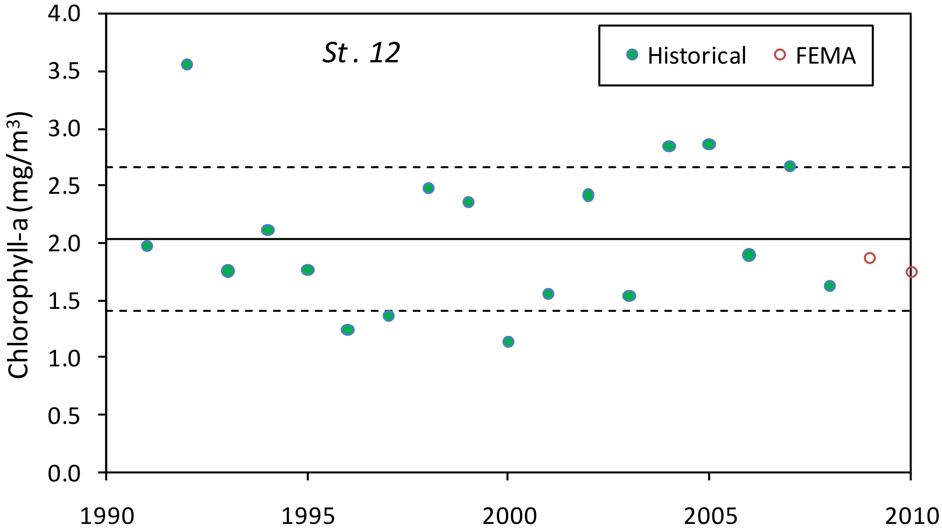


Figure 3.8 Concentration of chlorophyll-a in 0-10 m averaged over summer (May-August) from Helcom monitoring (historical data) and from FEMA baseline (2009-2010). A typical range in concentrations is defined as Chl-a_{avr} ± SD (summer average: full line – ±SD: dashed lines)

Table 3.15 summaries the grades of degree of impairment of chlorophyll-a and zooplankton biomass

Minor level of impairment is defined by the 50% of the coefficient of variation (0.5*SD/Chl-$a_{avr.}$) which - based on long-term data from St 12 and St 46 (Table 3.15) - is equivalent to a reduction of 5% compared to baseline levels. Assuming a 'baseline level' of 1.8 mg Chl-a/m^3 the growth rate in copepods (i.e. *Acartia tonsa*) will be reduced by up to 10% assuming that phytoplankton constitutes the only food source.

Medium level of impairment is defined by the interval -10% of baseline< Chl-$a_{baseline}$ < -20% of baseline conc. Growth rate in *Mytilus* will potentially be reduced between 10% and 25% (compared to the growth rate at a 'baseline level' of 1.8 mg Chl-a/m^3), and copepod growth rate may be reduced by up to 40%.

High level of impairment is defined by the interval -20% baseline conc. < Chl-a < -50% of baseline conc. In this interval, individual *Mytilus edulis* can maintain a positive energy balance, but growth rate will be very low and condition (meat content) will approach 'point of no return'. Copepods such as *Acartia tonsa* have a higher maintenance food concentration at 0.85 mg Chl-a/m^3 (Kiørboe at al. 1985), and populations are expected to just be sustained at this level of impairment.

Very high level of impairment are expected to affect mussel populations provided that chlorophyll-a concentration below 0.85 mg Chl-a/m^3 persist for extended periods (month – months). Copepods will starve at phytoplankton concentrations below ca 0.85 mg Chl-a/m^3 and because of limited storage capacity copepods such as *Acartia* will not be able to survive for more than 4-8 days without food (Calbet and Alcaraz 1997), while *Pseudocalanus* probably can survive for weeks due to larger storage capacity. However, recruitment of copepods will take place continuously by advection from non-affected areas; hence impacts mediated through reductions in food will be much less important than for mussels, because they can only be replenished by the yearly reproduction.

Table 3.15 Grades of degree of impairment used in the assessment of impacts on plankton (chlorophyll-a and zooplankton) due to suspended sediment.

Component	Criteria				
	Very high	High	Medium	Minor	Negligible
Chlorophyll-a Zooplankton	> 50%	20-50%	10-20%	5-10%	< 5%

3.8 Assessment of degree of loss

Volume occupation caused by permanent structures such as bridge piers and pylons, protection reefs and land reclamation areas inevitably will reduce the total production and biomass of plankton in the Fehmarnbelt. The loss is quantified in terms of summed production and summed biomass within the areas planned to be occupied by structures. The loss is structure-related and will persist in the entire life of the Fixed Link.

In comparison, water quality will not be affected by volume and area loss because the parameters defining the status of water quality always are expressed per unit, distance (Secchi depth), or concentration (e.g. mg O_2/l).

3.9 Assessment of degree of impairment

This section gives an overview of the assessment strategies used in the impact assessment, and an overview of the models used.

3.9.1 Overview of assessment strategies

An overview of assessment strategies is shown in Table 3.16.

Table 3.16 Overview of assessment methods for different pressures and impacts

Pressure	Component/sub-component	Impairment Assessment Method
Suspended sediment (construction-related)	WQ: Secchi depth Plankton: Primary production, chlorophyll-a, oxygen concentration, zooplankton production	Numerical modelling for full construction period
Suspended sediment (construction-related)	Plankton: phytoplankton composition, facilitated sedimentation, zooplankton consumption and survival	Numerical modelling: Overlay of sediment conc. & phyto- and zooplankton conc.
Sedimentation (construction-related)	Plankton: Zooplankton production and biomass – by burial of copepod resting eggs	Numerical modelling: Overlay of sediment conc. & zooplankton conc.
Solid substrate (structure-related)	WQ: oxygen production (macroalgae) Plankton: Jellyfish (recruitment), reduction in phytoplankton (blue mussels)	Quantitative /qualitative
Hydrographical regime (structure-related)	WQ: Secchi depth and oxygen Plankton: Chla-a	Numerical modelling for a typical year (2005)
Hydrographical regime (structure-related)	Cyano-bacteria bloom	Quantitative /qualitative

Three types of assessment approaches are used to assess the degree of impairment:

- Numerical modelling is used to quantify impacts related to shading effects of sediment spill on water quality (nutrient concentrations, chlorophyll-a, Secchi depth, oxygen concentration) and plankton ecology (primary production and zooplankton biomass). The approach combines magnitude of pressures, sensitivity, impacts, recoverability as models are run for the entire construction period.

- Numerical models are also use to quantify structure-related impacts such as changes in hydrodynamic regime, including changes in water column stratification, vertical mixing and associated changes in bottom water oxygen concentration.

- A combination of quantitative and qualitative assessment are used to

    - evaluate the impacts of sediment spill on composition of phytoplankton and survival of zooplankton. Numerical data on modelled sediment concentrations in the water is combined with literature on sediment concentrations and "dose-response" relationships between zooplankton and sediment concentration.

    - evaluate the impacts of sediment spill on the risk and magnitude for additional sedimentation of phytoplankton along with sedimentation of spilled sediment and potential impacts of reduced recruitment of zooplankton due to burial of resting eggs. Modelled sedimentation rates (from spill) are combined with literature information on facilitated phytoplankton sedimentation and effects of sediment cover on hatching success of eggs.

    - evaluate the impact of additional substrate including assessment of risks of additional recruitment of jellyfish due to settlement of polyps, estimating reductions in chlorophyll-a around bridge structures due to establishment of mussels on structures, and estimating additional oxygen production below pycnocline from macroalge developing on pylons and pillars.

    - evaluate the impacts of changes in hydrography on risks for cyanobacterial bloom. Modelled changes in stability of water column are combined with habitat requirements for cyanobacteria.

Impact from sediment spill
Impacts from sediment spill are partly assessed using numerical modelling; meaning that assessment of magnitude of pressure, sensitivity and the resulting effects on water quality and plankton are done in one integrated process as all elements are embedded in the modelling. The status of water quality and plankton is highly dynamic in the Fehmarnbelt because of large exchange with the adjacent waters, a direct influence of meteorology and also short generation times of plankton organisms. Therefore, the expression of fixed link-related pressures will be highly variable both what concerns the direction and spatial extension of impacts but also the magnitude of impacts. An example is given below.

Impacts from sediment spill are partly assessed using numerical modelling; meaning that assessment of magnitude of pressure, sensitivity and the resulting effects on water quality and plankton are done in one integrated process as all elements are embedded in the modelling. The status of water quality and plankton is highly dynamic in the Fehmarnbelt because of large exchange with the adjacent waters, a direct influence of meteorology and also short generation times of plankton organisms. Therefore, the expression of fixed link-related pressures will be highly variable both what concerns the direction and area-extension of impacts but also the magnitude of impacts. An example is given below (Figure 3.9).

The sensitivity of water quality and plankton to temporary pressures varies over seasons but also on much shorter time scales. During periods of strong winds and passing low-pressure fronts the pycnocline is eroded several meters and with 3-4 days delay easterly surface current speed may increase up to 10 times from an average speed. In the example in Figure 3.9 a low speed of 0.06 m/s was measured from 10-14 September 2010. An average current speed recorded on 16 September 2010 of 0.8 m/s is equivalent to a displacement of surface water by ca. 50 km per day eastwards. In effect, the Fehmarnbelt planktonic communities can be replaced by communities representing the Great Belt and the Southern Kattegat in few days.

Besides the replacement of plankton communities a ten-time increase in current speed will lead to much lower concentrations of sediment spill (caused by increased dilution) but also a much larger extension of the spill plume during dredging activities. These highly dynamic conditions are encompassed in the dynamic model.

For other and less straightforward relationships between the pressure and impacts on water quality and plankton numerical modelling cannot be used with confidence and, other approaches must be applied in the assessment.

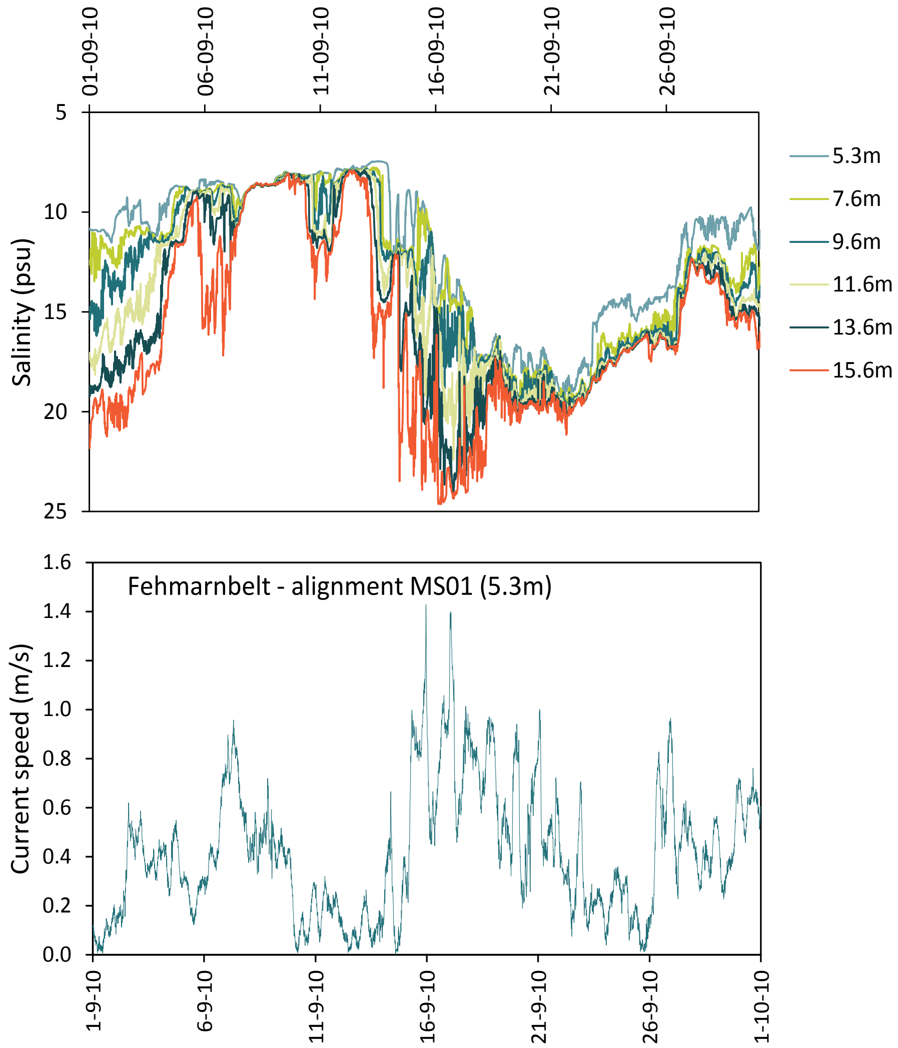


Figure 3.9 Salinity at 6 depths and current speeds of surface waters in the central Fehmarnbelt during September 2010. Peaks in current speed (eastwards into the Baltic) on 15-18 September was an effect of two low pressure fronts passing through Southern Scandinavia from 6 to 12 September 2010.

3.9.2 Overview of numerical models

Based on a large assessment of published studies Foden et al. (2008) recently reviewed current methodologies applied in aquatic EIAs summarizing advantages and disadvantages of the different approaches. If logistic constraints (e.g. computing power and expertise) can be overcome, the evaluation showed that dynamic models provide strong tools to support EIA and the down-stream decisions.

Dynamic models differ from "step-wise" assessment methods (i.e. map overlays each describing pressures, sensitivity etc.) by taking account of multi-directional forcing and feedback mechanisms between the biota, environment and anthropogenic factors. Dynamic models include well-documented dose-response relationships between sediment spill and light attenuation, between light attenuation and phytoplankton growth and, between phytoplankton biomass and zooplankton production. Therefore, direct and indirect impacts of dredging works can directly be quantified using such models.

Besides, dynamic models may take account of cumulative effects, e.g. additive effects where several pressures acting at the same time or interactive accumulation effects where multiple activities accumulate non-linearly; i.e. causing lesser or greater effects than the sum of their parts. An example of the latter is that sediment spill leads to reduction in primary production and algal growth due to shading effects, but at the same time uptake of nutrients from water and sediments is lower. In effect, when sediment spill settles and light becomes available algal growth may be larger because higher concentrations of nutrients are present. Using traditional overlays of sediment concentration (or light maps) and of phytoplankton distribution impacts will be overestimated.

Two types of numerical models were applied:

- The FEHY local models (MIKE 3-FM and GETM) cover the western Baltic, the Belt Sea and the southern Kattegat (see map Figure 3.10). The primary aim of this modelling is to quantify permanent changes in hydrodynamics and water quality in the western Baltic due to structures and reclamations. The models use the typical year 2005 boundary conditions as basis for the simulations.

- The FEMA model covering the the same area as the FEHY local models. The model builds on the FEHY local MIKE model with identical process descriptions for water column and sediment processes, except for phytoplankton where the FEMA model include one group for phytoplankton (versus differentiation into three functional groups in FEHY model). Opposite to the FEHY local model, the FEMA model also includes specific descriptions of filter-feeding benthos (i.e. mussels), eelgrass and macroalgae (three different groups with different light requirements). The FEMA model are used to predict temporary impacts (primarily related to dredging) of the Fehmarnbelt Link construction works. The model is executed for the entire construction period, using year 2005 as hydrodynamic basis for all construction years.

In addition to models mentioned above, larger scale models (FEHY regional models) that cover the entire Baltic Sea, the Belt Sea, Kattegat and the Skagerrak were applied to quantify impacts on hydrodynamics and water quality in the Baltic Sea related to structure of bridge or tunnel solution. Scenario results from the FEHY regional model are not used is this report.

The basic characteristics of the hydrographical models are described in (FEHY 2013b) and the FEMA model is described in Appendix A.

Uniform and widely accepted approaches or guidelines on how objectively to evaluate the performance of water quality and ecological models do not exist. If considered in reported studies, scatter-plots between observations and ecosystem model prediction are presented in about 25% published aquatic ecosystem models (Arhonditsis and Brett 2004, Arhonditsis et al. 2006), but strict and comparable evaluation methods are rarely applied, probably due to lack of accepted guidelines. (Allen 2009) suggested a series of univariate and multivariate indices to be used when evaluation aquatic ecosystem models.

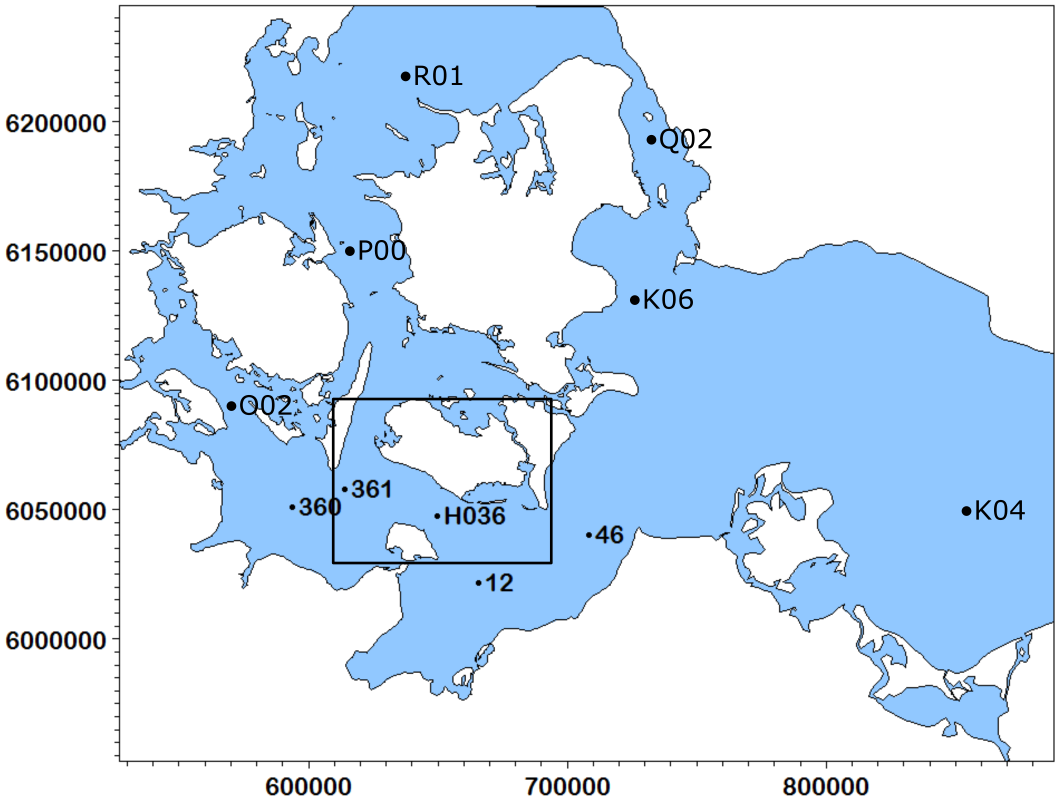


Figure 3.10 *Modelled area for the local FEMA ecological model; Station data from K04, K06, Q02, R01, P00, Q02 were used for calibration (2005) and, station data from360, 361, H036, 12 and 46 were used for validation (2009-2010 Baseline). Rectangle delineates the assessment area*

The performance of FEMA model used for impact prediction was evaluated using three indices suggested by (Allen 2009) to quantify agreement between observations and model prediction of nutrients, secchi depth, oxygen and chlorophyll-a:

- The *regression coefficient* R^2 expresses to what extent that the model can explain variation in observations.

- The *Nash Sutcliffe Model Efficiency ME*; (Nash and Sutcliffe 1970) of a state variable in a model is a measure of the ratio of the model error to the variability of the data. Originally *ME* was developed to assess the performance of river catchment models, which show a comparable variability to water quality and plankton (i.e. rapid season increases and decreases).

$$ME = 1 - \frac{\sum_1^N (O_i - P_i)^2}{\sum_1^N (O_i - \bar{O})^2}$$

where O_i is the observations, P_i the corresponding model estimate and the O-overbar indicates the average of observations, N is the total number of model data corresponding to observations and, *i* is the i^{th} comparison. By squaring the error the range is exaggerated separating good models from poor models. Following (Marechal 2004) performance levels are categorised as >0.65 excellent, 0.65-0.5 very good, 0.5-0.2 good, and < 0.2 poor (Marechal 2004 in Allen 2009).

- The *Percentage Model Bias* (the sum of model error normalized by the observations) is given by:

$$P_{bias} = \frac{\sum_1^N (O_i - P_i)}{\sum_1^N O^i} * 100$$

and provides a measure of whether the model is systematically under-estimating or over-estimating the observations. The closer the value is to zero the better the model. Performance levels are categorised as follows |Pbias| < 10 excellent, 10-20 very good, 20-40 good, > 40 poor (Marechal 2004).

Table 3.17 summarises the performance (validation) of FEMA water quality and plankton model based on 3 performance indices applied to inorganic nutrient concentrations, chlorophyll-a and dissolved oxygen in bottom water collected during baseline study (March 2009 – 1 June 2010).

Table 3.17 Performance of FEMA model quantified by regression coefficient (R^2), Model efficiency (ME) and Percentage Bias (P_{bias}) calculated a validation period from March 2009-1 June 2010) using baseline data representing surface waters (NOx, PO_4, Chl-a) and bottom water dissolved oxygen, DO) and secchi depth (entire water column). Values in brackets represent index values where data from the spring bloom were omitted.

Validation	R^2		ME		P_{bias}		n	
NO_x	0.72	(0.86)	0.70	(0.86)	8	(4)	45	(33)
PO_4	0.61	(0.77)	0.62	(0.75)	15	(10)	45	(33)
Secchi depth	0.41	(0.61)	0.64	(0.90)	1.7	(1.1)	45	(33)
Chl-a	0.52	(0.69)	0.50	(0.62)	9	(6)	44	(33)
DO	0.56		0.38		18		32	-
DO (mooring)	0.44		0.33		34		468	-

Except for simulating the timing of spring bloom (see Appendix A) the FEMA model scores "very good" to "excellent" based on the performance of the three indices, see Table 3.17. Even if water quality and plankton data from the spring bloom period are included in model assessment, the scores of model performance are "good" or "very good" according to the ME scale (Marechal 2004).

It should be underlined that the FEMA model was calibrated on 2005 data collected outside the assessment area because high-resolution data from the Fehmarnbelt in 2005 was not available (see Appendix A). Therefore, model validation using another year and for another area compared to calibration is a challenge to the FEMA model. As shown in Table 3.17, the outcome of the validation was very satisfactory.

3.10 Assessment of severity and significance

Using the calculated degree of impairment the severity is assessed using expert judgement, and taking account of experiences from comparable impact assessments. When applicable, areas of specific and general importance for water quality and plankton in the assessment area are taking into account.

Significance of an impact is assessed quantitatively and qualitatively taking into account the relative impairment compared to the whole area the duration and severity of the impairment.

4 MAGNITUDE OF PRESSURE

Six types of pressures are relevant for water quality and plankton biology. They include 4 temporary pressures related to construction works:

- Increased concentration of suspended sediments (spill from dredging operations) influencing light penetration into the water column (Secchi depth), that in turn affects primary production, phytoplankton biomass and composition and, zooplankton production. Besides, high concentration of suspended sediments can lead to increased (facilitated) sedimentation of phytoplankton by flocculation and high turbidity along beaches caused by high concentrations of suspended sediments constitutes an aesthetical nuisance for bathers.

- Increased sedimentation of suspended sediments can bury resting eggs of copepods and potentially affect recruitment of copepods affecting the composition of zooplankton community.

- Release of toxic substances during dredging operations potentially harming plankton organisms.

- Increase of oxygen demand caused by reduced substances released during dredging

Two relevant pressures are of permanent nature (operational phase). They include:

- Changes in hydrography, especially changes in vertical mixing across pycnocline affecting strength and duration of water column stratification. Changes in vertical mixing can affect risks for blooms of cyanobacteria and change the oxygen ventilation of near-bed water.

- Bridge pylons and pillars will increase the area of hard substrate thus constituting additional substrate for blue mussels, macroalgae and ephyra of jellyfish. Mussels will remove part of phytoplankton passing between pillars, macroalgae growing below pycnocline can add to oxygen content in water passing pillars and higher abundance of ephyras may increase recruitment of jellyfish that is regarded as a nuisance in the western Baltic Sea.

4.1 Suspended sediment

4.1.1 Concentration of suspended sediment for tunnel solution

In total, 55.8 mill m^3 of soil, sediment and sand will be handled during construction work, including 20 mill m^3 for the land reclamation work at Lolland. Depending on actual work the estimated spill will vary between 0.1% and 3.5% of handled material (FEHY 2013a). Spill volume is largest from October 2014 through May 2016 when tunnel trench is dredged and sand mining outside Rødby takes place. For the remaining construction period spill intensity (volume per month) is reduced to about one tenth of the intensity at start.

Concentration and spatial distribution of suspended sediments originating from dredging spill varies strongly with the character of spilled sediments (spill rate and particle size distribution in spill) and especially with speed and direction of current abound dredgers. Maximum excess concentration in surface water is found along the Lolland coast, in the western part of Rødsand Lagoon (exceeding 100 mg/l at times) and in the alignment corridor where the major dredging activity takes place (Figure 4.1).

Duration of higher excess surface concentrations is limited and concentrations exceeding e.g. 2 mg/l in surface waters occur in less than 10% of the time except along Lolland, in Rødsand Lagoon and along Hyllekrog barrier in the first year of construction works for the tunnel alternative (Figure 4.2). Locally, in these areas 2 mg/l may be exceeded up to 50% of the time in the first year of dredging. It shall be underlined that concentrations of spilled sediment will increase with depth in the water column which will affect exceedence periods.

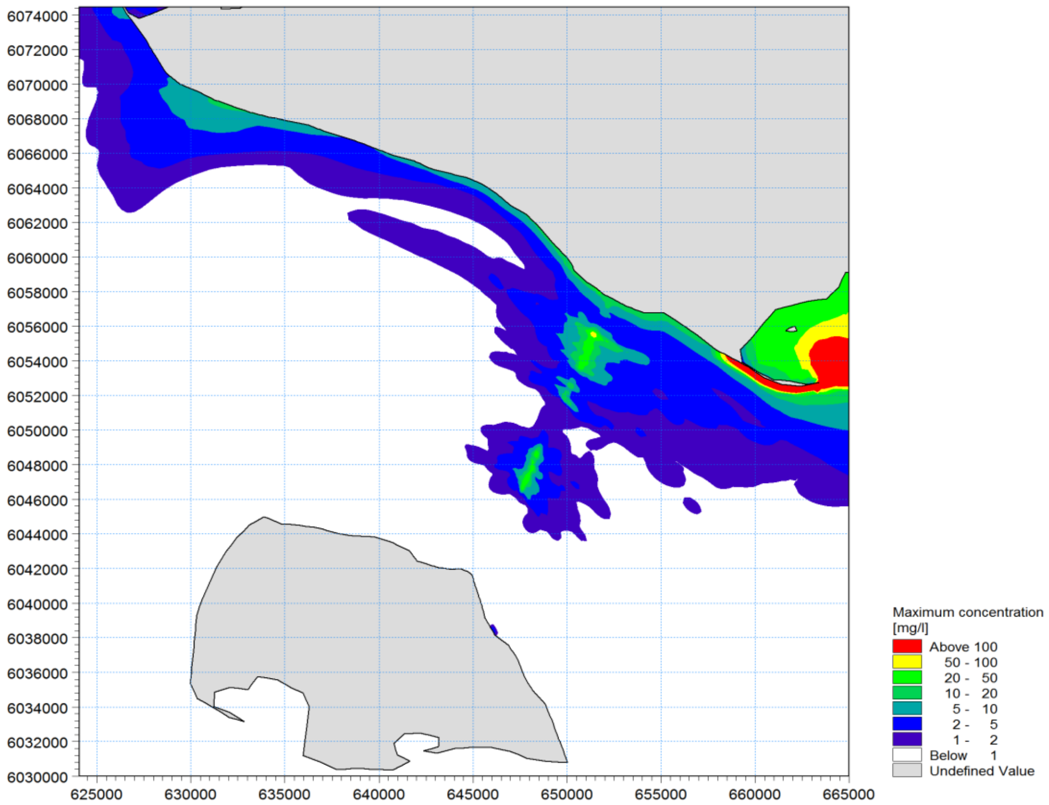


Figure 4.1 Maximum concentration of suspended matter in surface water for the tunnel alternative during the period 1.5 2015 – 1.9 2015. From FEHY (2013a).

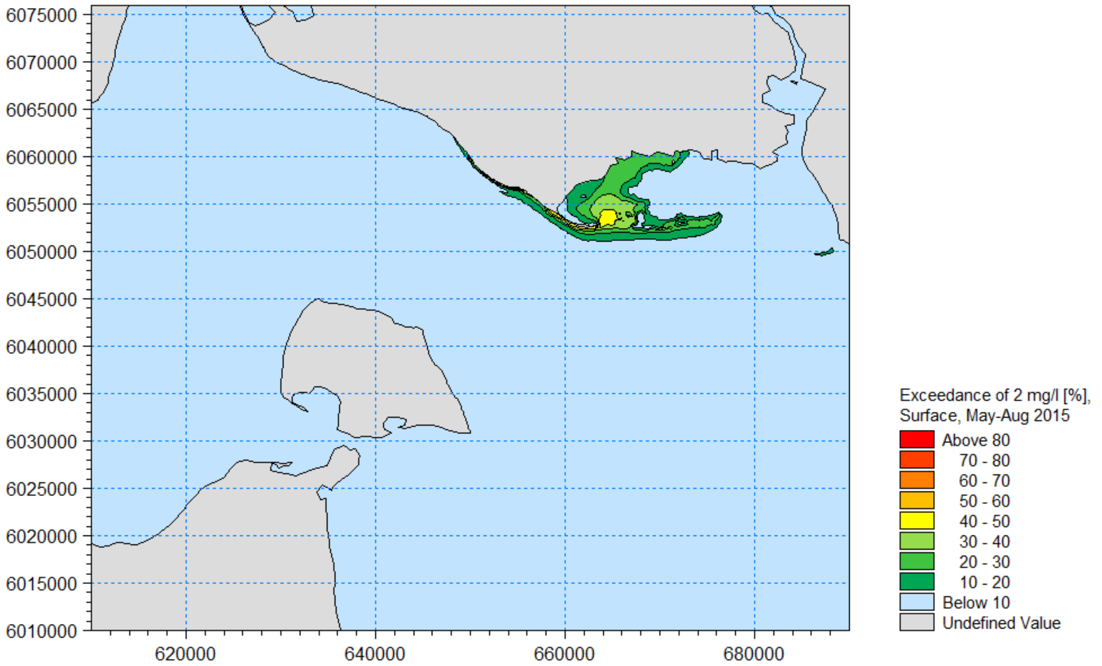


Figure 4.2 Exceedance time of 2mg/l, 1/5-1/9 2015 for the surface (top layer in the numerical model results) for the tunnel alternative. From FEHY (2013a).

4.1.2 Concentration of suspended sediment for bridge solution

The amount spilled sediment in connection with soil works for the bridge alternative is roughly one tenth of the spill expected for the tunnel alternative and, both maximum, average and excess concentrations will be much lower (≈ 10%) compared to the tunnel solution, e.g. Figure 4.3.

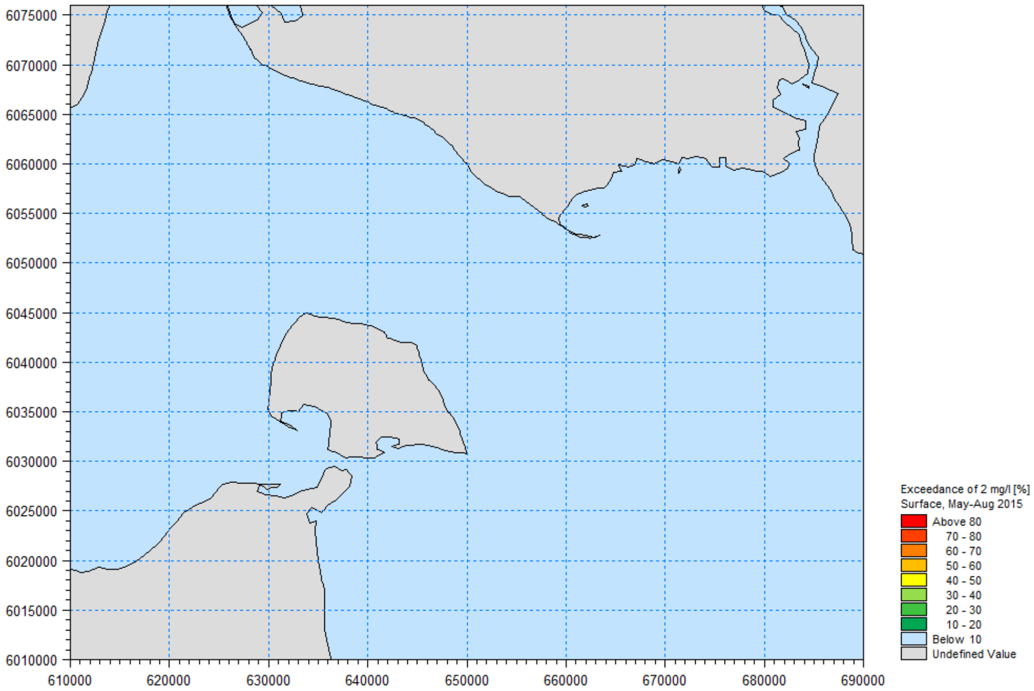


Figure 4.3 *Exceedance time of 2mg/l, 1/5-1/9 2015 for the surface (top layer in the numerical model results) for the bridge alternative. From FEHY (2013a).*

4.2 Sedimentation

4.2.1 Deposition of spilled sediments for tunnel solution

At the end of construction period (2019) coarse sediments (sand) will have accumulate along the tunnel trench in heights between <5mm to 20mm (Figure 4.4), and at comparable heights (10-50mm) in the western part of the Rødsand Lagoon (Figure 4.5) Outside the Fehmarnbelt the final deposition of fine sediments is predicted in deeper basins primarily in the Arkona Basin east of the Darss Sill and east of Als in southern Little Belt (Figure 4.6).

Figure 4.4 Final accumulation of sand along the alignment after termination of dredging works for the tunnel solution map from (FEHY 2013a).

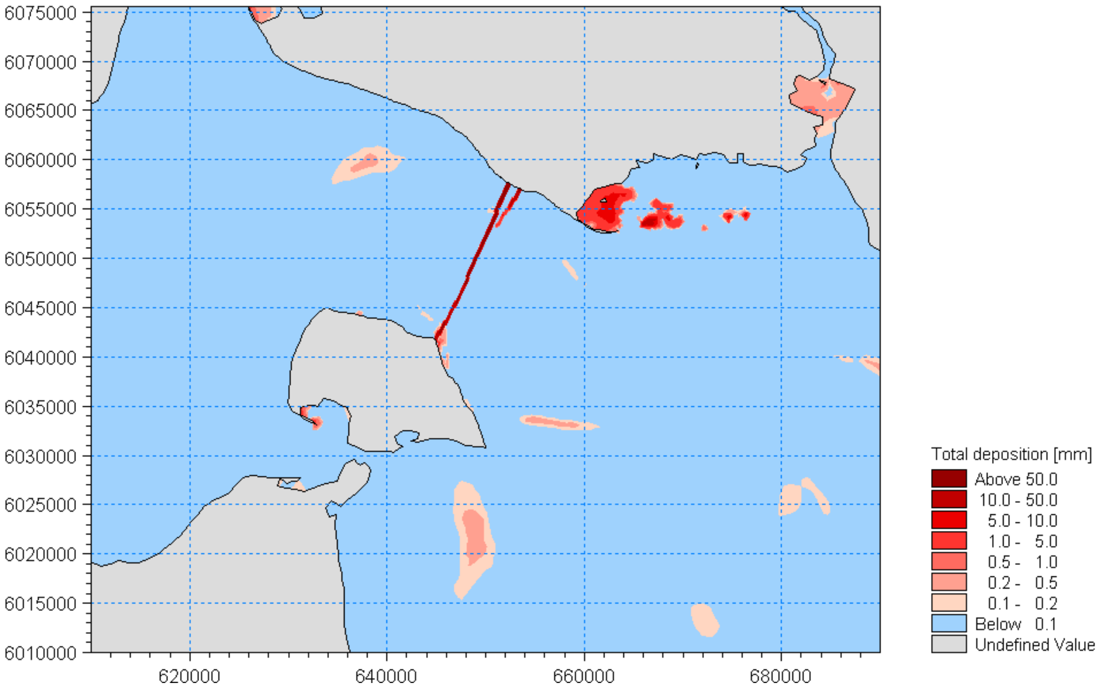


Figure 4.5 Final accumulation of spilled sediments in the Fehmarnbelt after termination of dredging works for the tunnel solution map from (FEHY 2013a). Zoom-in on Fehmarnbelt area.

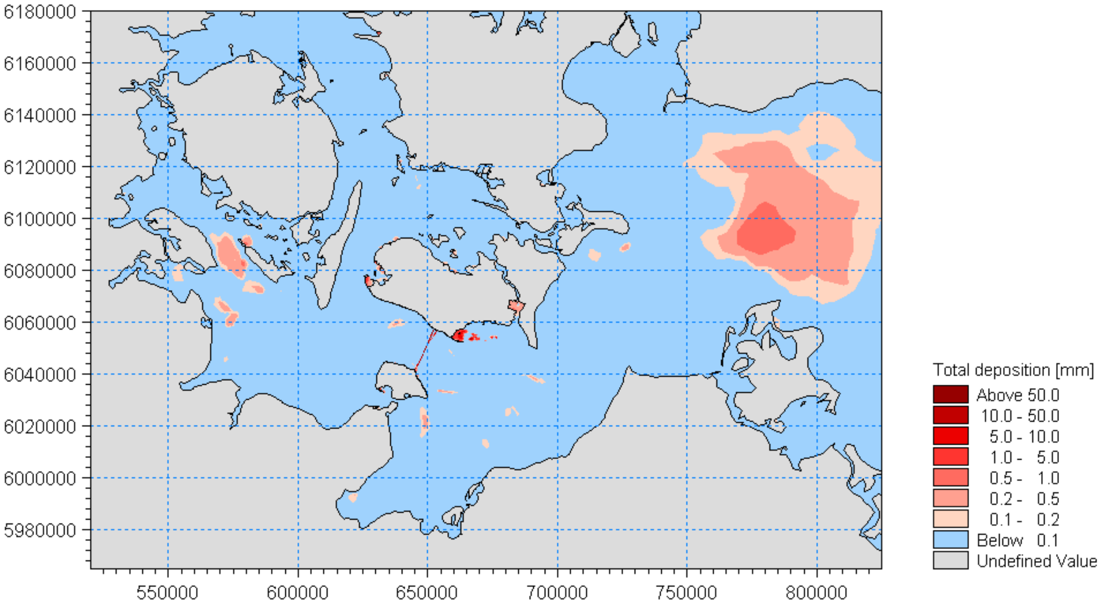


Figure 4.6 Final accumulation of spilled sediments in the Fehmarnbelt after termination of dredging works for the tunnel solution map from (FEHY 2013a). Entire local modelled area.

4.2.2 Deposition of spilled sediments for bridge solution

The final accumulation of sediment spill for the bridge solution will take place in the same areas as for the tunnel solution, but because the spill amount is roughly one tenth of spill from the tunnel solution the deposition heights will be about ten times lower for the bridge (FEHY 2013a).

4.3 Toxic substances

Upon dredging and disposal of dredged sediment in Fehmarnbelt there is a risk of release of toxic substances to the water column which potentially can affect planktonic organisms.

Depending on intensity of dredging (production rates), spill rates, concentrations of toxic substances in sediments, water depth and current velocity the concentrations of toxic substances in the water column will be highly variable. Also, a spatial and seasonal varying character of sediments (anaerobic/aerobic), concentration of calcium in sediment and water etc. are important for how much of toxic substances that can be released from sediment (Eggleton and Thomas 2004). Therefore, fixed and robust predictions of toxic releases cannot be made. Instead, the risk for exceeding Environmental Quality Standards (EQS) set by EU and national authorities to protect the aquatic environment against toxic impacts can be estimated combining distribution-functions of:

- Concentrations of toxic substances in Fehmarnbelt sediments ("source" strength)
- Release rates (% of sediment content) obtained in laboratory experiments and field studies
- Spill rates expected during Fehmarnbelt dredging
- Current velocity in Fehmarnbelt (affecting initial dilution)

Along with the daily dredging intensity the distribution functions for concentration, release rate, spill rate and current velocity were combined in a Monte-Carlo analysis and the calculated concentrations of toxic substances in vicinity of dredger were compared to EU Environmental Quality Standards, EQS (EU 2008) and provisional Danish standards (Naturstyrelsen 2010), see Chapter 7.

4.3.1 Toxic substances in Fehmarnbelt sediments

Concentration of toxic substances (heavy metals, HCB, PAH, PCB, DDT, TBT) was measured under the baseline study in samples taken across Fehmarnbelt along the alignment (FEHY/FEMA sediment baseline), and heavy metals was analysed in sediment samples analysed under the Feasibility Study (COWI-Lahmeyer 1998).

In total, 15 samples representing of heavy metal concentrations from the upper 0-1m of the sediment column are available and 7 samples representing metal concentrations from the sediment column below 1m from surface are available. Six samples representing subsoil background concentration of heavy metals came from the Great Belt (DHI/LIC Joint Venture and COWI/VKI Joint Venture 1990). Summary statistics of concentrations for heavy metals and POP's are presented in Table 4.1 and Table 4.3. Overall, except for Hg (uniform distribution) the concentrations in sediments appeared to be log-normally distributed.

Table 4.1 Concentration of toxic substances in surface and subsurface sediments from Fehmarnbelt alignment. Average concentrations, standard deviations (SD) and number of stations/samples analysed shown. Subsurface data include samples from the Fehmarnbelt feasibility study (COWI-Lahmeyer 1998) and the Great Belt Link study (DHI/LIC Joint Venture and COWI/VKI Joint Venture, 1990).

Surface sediment	**Average**	**SD**	**Range**	**n**
	---------- *mg/kg DW* ----------			
Cd	0.11	0.10	<0.05-0.34	15
Cr (total)	16.4	14.6	1.6-45	15
Cu	9.9	7.2	<1-21	15
Hg	0.013	0.009	<0.01-0.03	15
Ni	12.0	10.2	1-31	15
Pb	13.2	8.6	2.8-28	15
Zn	31.1	19.2	5.5-61	15
Sub surface sediment		mg/kg		
Cd	0.12	0.04	0.06-0.26	11
Cr (total)	12	11	9.0-14	11
Cu	9.1	9.2	4-17	11
Hg	0.03	0.01	<0.01-0.05	8
Ni	9.0	7.6	8.0-15	11
Pb	5.2	4.5	2.3-14	11
Zn	22.0	4.5	14-33	11

4.3.2 Release of heavy metals from Fehmarnbelt sediments

Potential for release of heavy metals in connection to dredging has been quantified in a number of studies using elutriation experiments ("shake bottle") with natural sediments and with varying conditions such as anaerobic – aerobic conditions, addition of chelating agents (e.g. EDTA), acidification etc. (Lee et al. 1975, Petersen et al. 1997, Huang 2003, Khalid et al. 1977, Calmano et al. 1994, Shipley et al. 2006). Another set of data comes from commercial or experimental dredging operations where concentration of dissolved heavy metals or organic pollutants was monitorind during dredging (e.g. Van der Berg et al. 2001).

In the Fehmarnbelt Feasibility study and in the Øresund Link study mobility of sediment-bound metals was studied in elutriation tests. Briefly, 15 g wet sediment in 200 ml filtered seawater for 24 h on a shaking table. After settling of larger particles supernatant was filtered and analysed for metals. Release was calculated as a percentage of the metal amount in the 15 g sediment added. The rationale for such a test is that an equilibrium between metals associated to suspended matter and dissolved in water is expected to be established mimicking the sorption/desorption processes when sediment is exposed to natural sea water under dredging.

The %-release of metals generally agree between different studies (Table 4.2), with Cd, Ni and Zn consistently showing the largest release, while the relative release is very low for Cr, Pb and partly for Cu.

Table 4.2 Release of metals from natural sediments mimicking a dredging operation. Values represent %-release of total amount in shaking bottle experiments. For the Fehmarnbelt feasibility and the Øresund Link study values are shown as average and SD in brackets). For comparison, average values from 2 studies carried out using comparable methods are shown

	Fehmarn* feasibility	Øresund	Calmano et al. (1994)	Shipley et al. (2011)	
	Average (±SD)	**Average (±SD)**		**Anaerobic**	**Aerobic** (pH=4.8)**
Cd	10.0 (±7.1)	1.20 (±0.6)	5.0	0.9	5
Cr	0.07 (±0.09)	0.10 (±0.1)			
Cu	0.11 (±0.09)	0.14 (0.09)	1.0	0.06	0.07
Ni	1.84 (±2.36)	4.80 (±4.1)		0.3	2.80
Pb	0.10 (±0.05)	0.10 (±0.1)	0.7	0.03	0.01
Zn	3.50 (±1.94)	3.90 (±2.4)	1.5	0.35	9.00

**data from experiments with sand not included due to very low metal concentration in sediment and water (low reliability of analytical results)*
***Aerobic data used in distribution functions*

4.3.3 Predicted concentration of heavy metals in the spill plume during dredging

The range in metal concentrations resulting from sediment release in connection with dredging is likely to be quite wide due to variability in the chemical and physical characteristics of sediments, the dredging losses (% sediment spill), dredging production rates (volume/time), water depths, and flow velocity of water at site of dredging. One way to represent the high variability is to combine probability distributions for key factors in a Monte-Carlo analysis.

An example of distribution functions for sediment spill (0.3 – 5%, uniform probability), flow velocity, sediment concentration and %-release of nickel (Ni) from sediments is shown in Figure 4.7. Ni was chosen because of high release rates (Table 4.2) and low EQS values (see section 3.7). Predicted concentrations of heavy metals in the spill plume are estimated by "multiplying" the distribution functions, see below. Calculations were also carried out for Cd and Zn due to their high release rates (Table 4.2).

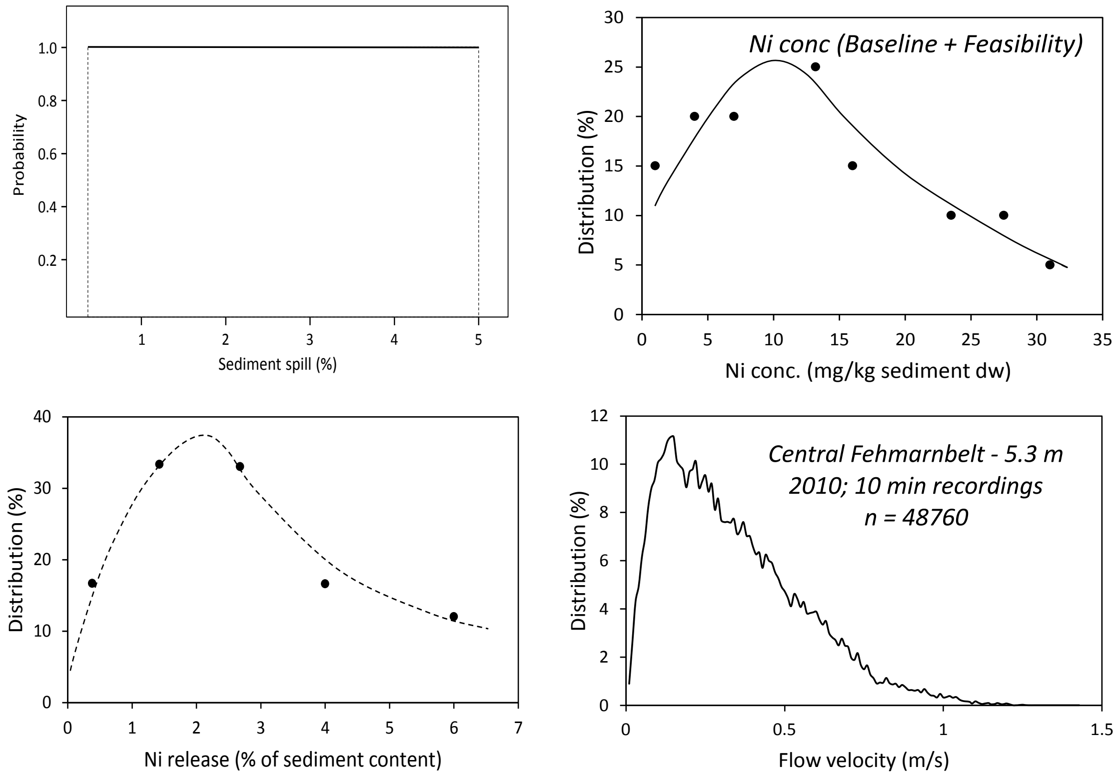


Figure 4.7 *Distribution and probability (functions) for sediment spill, concentration of Ni in Fehmarnbelt sediments, release of Ni from sediments and flow velocity in surface water under simulated dredging conditions*

Assumptions

Assumptions for calculating concentration of heavy metals and persistent organic pollutants in spill plume are listed in Table 4.3.

Table 4.3 *Overview of assumptions used in Monte-Carlo analysis for prediction concentrations of heavy metals and POP's in the spill plume from a dredger during dredging in Fehmarnbelt*

Activity/process	**Value (range)**	**Comments**
Dredging intensity	5000 m^3/dredger/d	Fixed (FEHY 2013a)
Sediment bulk density	1800 kg/m^3	Fixed (FEHY 2013a)
Plume width	25 m	Fixed (very conservative)
Depth	15 m	Fixed (to primary pycnocline)
Sediment spill	0.3-5%	Variable: uniform probability
Tox conc in sediments	Measurement range	Variable: log-normal distribution
Tox release from sediments	Measurement range	Variable: log-normal distribution
Flow velocity	Measurement range	Variable: log-normal distribution

The predicted increase in concentration of a toxic substance (ΔTox) during dredging operations is calculated as:

$$\Delta\text{Tox} = \text{Dredging intensity x Bulk density x Spill (\%) x Tox conc x Tox release / (Plume width x Depth x Flow velocity).}$$

Monte-Carlo analyses were carried out using the assumptions in Table 4.3 and distribution functions for flow velocity, concentration and release rates of metals (Table 4.1, Figure 4.7). The analyses were performed for the metals Cd, Ni and Zn which showed the highest release rates and also for Benz(a)pyrene. Calculations were carried out using the RiskAMP software and using 1000 simulations as a standard.

Predicted metal concentrations
The calculations predicted highly varying concentrations of heavy metals in plume water (Table 4.4). Using the 1-percentiles and 99-percentiles to represent the range, the predicted concentrations varied a factor of 383 for Cd, 360 for Ni and 140 for Zn. The range in predicted concentrations is primarily due to variations in sediment concentrations and release rates, which were highest for Cd and Ni and lowest for Zn (Table 4.1 and Table 4.2).

The predicted median concentrations will represent central estimates while the 99-percentiles can be seen as estimates of maximum concentrations in the spill plume. These toxic pressures can be compared to EU and national Environmental Quality Standards for heavy metals).

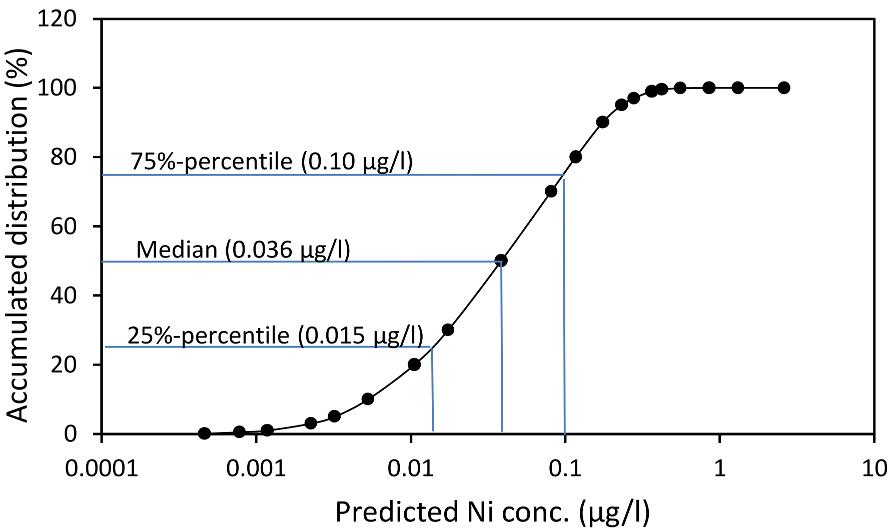


Figure 4.8 Distribution for predicted concentration of Ni in sediment plume down-stream dredger.

Table 4.4 Predicted percentiles of increases in concentrations of Cd, Ni and Zn (µg/l) for dredging activities in the Fehmarnbelt. 50%-percentile (median) will be central estimates of concentration in the spill plume, and 99-percentiles will represent maximum concentrations during dredging.

Percentile	Cd	Ni	Zn
1	0.00006	0.001	0.010
25	0.0009	0.015	0.058
50	0.0033	0.036	0.16
75	0.007	0.10	0.33
95	0.0132	0.23	0.91
99	0.023	0.36	1.4

4.3.4 *Predicted concentration of Persistent Organic Pollutants in the spill plume during dredging*

Predicted concentrations of Persistent Organic Pollutants (POP's) in the spill plume during dredging were measured under the Baseline Study at 13 stations (Table 4.5). Synthetic substances (PCB, DDT and TBT) could only be detected in the upper 0.2 m of the sediment column, while PAH compounds were detectable at least to 1.0m below sediment surface. However, concentrations fell to between 5% and 10% of surface values at 9-12 cm depth below the sediment surface (FEHY 2013d).

Table 4.5 Concentration of persistent organic pollutants in surface sediments from Fehmarnbelt alignment. Average concentrations, standard deviations (SD), range and number of stations/samples (n) analysed shown.

Surface sediment	**Average**	**SD**	**Range**	**n**
	------------ µg/kg DW ------------			
HBC	50.7	20.4	26-96	13
PCB (sum 7)	5.7	18.9	0.15-74	13
DDT (p,p'-DDE,p,p'-DDD, p,p'-DDT)	0.28	0.35	<0.03-1.13	13
PAH (sum 16)	158	191	50-640	13
Benz(a)pyrene	9.3	14.7	0.08-44	13
TBT	1.1	0.8	<0.3-2.2	5
DBT	0.6	0.3	<0.4-1.2	5
MBT	0.4	0.2	<0.3-<0.6	5

Considering that the concentrations of POP's in surface sediments are much below current sediment quality guidelines (i.e. non-toxic for sediment-living organisms, FEHY 2013d) and that non-enriched sediments with zero concentration (synthetic POP's) or background concentrations (PAH) by far dominate the sediment volume dredged, impacts in the water column due to release is not likely. A maximum pressure estimate for benzo(a)pyrene is given below.

Benzo(a)pyrene

Benz(a)pyrene (B(a)P is among the most toxic (and carcinogenic) PAH compounds and in accordance a low Environmental Quality Standard has been set to protect aquatic organisms (Table 3.9; EU 2008). The maximum concentration found in surface sediments in the Fehmarnbelt during baseline was 44 µg B(a)P/kg. Below 10-12 cm depth sediment concentration fell to 1-2 µg/kg (FEHY 2013d).

Along with other POP's there is only limited information on B(a)P release from sediment, but in the few studies release rates are very low, between 0.002 and 0.1% of sediment content (Geffard et al. 2003).

In a 'worst-case' scenario it is assumed that

- Benz(a)pyrene concentration is 44 µg/kg in the upper 20 cm of the sediment column and 4 µg/kg from 20 cm to 12 m (lowest dredging depth)
- release rate is 10% (i.e. 100 times larger than the maximum rate published)
- sediment spill of 5%
- current speed at 0.01 m/s (occurring in less than 0.1% of the time)
- sediment density, water depth and plume width as in Table 4.3.

The worst-case estimate of Benz(a)pyrene is 0.0003 µg/l, while a more likely estimate is 50-100 times lower. These values will represent the pressure from B(a)P and they can be compared to the EQS for B(a)P (see Chapter 3 and 7).

4.4 Oxygen demand

During dredging, part of accumulated oxygen demand in sediments will be released momentarily to the water column where reduced substances such as H_2S will be oxidized consuming part of the dissolved oxygen in water. Eventually, concentration of oxygen may decrease to below critical levels and the criteria set to protect marine organisms (see section 3.9.3). Analogously to toxic substances the probability to violate criteria depends on source strength (dredging intensity, spill rate, content of oxygen demand in spilled sediments) and, oxygen concentration and current speed in water column.

4.4.1 Oxygen demand in Fehmarnbelt sediments

Oxygen demand in sediments from the Fehmarnbelt was examined in laboratory experiments as part of the baseline study (FEHY 2013d). Using distribution statistics for organic content, weight-specific oxygen demand, and density of sediments the median oxygen demand (50-percentile) was calculated at 23.25 kg O_2/day at a daily dredging intensity of 5000 m^3/dredger and, 25- and 75-percentiles at 17 and 45.25 kg O_2/day.

4.4.2 Oxygen flux in bottom water

Below the photic zone, i.e. at depths greater than 15 m oxygen consumed is continuously replenished fully or partly by vertical mixing and horizontal transports.

Except in dense eelgrass beds, concentration of oxygen above pycnocline can be assumed to be fully saturated, while below the pycnocline oxygen exchange with atmosphere is hindered by the pycnocline and, the concentration reaches a minimum in late summer (see Figure 3.6).

Modelled current speed (daily averaged) in bottom water along Fixed Link alignment varies between 0.005 m/s to 0.24 m/s. Median current speeds generally increases with water depth to a plateau of 0.075 m/s, but with a local minimum within the depth range 12-18m (Table 4.6).

Table 4.6 Modelled current speed in near-bed layer (lowest 2 model layers) along the depth profile in the alignment. Data extracted from FEHY MIKE 3 model for the model year 2005.

	3-6m	**6-9m**	**9-12m**	**12-15m**	**15-18m**	**18-21m**	**21-24m**	**>24m**
				m/s (daily average)				
25%	0.024	0.026	0.040	0.033	0.028	0.041	0.048	0.049
50%	0.045	0.061	0.078	0.071	0.063	0.077	0.071	0.078
75%	0.106	0.133	0.142	0.132	0.111	0.128	0.125	0.16

The daily horizontal flux of oxygen in the near-bed layer can be estimated by multiplying the current speeds (Table 4.6) with the corresponding concentrations of dissolved oxygen.

4.4.3 Assessment of dredging on oxygen levels

Worst-case scenarios of dredging impacts can be quantified by combining a high release of oxygen demand (75%-percentile = 45.25 kg O_2/day) with low current speeds (25%-percentiles: 0.024 – 0.049 m/s), and during periods of low oxygen in bottom water (≈ 2.6 mg O_2/l). Four examples (3 worst-case and a central estimate) are shown in Table 4.7.

The most critical environment for release of oxygen demand is benthic habitats located just below the pycnocline at 15-18m; benthic organisms are sensitive to hypoxia, current speeds are comparable low and, the height of bottom water (from seabed to pycnocline) is low implying that the maximal oxygen content is limited and additional oxygen demand may exhaust the entire oxygen content.

Table 4.7 Estimated reductions in oxygen concentration in bottom water caused by release of oxygen demand during dredging. Three examples from a hypothetical 18 m station shown; worst-case 1: high oxygen demand, low current speed, 3% spill distributed evenly over the water column; worst-case 2: as worst-case 1 but all spill occurs below pycnocline; median; worst-case for station located above pycnocline.

	Below pycnocline (18 m)			**Above pycnocline**
	worst-case 1.	**worst-case 2.**	**median**	**worst case**
Current speed (m/s)	0.028	0.028	0.063	0.025
Diss. oxygen (g/m^3)	2.6	2.6	7.5	7.5
Oxygen demand (kg/d)	45.25	45.25	23.25	45.25
Plume width (m)	25	25	25	25
Oxygen demand (kg/m/d)	2.51	15.08	1.29	7.54
Depth range (spill)	18	3	18	6
Daily oxygen flux (kg/m)	157	157	1021	405
Oxygen reduction	1.6%	9.6%	0.1%	1.9%

Worst-case scenario impacts on bottom water oxygen due to release of oxygen demand showed reductions between 1.6% and 9.6% from an already low concentration of 2.6 mg O_2/l. The highest reduction (worst-case 2) will occur if all oxygen demand (and sediment spill) takes place in a shallow (3 m) water column below the pycnocline. Such situations are highly unlikely.

A more likely situation is represented by the "median" case in Table 4.7; concentration of oxygen below pycnocline is close to saturation (as during October through May), current speeds and oxygen demand are at median levels, and sediment spill and oxygen demand are distributed all over the water column. In such situations a reduction in bottom water oxygen of ca. 0.1% can be expected.

4.5 Loss of pelagic habitats

Both tunnel and bridge solution will imply a loss of pelagic habitats. Loss of volume will affect plankton but not the water quality (see 2.2.6).

4.5.1 Loss of pelagic habitats - Tunnel solution

Implementation of the tunnel solution will lead to a loss of pelagic habitats due to land reclamation and protection reefs covering 355 ha of seabed in shallow areas (FEMA 2013a). The loss of pelagic volume can be estimated to ca. 9,900,000 m^3. More than 90% of this volume loss is confined to waters of general importance for plankton (see 1.2.1).

4.5.2 Loss of pelagic habitats - Bridge solution

Implementation of the bridge solution will lead to a loss of pelagic habitats due to land reclamation (36 ha), piers and pylons (20 ha). The volume lost to land reclamation (ca. 1,000,000 m^3) is confined to waters of general importance for plankton (see 1.2.1), while volume loss due to piers and pylons (455,000 m^3) almost exclusively is located in waters of special importance for plankton.

4.6 Solid substrate

Additional solid substrate will be available both for the tunnel and bridge solutions.

4.6.1 Solid substrate - Tunnel solution

The introduced solid substrate has been calculated to 149 ha distributed between embankment at Lolland (10 ha), protections reefs (12 ha) and a protection layer covering the tunnel elements (FEMA 2013a). Also, as a consequence of the tunnel solution about 355 ha of seabed will permanently be lost, covered by reclamation areas and protection reefs. About 50% of this loss consists of habitat for mussels (FEMA 2013a). In terms of suitable and partly new solid substrate for epifauna the protection reefs in particular are relevant.

In contrast, the protection layer covering the tunnel elements will be of semi-permanent nature, because the trench gradually will be covered with mud, and it is not clear to what extent the substrate used to cover the embankments will be suitable for e.g. mussels.

In total, the tunnel solution will imply a loss of "semi-solid" substrate of ca. 175 ha (mussel habitat) and a gain of 12 ha of true solid substrate and, an additional gain of 10 ha embankments of questionable habitat value for epifauna.

4.6.2 Solid substrate - Bridge solution

The structure-related loss of seabed area amounts to 56 ha distributed between reclamation areas, piers and pillars. Thirteen ha of the lost area is located in the *Mytilus* community along Lolland (FEMA 2013b).

The bridge solution comprises 81 pylons and piers and 2 artificial peninsulas on either end of the bridge with a total surface area of 24 ha integrated to a depth of 20m (FEMA 2013a). The new solid substrate potentially will be colonized by macroalgae, epifauna, primarily filter-feeders such as blue mussels, and epiphyra of jellyfish. The major part of the hard substrate (ca. 90%) is located in the area of special importance for plankton (depth > 6 m), see Figure 2.1.

4.6.3 Oxygen production by macroalgae

Based on extensive investigations of fouling organisms on foundations in the nearby Nysted windfarm, on pillars and pylons of the Øresund, Great Belt and Öland bridges FEMA (2013a) concluded that colonization, growth and area cover of macroalgae will be very limited in the Fehmarnbelt bridge solution.

Additional oxygen production below pycnocline will therefore be minor and insignificant for oxygen concentrations in the Fehmarnbelt.

4.6.4 Reduction of phytoplankton by mussels on submerged bridge structures

The additional filtration capacity due to expected population of mussels on the piers and pylons was calculated from abundance and length distribution (Figure 4.9) determined at Great Belt bridge (COWI/DHI 2008) pillars:

$$\text{Filt } (m^3/d) = \text{Abu}*0.185*(\text{L;cm})^2*24/1000,$$

where Filt is the filtration capacity at 10-20 ^{o}C, Abu is the mussel abundance on hard substrate (ind/m^2), L is the shell length in cm, and 0.185 is a scaling factor (Kiørboe and Møhlenberg 1981). The total filtration capacity in a depth interval is calculated by summing filtration capacity for all mussel size classes and multiplying with the area of hard substrate in the appropriate depth interval. Referring to the study from the Great Belt Bridge (COWI/DHI 2008) mussel abundance and size distribution are assumed to be uniform from 2 m below sea surface to 14m depth. Outside this interval mussels are not expected to permanently populate the solid substrate because of wave and ice stress near sea surface and low food concentration below the seasonal pycnocline.

The theoretical filtration capacity is calculated to 315 $m^3\ m^{-2}\ d^{-1}$. Because mussels is expected to occur in an approx. 5 cm dense mat the realized filtration is lowered by 30% to account for "refiltration" within the mussel mat (Møhlenberg and Petersen 1998).

Neglecting the artificial peninsulas (because habitat suitability is not known) the area of hard substrate available to mussel settlement is ca. 140 $m^2\ m^{-1}$ water column between pylons. The average daily directional advection of water between pylons is calculated as: average daily current speed (from mooring in central Fehmarnbelt: MS01) * distance between pylons = 0.27(m/s) * 3600 * 24 * 220 (distance between pylons).

The relative impact of mussels on chlorophyll-a concentration in surface waters passing between bridge pylons are estimated by:

$$\frac{filtration}{advection} = \frac{315 * 0.7 * 140}{0.27 * 3600 * 24 * 220} = 0.006$$

With a potential capacity of filtering 0.6% of the water passing the bridge the impact of mussels populating solid bridge structures is found to be insignificant for chlorophyll-a concentration in the Fehmarnbelt.

An alternative calculation based on the sum of annual baroclinic and barotrophic transports in surface waters 0-15 m (ca. 800 km^3/y; Bendtsen et al. 2009) and a total solid substrate of bridge piers and pylons (0-15 m) of 147420 m^2 gives a potential reduction of 1.5% of water passing under the bridge.

This supports that the impact of mussels on bridge structures on plankton communities will be negligible.

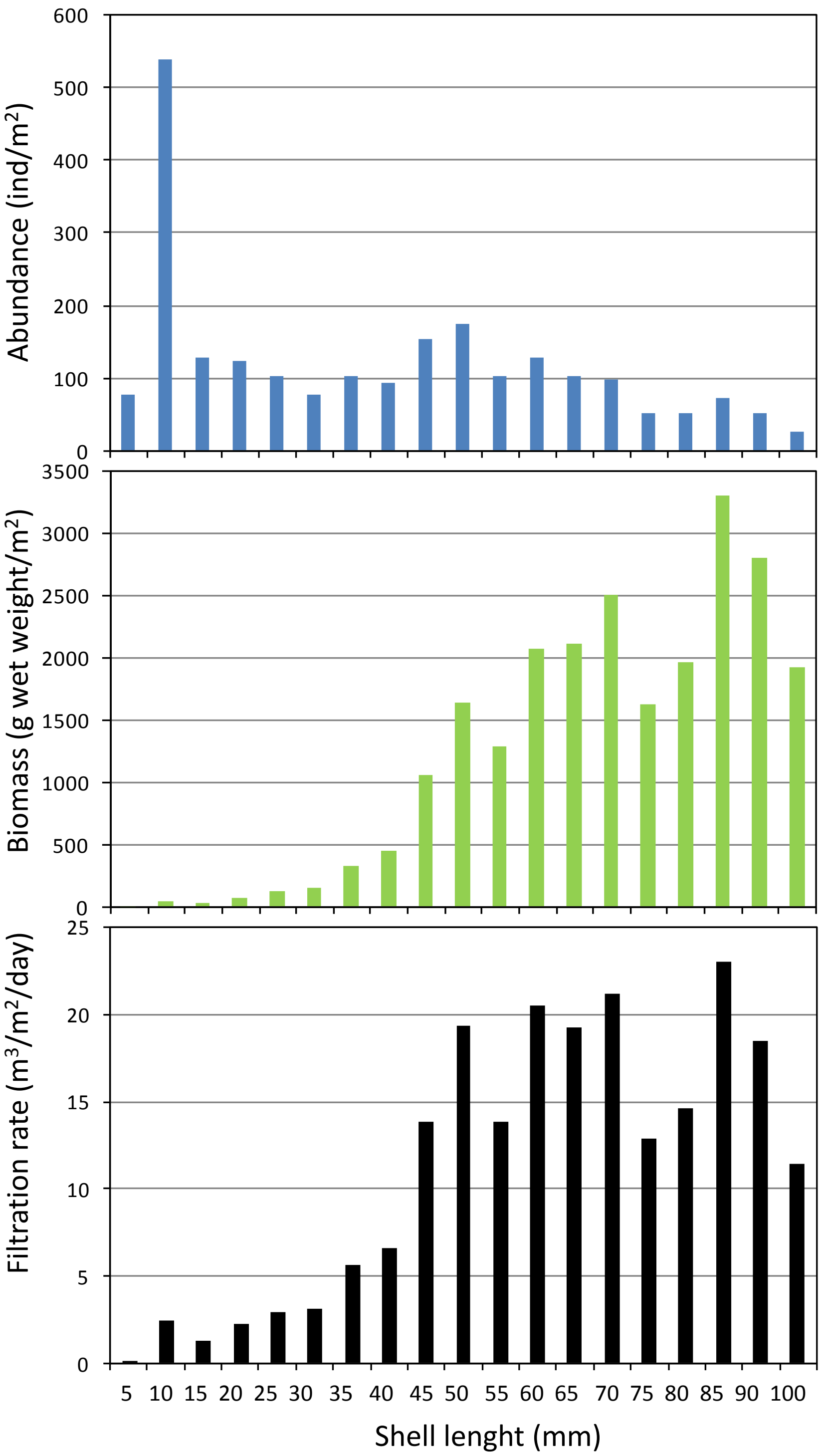


Figure 4.9 Abundance, wet weight and theoretical filtration rate of different sized Mytilus edulis on Great Belt piers. Calculated from (COWI/DHI 2008).

4.6.5 *Decrease in oxygen around pylon foundations caused by sedimenting mussel feces.*

Mussels feeding in environments with low biomass of phytoplankton will egest about 20% of the ingested phytoplankton carbon (C) as feces (Kiørboe et al. 1980). Assuming that mussels will populate pylons evenly from 2 m below MWL to summer pycnocline at 13 m depth and that the average phytoplankton concentration above pycnocline is 65 mgC m^{-3} (estimated from 1.85 µg chlorophyll-a and a C:chla ratio of 35; FEMA-FEHY 2013) the feces production per pylon (at depths ≥13 m) can be estimated by:

*Feces production (per pylon) = 315*0.7*140*0.2*65*11 = 4414 gC/d*

The layer of mussels on pylons will attract invertebrates especially amphipods and omnivorus polychates such as *Hediste* that feed on feces retained in the 3-dimensional matrix created by the cover of mussels. There is no quantitative information in the literature but it is likely that about 50% of produced feces is sequestered in the mussel matrix leaving 50% (ca. 2200 gC/d) to settle out of the water column.

The rate of feces input to the sediment and the area extension where feces will settle and ultimate deposit depends on a large suite of factors, including

- settling velocity of feces
- critical shear stress for feces to resuspend
- direction and magnitude of current speed above and below pycnocline
- horisontal dispersion

Settling velocity of mussel feces has been reported to vary between 0.003 and 0.027 m/s (Chamberlain et al. 2001; Chamberlain 2002; Callier et al. 2006) depending of size of mussels and type of ingested material (Giles and Pilditch 2004; Callier et al. 2006), while critical shear stress (*u*)* to erode feces at 0.4 cm/s (Giles and Pilditch 2004) is substantial higher than for very fine particles including phytoplankton cells. Except near foundations of pylons where currents are high settled feces are not likely to resuspend.

Assuming a daily average current speed at 0.27 m/s (see section 4.6.4) and a settling velocity of 0.01 m/s feces theoretically can advected between 0 (feces produced by mussels at pycnocline) and 300 m (produced 2 m below surface) before sinking through the pycnocline, where currents are lower (median at 0.075 m/s, Table 4.7) and often in opposite direction of the current in surface waters. Due to the roughness created by the mussel cover currents near to the pylons will be turbulent therby increasing horizontal dispersion and reducing advection. Given the complexity of environmental factors affecting deposition of feces, reliable explicite modelling would require extensive experimental work for model calibration. As an alternative approach it is assumed that feces are deposited evenly in a 5 m broad strip along the side of pylons and in a 20 m wide and 50 m long area in either end of pylons giving a total area of 2600 m^2 for feces sedimentation. The assumptions build on measurements of feces sedimentation within and outside bivalve cultures reviewed by (Barnes, 2006).

The oxygen demand (OD) of sedimenting feces can be estimated by:

$$OD = 2200/2600*2.4 = 2.03 \text{ g oxygen}/m^2/d,$$

where 2.4 is the O:C stochiometric ratio for aerobic oxygen degradation of carbon in feces. The predicted oxygen demand is 5-10 higher than the oxygen uptake estimated by the water quality model, indicating that benthic communities will be stressed in these areas.

Fall-off of mussels will occur regularly at small rates and presumably on rare occations as mass fall-off if anoxia delevops in the inner part of the mussel matrix. Death of mussels will weaken the attachment of matrix to the pylon. Compared to sedimentation of feces the area of seabed affected would be much smaller because high settling velocity (in range of meters per second). Detached mussels on the seabed would serve as potential food for epibenthic predators and scavengers.

The additional oxygen demand on the seabed will not affect oxygen concentration in bottom water passing through bridge pylons, because fluxes are 3-5 orders of magnitude higher than the demand.

4.7 Hydrographical regime

Hydrographic regime and especially the intensity of vertical mixing have strong influences on water quality and plankton. Long-lasting density stratification is a prerequisite for development of oxygen deficiency in bottom waters because exchange of oxygen with atmosphere is prevented (e.g. Stanley and Nixon 1992, Møhlenberg 1999). Besides preventing oxygen deficiency in bottom water, vertical mixing structures the composition of plankton communities and increases coupling between benthic grazers and phytoplankton and thus promotes benthic grazing control.

4.7.1 Hydrographic regime – tunnel solution

The assessment of changes in the hydrographical regime shows only very small and local effects on current speeds in vicinity of reclamation areas (FEHY 2013b). Larger scale effects on water column stratification could not be demonstrated and accordingly, neither water quality nor plankton will permanently be affected by the tunnel solution.

4.7.2 Hydrographic regime – bridge solution

The bridge structures will lead to increased vertical mixing of water passing piers and pylons. East of the alignment in the main stem of Fehmarnbelt, the density stratification (difference between bottom and surface waters) will be reduced by up to 0.20 kg/m^3 especially during summer, while density stratification will increase in the Mecklenburg Bight (up to 0.12-0.16 kg/m^3), see Figure 4.10.

Comparable results were predicted using the local GETM model (FEHY 2013b).

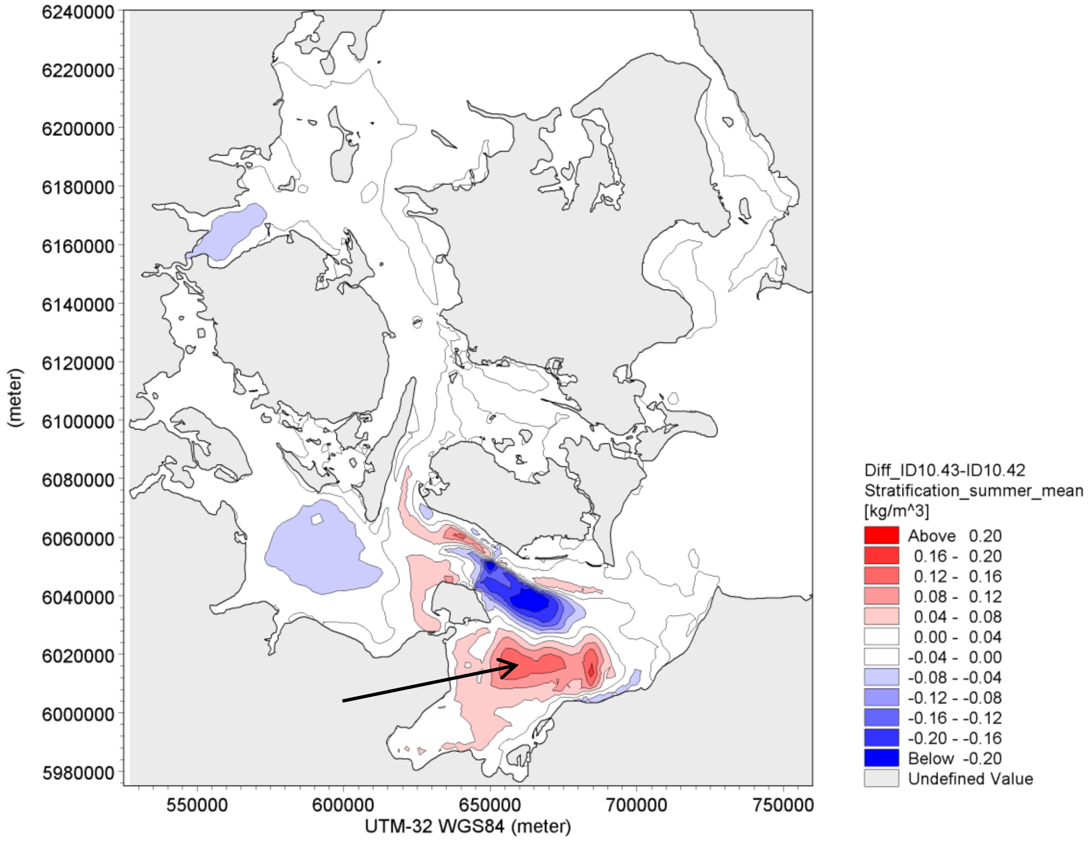


Figure 4.10 Predicted change in stratification during summer using MIKE 3 local model for "bridge+ferry" case, from (FEHY 2013b). Arrow indicates the position where vertical density profiles were extracted (see Figure 4.11).

The difference in water density between bottom and surface was due to a lower density in the the upper 10 m of the water column, rather than changes in bottom water (Figure 4.11).

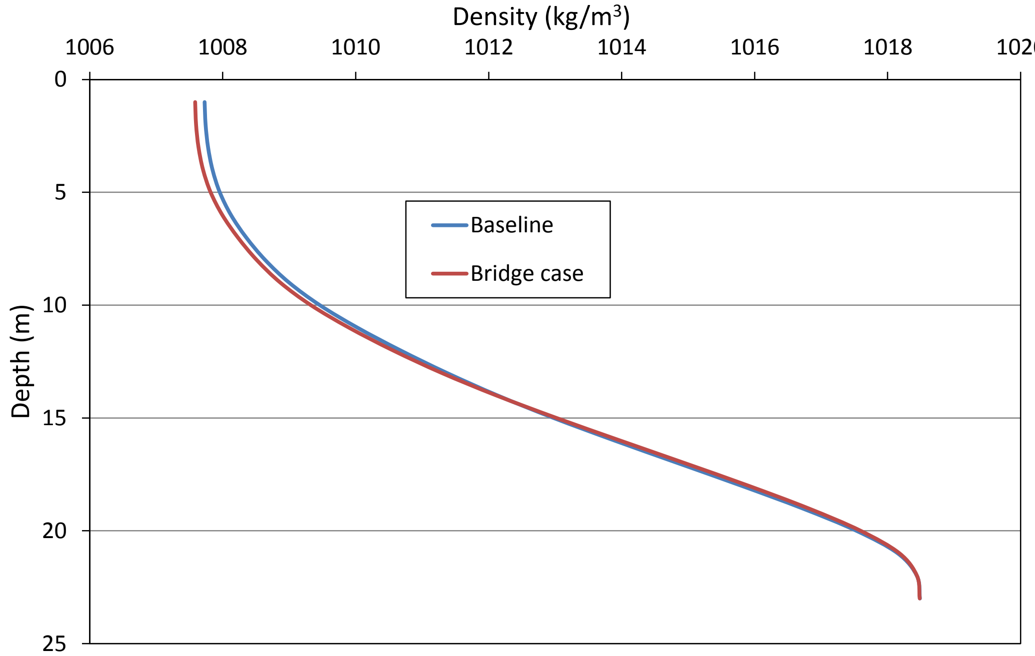


Figure 4.11 Depth distribution of water density for Baseline and "bridge + ferry" case in Mecklenburg Bight (average June-August). Based on (FEHY 2013b) model.

5 SENSITIVITY ANALYSIS

The most important aspects of Fixed Link related pressures on water quality, plankton production and biomass are modelled numerically because of well-established relationships between sediment spill, concentration of suspended matter and light attenuation, between light attenuation and phytoplankton growth and, between phytoplankton biomass and zooplankton production.

For toxic substances the predicted concentrations due to release from dredged sediments are evaluated against environmental criteria (EQS) which are included in national legislation and as such, considered to be firm. Therefore, sensitivity (and uncertainty) is associated with the four variable assumptions (concentration in sediments, release rate from sediments, spill rates, current speed) used in calculation of pollutant concentrations. By using a Monte-Carlo approach uncertainties in assumptions are carried through calculations to pressure concentrations. Therefore, uncertainties can be estimated from distribution function of pollutant concentrations.

For other processes that are less straightforward and where documentation of dredging-related effects is less extensive, prediction of potential impacts is made by a combination of numerical model outputs (including concentrations, process rates such as natural sedimentation) and information from the scientific literature. In the following, sensitivity and relevance of facilitated sedimentation of phytoplankton, changes in phytoplankton composition due to shading from suspended sediments and, impacts on zooplankton from high concentration of suspended matter and due to burial of resting eggs.

5.1 Facilitated sedimentation of phytoplankton

Sedimentation is an important process of biomass loss of phytoplankton from the water column (and enrichments of sediments). The sedimentation rates of phytoplankton cells increase through aggregation with other suspended particles, which could result from interactions of plankton cells during strong bloom events or from interactions of phytoplankton cells with suspended clay particles. The process is often called 'flocculation' (Verspagen 2006) or 'coagulation' (Boehm and Grant 1998). The processes of aggregation would lead to the formation and sedimentation of larger particles and thus to the clarification of water. Aggregates have different forms and sizes and three size classes are distinguished: 1) macroscopic aggregates >150 µm; 2) micro aggregates <150 µm; and 3) submicron particles <1 µm (Zimmermann-Timm 2002; Alldredge and Silver 1988).

For phytoplankton cells the probability and extent of aggregation depend on species-specific presence of extracellular polymers (e.g. Avnimelech et al. 1982; Engel 2004), as well as the concentration of cells, the concentration and size distribution of inorganic particles and also the mineralogy of particles (e.g. Søballe and Threlkeld 1988; Jackson 1990; Zimmermann-Timm 2002; Guenther and Bozelli 2004). Even though models, e.g. by (Riebesell and Wolf-Gladrow 1992) can describe the role of particle size, stickiness, form and concentration on aggregation, a suite of other processes that are not know in detail will make predictions uncertain unless site-specific "stripping" experiments are carried out (Hamm 2002).

"Stickiness" is defined as the probability of 2 colliding particles (phytoplankton cells) stick together, with values ranging from −1 to 1. Negative stickiness indicates that particles fall apart when they collide, whereas positive values indicate that particles adhere when they collide. For example a value of 0.2 means that 20% of the particle collisions resulted in aggregation. Various studies (e.g. Kiørboe et al. 1990) demonstrated that several species of phytoplankton are significantly sticky and may coagulate under specific shear conditions. Later studies by (Kiørboe and Hansen 1993; Hansen and Kiørboe 1997; Waite et al. 1997; Dam and Drapeau 1995; Engel 2000 and Verspagen et al. 2006) strongly indicate that diatoms flocculate more readily than other taxa, but also that flocculation increases when cells become nutrient limited such as in the late phase of the spring bloom.

5.1.1 Natural aggregation in blooms

Aggregation of phytoplankton cells increases with concentration and with turbulence in the water column (Hietanen 1998; Jackson 1990; Riebesell 1991a,b; Kiørboe et al. 1994). At the beginning of a bloom, characterised by low biomass and abundance, there is no coagulation and sedimentation of cells. If the concentration of cells reaches a critical limit (see Table 5.1), which depends on cell sizes, cell surface, stickiness, and turbulence of water, the entire bloom might aggregate very fast followed by fast sedimentation (Alldredge and Gotschalk 1989; Leppänen 1988; Leppänen and Kononen 1988). Aggregation and sedimentation processes are more important in turbulent environments e.g. in windy periods than in stagnant waters (Table 5.1) because collision rates increase with turbulence.

The aggregation of cyanobacterial blooms differs from that of e.g. diatoms (Hietanen 1998), because the intracellular gas vacuoles prevent sedimentation and keeping aggregated filaments in the upper water layer (Hoppe 1981; Worm and Søndergaard 1998).

Table 5.1 Critical phytoplankton concentration (C, cells ml^{-1}) and biomass (B, µg carbon l^{-1}) for coagulation control of plankton population size, calculated for various fluid shear rates (s^{-1}) and cell sizes. Table originated from modeled data of Kiørboe (1993). Limitation of phytoplankton population sizes is potentially important only when B > 2 x 10³ µg carbon l^{-1}, i.e. only for shadowed combination of cell sizes and shear rates in the table.

Cell diameter µm	**Shear rate (s^{-1}) 0.1**		**1.0**		**10.0**	
	C	**B**	**C**	**B**	**C**	**B**
1	$1.2\ 10^{9}$	$6.9\ 10^{4}$	$1.2\ 10^{8}$	$6.9\ 10^{3}$	$1.2\ 10^{7}$	$6.9\ 10^{2}$
10	$5.6\ 10^{5}$	$3.2\ 10^{4}$	$5.6\ 10^{4}$	$3.2\ 10^{3}$	$5.6\ 10^{3}$	$3.2\ 10^{2}$
100	$2.6\ 10^{2}$	$1.5\ 10^{4}$	$2.6\ 10^{1}$	$1.5\ 10^{3}$	$2.6\ 10^{0}$	$1.5\ 10^{2}$
1000	$1.2\ 10^{-1}$	$7.2\ 10^{3}$	$1.2\ 10^{-2}$	$7.2\ 10^{2}$	$1.2\ 10^{-3}$	$7.2\ 10^{1}$

5.1.2 Aggregate settling velocity

Settling velocity increases as a function of particle diameter often described by Stokes Law, however, settling velocities in the sea are often much higher than calculated from Stokes' law or measured on individual cells in the laboratory (see Table 5.2) because of aggregation of cells and particles in the natural environment. Usually, aggregates have a lower density as compared to the original primary particles because aggregates have a higher content of organic matter and/or containing more enclosed water. Aggregates dominated by mucopolysaccarids, cyanobacteria, faecal pellets and detritus have lower sinking rates than aggregates dominated by denser diatoms or inorganic particles (Zimmermann-Timm 2002; Grossart et al. 1998; Zimmermann-Timm et al. 1998).

Table 5.2 Settling velocity of Skeletonema costatum and Thalassiosira punctigera

Inorganic particles (10-50mg/l)	Algae	Settling velocity (m/d)	Reference
Lab	*Skeletonema costatum*	1.35	Smayda and Boleyn 1966
Field/mesocosm		30-50	von Bodungen et al. 1981
Lab (= Stokes Law)		0.07	Riebesell 1989
No clay (field)		4	
Kaolinite	*Thalassiosira punctigera*	250-320	Hamm 2002
Quartz		850	
Smectite		720	
Illite		560	
Kaolinite		480	
No clay		230	

In experiments with the diatoms *Skeletonema costatum* and *Thalassiosira punctigera* settling velocity increased by 100% to 1000% if suspended clay was added (Table 5.2). Settling velocity of aggregates increased with density of inorganic particles, i.e. quartz-containing aggregates had almost the double settling velocity than aggregates containing kaolinite.

In the Fehmarnbelt region the settling velocities of natural sediments, without aggregation involving biological sources, were calculated to 0.08 - 544 m d^{-1} for sediments with grain sizes between 0.002 mm and 0.15 mm (FEHY 2013a).

5.1.3 Removal of plankton cells by artificial induced flocculation

While there are few - if any - published examples of increased sedimentation of phytoplankton in connection with dredge-related sediment spill several studies have quantified the efficiency of various clay types and concentrations to strip toxic phytoplankton blooms from the water column in order to protect fish farms (Table 5.3). However direct application of results in connection with sediment spill from dredging operation is difficult because nature of inorganic solids will be different and a much higher algal biomasses used in experiments (0.6-8 mg Carbon/l) summarised in Table 5.3 compared to typical algal concentrations under a spring bloom (0.3-0.8 mg C/l) in the Fehmarnbelt.

Table 5.3 Removal of plankton cells by sedimentation (% of reduction) in dependence from phytoplankton species and clay concentration. The table was compiled from various literature sources.

Concentration of clay	Investigated species (Abundance)	Removal of plankton cells by sedimentation	Reference
25mg l^{-1} phosphatic clay (IMC-P)	*Karenia brevis* (8 x 10^6 cells l^{-1})	90%[#1]	Sengco & Anderson 2004
50mg l^{-1} phosphatic clay (IMC-P)	*Karenia brevis* (13 x 10^6 cells l^{-1})	97%[#1]	
25mg l^{-1} phosphate clay (diverse material)	*Karenia brevis* (4 x 10^6 cells l^{-1})	<20%[#1]	
25mg l^{-1} phosphatic clay (IMC-P)	*Heterocapsa triquetra* (0.2 x 10^6 cells l^{-1}) Flume at 3 cm/s	100%	
	Heterocapsa triquetra (0.2 x 10^6 cells l^{-1}) Flume at 10 cm/s	89%	
	Heterocapsa triquetra (0.2 x 10^6 cells l^{-1}) Flume at 20 cm/s	41%	
36mg l^{-1} kaolin clay	*Prorocentrum minimum* 1.4 x 10^7 cells l^{-1})	100% (2.5h-4d)	Brownlee 2005
	Chattonella 1.7 x 10^6 cells l^{-1})	93% (2.5h-4d)	
	Cyanobacteria 1, 2.1 x 10^{10}	13-33% (2.5h-4d)	
	Cyanobacteria 2.2 x 10^9 cells l^{-1})	18-71% (2.5h-4d)	
250 mg l^{-1} clay (diverse)	Karenia brevis (7-10 x 10^6 cells l^{-1})	>90%	Sengco et al. 2001
250mg l^{-1} clay (diverse)	*Aureococcus anophagefferens* (3-5 x 10^9 cells l^{-1})	<40% 80% with brief agitation,	
100 mg l^{-1} clay	*Aureococcus anophagefferens* (7 x 10^9 cells l^{-1})	2-18%[#1]	Yu et al. 2004
200 mg l^{-1} clay		5-35%[#1]	
400 mg l^{-1} clay		35-50%[#1]	
600 mg l^{-1} clay		57-65%[#1]	
250 -750 mg l^{-1} (kaolinite/bentonite)	*Microcystis* strain V145 (70 x 10^6 cells l^{-1})	<10%[#1]	Kiørboe & Hansen 1993
10 mg l^{-1} kaolinitic clay	*Staurodesmus convergens* (0.5 x 10^6- 2 x 10^6 cells l^{-1})	0% (after 1 h) 53% (after 12 h)	Guenther & Bozelli 2004 [#2]
30 mg l^{-1} kaolinitic clay		3% (after 1 h) 87% (after 12 h)	
50 mg l^{-1} kaolinitic clay		16% (after 1 h) 96% (after 12 h)	
10 mg l^{-1} kaolinitic clay	*Phormidium amoenum* (0.5 x 10^6- 2 x 10^6 cells l^{-1})	20% (after 1 h) 61% (after 12 h)	
30 mg l^{-1} kaolinitic clay		16% (after 1 h) 68% (after 12 h)	
50 mg l^{-1} kaolinitic clay		15% (after 1 h) 78% (after 12 h)	
10 mg l^{-1} kaolinitic clay	*Chlorella sp.* (0.5 x10^6- 2 x 10^6 cells l^{-1})	35% (after 1 h) 31% (after 12 h)	
30 mg l^{-1} kaolinitic clay		54% (after 1 h) 32% (after 12 h)	
50 mg l^{-1} kaolinitic clay		69% (after 1 h) 57% (after 12 h)	

[#1] values read from figures in article
[#2] % values calculated from original abundance data

5.1.4 *Relevance for the dredging impacts on phytoplankton sedimentation*

The sensitivity of phytoplankton to increased sedimentation in presence of high concentrations of suspended matter from dredging spill cannot be quantified with high precision. However, based on relevant studies summarised above the probability for facilitated sedimentation will be highest during the spring bloom because diatoms (especially *Skeletonema costatum*) dominate the spring bloom (WQ-Plankton Baseline) and because diatoms are most prone to facilitate aggregation with suspended matter because of a high stickiness. During autumn facilitated sedimentation is less likely because algal concentrations are lower thaN during spring bloom and the stickiness of diatoms will be lower because phytoplankton is less nutrient limited. With an equivalent cell diameter of 10 µm *Skeletonema costatum* will begin to aggregate at a critical biomass of ca. 300 µgC/l at high turbulent intensity (see Table 5.1). In calm waters the critical biomass need be much higher.

By adding clay (e.g. spilled sediments) the critical biomass of *Skeletonema* is lower because of increased probability of collision between algal cells and inorganic particles. Apart from differences between algal species a concentration of 10 mg/l of clay particles will "remove" between 30 and 61% of algal cells from the water column within 12h (Table 5.3). *Skeletonema*-clay aggregates will have a settling velocity of about 250-300 m/d (Table 5.2) and will reach the seabed within 1-2 hours.

5.2 *Effect of sediment spill on phytoplankton composition*

The phytoplankton populations in estuaries are largely controlled by abiotic parameters: light and nutrient availability, mixing and temperature, and in Fehmarnbelt particularly the seasonality in these parameters was found to be the main driving force on the overall phytoplankton community development (FEMA-FEHY 2013).

The pelagic primary production, where inorganic carbon is incorporated into organic molecules in the photosynthetic process, showed the highest rates in late summer of both investigation years. The primary production showed a strong exponential decrease in production rates over depth, which reflects the effect of light attenuation in the water column. The light attenuation constitutes an important challenge for phytoplankton. At a depth of 5 m the primary production was reduced to around 65% of the primary production in the surface and in 15 m it constituted only 10% (Fig. 4-17 in FEMA-FEHY 2013). Accordingly, achieving sufficient light for photosynthesis is one of the major challenges for phytoplankton.

During the baseline study the light attenuation was found to be highest during the declining spring bloom in Fehmarnbelt underlining that input of suspended matter and sediment resuspension events are of minor importance in deep parts of the Fehmarnbelt (FEMA-FEHY 2013). Whether additional reduction of light caused by sediment spill during the construction phase of the fixed link will pose additional stress on the phytoplankton populations will be addressed in the following.

5.2.1 *Effect of light attenuation and water column stratification on the different phytoplankton groups*

The different phytoplankton groups have various functional adaptations to maximize survival and growth in light-limited environments. Additional concentration of suspended matter (sediment spill) in the water column will add to light attenuation and will also lead to changes in light spectrum because the length of light path is increased by scatter, and a longer light path will increase probability of photons intercepting with organic matter resulting in absorption. Consequently, additional suspended sediment in the Fehmarnbelt theoretical will force a change in the composition of phytoplankton towards algae with blue and red pigments (e.g. cyanobacteria and cryptophytes), which are especially well adapted to take up light in water columns where the spectral composition of light is shifted towards longer wavelengths (e.g. Dubinsky and Berman 1979).

In Fehmarnbelt the spring bloom is initiated by diatoms (Wasmund et al. 2008, FEMA-FEHY 2013) which can take advantage of the high nutrient concentrations in a well-mixed water column and attain high biomasses at low temperatures when the incident light increases in February/March, but when light intensity still limits other phytoplankton groups (Sarthou et al. 2006). If light attenuation is increased by sediment spill, a delay in the spring bloom can be expected, but only locally in the sediment plume. The onset of spring bloom differs from year-to-year, which was also observed in the baseline investigation where the spring bloom was delayed one month 2010 due to ice compared to the bloom occurring in February in 2009.

Depletion of nutrients causes collapse of the spring bloom. During the post-bloom period smaller phytoplankton flagellates develop and can utilise the residual and regenerated nutrients in concentrations which are too low for the larger spring bloom algae (Andersson et al. 1994). During the post-bloom period in early summer where the pycnocline is strengthened (FEMA-FEHY 2013) the water temperature is still low, and motile algal have an advantage: these species make diurnal migrations into deeper nutrient-rich water during night, take up nutrients and ascend to the euphotic zone during day to harvest light for photosynthesis (Eppley et al. 1968, Cullen and Holligan 1981 Olli 1999). Dinoflagellates are one of the important groups which can migrate (e.g. Cullen and Horrigan 1981, Olsson and Granéli 1991), but the ability has also been demonstrated for cryptophytes, euglenophytes, the important autotrophic ciliate *Mesodinium rubrum*, rhadidophytes and other flagellates (Watanabe et al. 1991, Villarino 1995, Olli 1999). For non-flagellated species such as diatoms and cyanobacteria buoyancy regulation has been demonstrated as a mean for adjusting their position in the water column for resource optimization (Cullen and MacIntyre, 1998 and references herein).

If light in the Fehmarnbelt waters is further reduced by sediment spill the migration distance needed to bring the motile algae into the euphotic zone theoretically will be longer. Probably, the reduction in Secchi depth (m) will be a good estimate of the extra travel distance for phytoplankton to reach sufficient light. Investigations in the Baltic Sea have shown that phytoplankton can migrate 30m during one dial cycle (Olli 1999; Kononen 2001), while buoyant colonies of the cyanobacteria *Aphanizomenon flos-aqua* have showed a net-upward move of 20m in the Baltic Sea (Walsby et al. 1997). Hence, we assume that dredging locally give rise to a reduction in Secchi depth of 2-3 m migrating phytoplankton may have to travel an extra 10% distance to meet optimal light conditions or they will experience a lower growth rate.

In mid and late summer high temperatures combined with low nitrogen concentrations cause a domination of cyanobacteria and various flagellates (FEMA-FEHY 2013). Filamentous cyanobacteria, particularly the harmful, toxic *Nodularia spumigena*, form recurrent dense surface blooms in the Baltic Sea and the (Eastern) Fehmarnbelt area (Kononen, 1992) and by being buoyant and capable of fixing nitrogen from the atmosphere *N. spumigena* and a few other buoyant cyanobacteria, create large blooms of great nuisance (Kahru et al. 1994, Kanoshina et al. 2003, Schlüter et al. 2004).

The effect of increased light attenuation caused by sediment spill in the water column will obviously not affect <u>surface blooms</u> of cyanobacteria. However, below the surface the light available for the remaining part of the phytoplankton population will be reduced.

In late summer the non-cyano part of the phytoplankton population consists primarily of various flagellates (FEMA-FEHY 2013; Schlüter et al. 2004), and can constitute more than one-third of the phytoplankton biomass even during a dense bloom of *N. spumigena* (Schlüter et al. 2004). As described above under sediment spill the migrating phytoplankton may have to ascend further up in the water column to reach optimal light conditions but calm periods blooms of cyanobacteria will capture a large part of incoming light and other algal groups will experience sub-optimal light conditions. Therefore, sediment spill probably will favour cyanobacteria compared to other groups, at least during periods of calm weather.

During autumn the pycnocline gradually degrades and larger species such as dinoflagellates and diatoms become increasingly important due to introduction of new nutrients into surface waters by entrainment. Diatoms and dinoflagellates co-exist during autumn: during calm periods permits surface aggregation of dinoflagellates which become homogeneously distributed when turbulence intensifies at mixing periods (Lauria et al. 1999). Diatoms, conversely, rely on periods of increased turbulence to ensure entrainment into the upper water column and to prevent sinking from the photic zone during stable intervals (Estrada and Berdalet 1997, Lauria et al. 1999). Diatoms are able to grow at very low light intensities (<1 µE m^{-2} s^{-1}, Richardson et al. 1983) and are consequently well adapted to situations where additional sediment is introduced into the water column. Therefore, diatoms are considered to have an advantage over dinoflagellates in situations of increased light attenuation in Fehmarnbelt due to sediment spill.

5.2.2 Relevance of dredging impacts on phytoplankton composition

Light requirement for optimal growth differs among phytoplankton groups and dredging-related sediment spill theoretical can lead to changes in growth conditions and in turn to shifts in different groups' dominance. We expect that diatoms will be favored by dredge-related reductions in light availability during spring and autumn because diatoms have very low requirements for light, and it is also likely that cyanobacteria can be favored by sediment spill during summer because they can regulate buoyancy and accumulate at sea surface thereby escaping light reduction in water column.

It must be stressed, that predictions are based on theoretical arguments only, and that changes in phytoplankton composition to our knowledge have not been reported in connection with dredging works.

5.3 Impact of excess sediment on zooplankton

Zooplankton communities are, apart from various environmental variables, affected from "bottom up" processes by available food resources (phytoplankton, microzooplankton) as well as "top down" by predation pressure (fish, jellyfish). Hence, the potential impairment of increased concentration of suspended sediments (SS) on zooplankton due to dredging is associated with (1) the reduction of food supply due to reduced primary production caused by light attenuation, (2) reduced energy uptake of zooplankton species due to a decreased food/inorganic particle ratio and mechanical inhibition of feeding efficiency and, (3) decreased predation pressure of visual predatory fish.

5.3.1 Impact through reduction in primary production

Based on a 11-year time series of historical data from two stations (St 46 – Kadett Rende; St 12 – east of Fehmarn, see Figure 3.8) there is a significant ($p<0.05$)

positive relation between concentration of chlorophyll-a and biomass of holozooplankton (sum of copepods and daphnids, Figure 5.1), indicating that the zooplankton community overall is food limited. However, the data show a large scatter probably as a result of year-to-year variation in loss processes such as predation pressure on zooplankton. Interestingly, the upper boundary of data shows an almost one-to-one relationship between concentration of food and biomass of zooplankton.

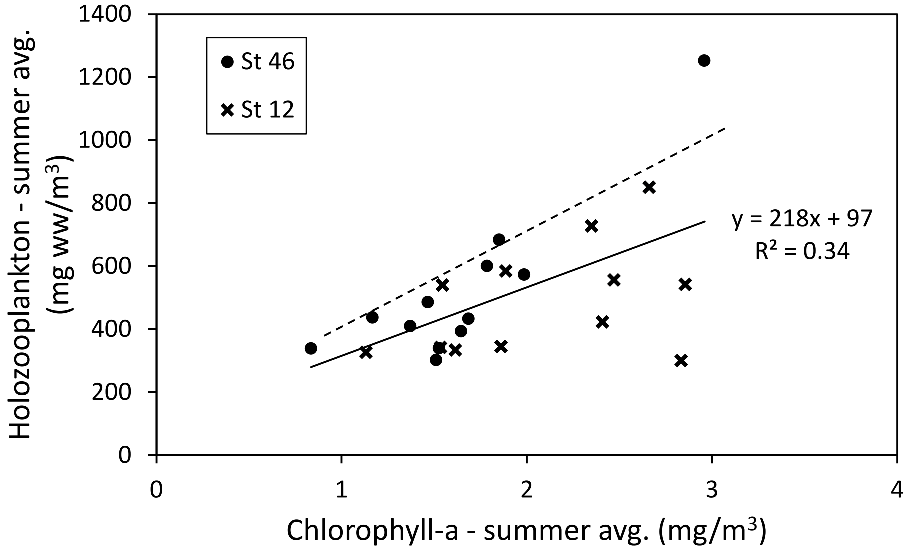


Figure 5.1 Scatter-plot between chlorophyll-a (phytoplankton biomass) and biomass of holozooplankton at two stations in the Fehmarnbelt (1998-2009). Regression line (solid line) and equation encompassing data from both stations are shown. Dashed line indicates the upper boundary of data (drawn by eye). Based on data from Helcom database

Effects on the zooplankton community mediated through reduction in primary production are well-described and based on numerous studies. Accordingly, these effects can be and are explicitly modelled.

5.3.2 *Physical interference of feeding by suspended matter in zooplankton*

Direct impacts of sediment on zooplankton have been conducted at suspended solids concentrations ranging from 0-1000 mg l^{-1} which are relevant for the assessment of the expected impact of sediment spills. These studies indicate that microzooplankton (ciliates, some meroplankton) is relatively insensitive to suspended solids as effects on feeding or growth was very small or not found (Table 5.4) (Hansen et al. 1991, Boenigk and Navarino 2004). In contrast, a number of negative direct effects have been among mesozooplankton, such as copepods, cladocerans, and bivalve larvae.

Table 5.4 Summary of responses to elevated levels of Suspended Solids (SS) for other zooplankton taxa. NTU (Nephelometric turbidity units) can roughly be converted to SS concentration (mg/l) by multiplication with a factor of 2.

Taxon	**habitat**	**SS concentration, duration**	**Response**	**Reference**
Keratella cochlearis, Keratella crassa, Synchaeta pectinata	laboratory culture	laboratory: 50 mg/l (68 NTU)	no effect on feeding at high SS, because the Rotifera fed selectively and were thus able to avoid ingesting clay particles (in contrast to Cladocera)	Kirk 1991b
Favella ehrenbergii (Tintinnida) *Brachionus plicatilis* (Rotifera)	laboratory culture	laboratory (latex beads): 0 – 40 mg/l	insensitive, no reduction of feeding rates	Hansen et al. 1991
Philine aperta (Gastropoda Veliger)	laboratory culture	laboratory (latex beads): 0 – 40 mg/l	Veligers changed swimming behaviour by increasing sinking and jumping frequencies accompanied by a significant reduction in clearance	Hansen et al. 1991

The interference of high concentrations of suspended sediment with feeding is expected to be stronger in non-selective suspension feeders (e.g. cladocerans) than in selective feeders (e.g. copepods). In accordance, freshwater studies generally show negative impacts (feeding and growth) at lower sediment concentrations on cladocerans than on copepods (Arruda et al. 1983, Kirk 1991a, Kirk 1992, Hart 1992, Bozelli 1998, Levine et al. 2005, Soeken-Gittinger 2009), see Table 5.5. The mechanisms affecting the non-selective filter feeding cladocerans are related to the clogging of animals' filter-feeding structures by sediment particles, the incorporation of sediment particles as well as decreasing ingestion rates of energy rich food particles. Effects of suspended solids can thus result in a decrease of some species and accordingly, lead to changes in community structure.

Table 5.5 Summary of responses to elevated levels of Suspended Solids (SS) for cladocera. NTU can roughly be converted to SS concentration (mg/l) by multiplication with a factor of 2.

Taxon	Habitat	SS concentration, duration	Response	Reference
Daphnia lumholtzi (invasive) / *Daphnia parvula* (native)	river, temperate, USA	natural: 0-90 (120) mg/l laboratory: 0-80 mg/l 26 days	fertility of native *D. parvula* decreased at high SS (80 mg/l) in contrast to invasive *D. lumholtzi* mechanisms discussed: (1) high SS interfers directly with feeding (decreasing ingestion rates) and (2) indirect impact of high SS by light limitation for primary production (depression of food resources)	Soeken-Gittinger 2009
Daphnia barbata, D. gibba, D. laevis, D. pulex, Moina micrura, Diaphanosoma excisium	fresh water lakes, South Africa	natural: 7-70 mg/l laboratory: 0 mg/l 10-70 mg/l 70-235 mg/l (40 days)	results not consistent, Zooplank grouped into "turbid-water-species" (*D. barbata, D. gibba*) and "clear-water-species" (*D. laevis, D. pulex*) mechanisms discussed: (1) morphology of filtering limbs differs species specific (2) indirect impact of high SS by light limitation for primary production (depression of food resources)	Hart 1992
Daphnia carinata	fresh water lake, temperate, New Zealand	natural: 5-15 (100) NTU laboratory: 2.2, 10, 20, 60, 100 NTU (4 days)	inhibition of clearance of phytoplankton, flagellates and ciliates by 72-100%, and of amoebae and attached bacteria by 21-44%	Levine et al. 2005
Daphnia ambigua	laboratory culture	laboratory: 50-200 mg/l	clay (>2µm) reduced algal ingestion by 29-87%, fine clay (<1µm) had no effect mechanisms discussed: clay particles mechanically interfere with filtration of algal cells effects: lower body lengths, survival reduced, age of maturity and reproduction increased, fitness decreased by 70%, reduced population growth	Kirk 1991a, 1992
Diaphanosoma birgei, Moina minuta	fresh water lake, tropical, Brazil	laboratory: 10-200 mg/l	carbon ingestion rates declined with increasing SS (coarse clay particles up to 20 µm)	Bozelli 1998
Daphnia parvula, D. pulex, D.similis	Fresh water reservoir	laboratory: 0-2451 mg/l 50-100 mg/l	Ingestion rate of ^{14}C labeled Chlorella decreased by 95%, incorporation rate decreased by 99% Ingestion rate decreased to starvation level	Arruda et al. 1983

Reviewing numerous experimental studies effects of high concentration of suspended matter on copepods it is evident that impacts are not consistent. High concentration of SS can affect ingestion rates, swimming behaviour, vertical distribution in the water column of copepods and, egg production, while the impact on copepod population development in general was minor, which indicate that as long concentration of food is sufficient for selective feeding copepods in spite of increased suspended sediment concentrations (Table 5.6). In a few studies, impacts were inversely related to the sediment load at high concentration (Castellani and Altunbas 2006, Gasparini et al 1999). The impact of high sediment concentrations on copepod physiology may not exclusively depend on the amount of sediment present but also on the availability of food: a negative relation of egg production to the sediment load has been observed at low food availability, while at high food concentration negative effects on reproduction were much less pronounced (White and Dagg 1989). Starvation apparently amplifies the negative effect, indicating that physiologically stressed animals are more sensitive (White and Dagg 1989). Therefore impacts are likely to be more pronounced during the period of low food availability in late spring/summer than during diatom spring bloom and during autumn.

Table 5.6 Summary of responses to elevated levels of Suspended Solids (SS) for copepoda. NTU can roughly be converted to SS concentration (mg/l) by multiplication with a factor of 2.

Taxon	Habitat	SS concentration, (duration)	Response	Reference
Boeckella hamata	temperate lake,	natural: 5-15 (100) NTU laboratory: 2.5, 100 NTU (4 days)	inhibition of clearance of microzooplankton by 56% at increased turbidity	Levine et al. 2005
Tropodiaptomus spectabilis, Metadiaptomus colonialis	fresh water lakes,	natural: 30 *T.spectabilis* and 95 NTU *M.colonialis* laboratory: 0-200 NTU, (50 days)	results not consistent, SS free water: development of both species failed invariably, high (enriched) SS: development of *T. spectabilis* faster, but development of *M. colonialis* slower	Hart 1991
Diaptomus ashlandi	fresh water lake, temperate	natural: 0-20 NTU laboratory: 0 – 200 NTU	ingestion rate of algae decreased at high SS, but the natural population increased during periods of high SS, indicating sufficient food supply and feeding	Butler 1995
Diaptomus sp., *Cyclops* sp.	glacial lakes, Alaska	natural: 5-120 NTU	selective feeding herbivore calanoids and carnivore predators are much less sensitive to higher turbidity than non-selective filter feeding cladocerans	Koenings et al. 1990
Copepoda	fresh water lake, temperate,	in situ, experimental clay loading 100 g m^{-2} d^{-1}, Secchi depth decrease up to 75% (4 weeks)	higher turbidity caused changes in vertical distribution: in clay treatments copepods significantly concentrated in surface water (1m), in natural turbidity conditions copepods concentrated in a deeper layer (3m)	Cuker 1987
Acartia tonsa, Eurytemora affinis	estuarine	laboratory: 0 – 1000 mg/l	reduction of food ingestion rate: *E. affinis* >100 mg/l, *A. tonsa* at all increased concentrations, both species non-selective suspension feeders	Sherk et al. 1975
Acartia tonsa	estuarine	laboratory: 0 – 1000 mg/l	starved *A. tonsa*: Reduction of egg production by up to 40% at a sediment concentration of 250 mg/l and further reduction at higher sediment concentrations well-fed *A. tonsa*: no effect of high SS concentrations on egg production up to 500 mg/l, reduction of egg production at 1000 mg/l	White & Dagg 1989

Taxon	Habitat	SS concentration, (duration)	Response	Reference
Temora longicornis	marine, eastern Irish Sea, tidal habitat	naturally, seasonal changing: 0 – 50 mg/l	egg production rate was reduced at increased concentration of suspended matter	Castellani & Altunbas 2006
Eurytemora affinis	Estuaries, Europe	naturally: 0 – 1000 mg/l	*E. affinis* egg production rate was was reduced at very high concentration of suspended matter	Gasparini et al. 1999
Acartia tonsa	Laboratory culture	laboratory (latex beads): 0 – 40 mg/l	The copepodites changed swimming behaviour by increasing sinking and jumping frequencies accompanied by a significant reduction in clearance	Hansen et al. 1991

5.3.3 Impact of sedimentation on zooplankton resting eggs

Dredging and sediment spill during the construction phase potentially affect standing stocks of zooplankton populations due to burial of resting eggs and a presumably reduced hatching and recruitment rate. Several species of rotifers, cladocerans and especially copepods occurring in the Fehmarnbelt are known to produce resting eggs which are "stored" in the sediment (Viitasalo and Katajisto 1994, Madhupratrap et al. 1996, Marcus 1996).

On large scales (regional - ecosystem) resting eggs are important for maintaining copepod populations that seasonally disappear from the water column (Marcus 1996). In the Baltic Sea, hatching of resting eggs from the sediment is particularly important for the recruitment of *Acartia* species during the winter-spring transition (Katajisto et al. 1998). A crucial point for the hatching is the vertical distribution of eggs in the sediment and survival rate of eggs, since eggs do not hatch if buried (Kasahara et al. 1975, Uye et al. 1979, Marcus 1996) probably due to low oxygen concentration in sediment (Kasahara et al. 1975, Uye & Fleminger 1976, Uye et al. 1979). However, deeply (25 cm) buried resting eggs stay viable for several years (Marcus et al. 1994).

Buried resting eggs may be transported back to the sediment surface by resuspension or by bioturbation of the benthic infauna (Marcus and Schmidtgengenbach 1986). The longevity of buried resting eggs varies the species, on the type of the resting eggs (subitaneous or diapause eggs) as well as on the environmental conditions (Table 5.7). Survival of resting eggs is greatly affected by temperature with longer survival in cold waters/sediments (reviewed in Marcus 1996). In laboratory studies viability of resting eggs under anoxia varied between 6 months (e.g. *Centropages hamatus*) and 8 days (e.g. *Acartia tonsa*) (reviewed in Marcus 1996). In contrast, field studies have shown that viable resting eggs can be found from deeper sediments as old as 40 yrs (Marcus et al. 1994). In the Baltic Sea sediments viable eggs of *Acartia bifilosa* and *Eurytemora affinis* may be up to 19 year old (Katajisto 1996), but with a decrease in viability of eggs older than 7-8 years (8 cm sediment depth). Considering the fact that viable resting eggs occur in deeper sediment layers for decades, the existence of an "egg bank" in the Baltic Sea sediments is expected (Madhupratrap et al. 1996) as also described for other marine systems (Marcus et al. 1994).

Table 5.7 Summary of responses to elevated levels of sedimentation for resting eggs of zoo-plankton species.

Taxon	**habitat**	**Sediment thickness**	**Response**	**Reference**
A clausi, A. tonsa, Acartia sp. (spawned in laboratory)	California USA	laboratory: sediment layer of 0.3 to 0.5 cm, temperatures 17.5 and 3.5°C	hatching success after coverage with mud variable between species and temp.: *A. tonsa* (20 days at 3.5°C; 5d at 17.5°C) *A. clausi* (45d at 17.5°C) *Acartia* sp. durable for 25d (17.5°C) and 55d (3.5°C), hatching rates at 3.5°C higher in all species	Uye and Fleminger 1976
Calanopia thompsoni, Labidocera bipinnata, Acartia erythraea, A. clausi, Centropages abdominals, C. yamadai	central part of Inland Sea of Japan	field samples of buried eggs, parameters tested: mud coverage, temperature, oxygen, light, salinity	mud coverage: no hatching temperature: hatching success greatest at habitat temperature oxygen: no hatching <0.1 ml O_2 l^{-1} light: low effect salinity: low effect	Uye et al. 1979
Acartia bifilosa, Acartia sp., *Eurytemora affinis*	Northern Baltic Sea	field samples of buried eggs, (0 - 10 cm)	hatching success greatest in the upper 5 cm of sediment and decreased in deeper layers hatching inhibited, when eggs are buried,	Katajisto et al. 1998
Tortanus forcipartus	marine, Japan	laboratory	none of the eggs buried in the mud hatched within 3 days of incubation, while 80% of those on top of the mud hatched.	Kasahara et al. 1975
Acartia bifilosa, Eurytemora affinis	marine, Baltic Sea	field study, sediment layer 0-25 cm	hatching success was greatest in the upper 10 cm of sediment and decreased in deeper layers	Katajisto 1996
Brachionus, Tortanus discaudatus, Podon polyphemoides, Acartia hudsonica, Eurytemora affinis	marine, Rhode Island, USA	field study, sediment layer 0-24 cm	hatching success was greatest in the upper 5 cm of sediment and decreased in deeper layers	Marcus et al. 1994
Centropages hamatus, Acartia sp., Eurytemora affinis; A. tonsa,Podon polyphemoides, P. leuckarti, Evadne nordmanni	Baltic Sea, Kiel Bight	0-5 cm	Hatching success of copepod eggs collected from all depths was high (49 to 94%), but more variable (0 to 79%) for cladoceran eggs	Madhupratrap et al. 1996

5.3.4 Relevance of dredging impacts on zooplankton populations

Results from studies on effect of suspended matter on zooplankton are equivocal. Short-term studies at moderate concentrations of suspended solids (10-50 mg/l) do show effects on behaviour and feeding activity, but long-term studies even at much higher concentrations of SS do not show effects on population growth. However, studies indicate that sensitivity of zooplankton to excess SS increase at low food concentration. Because concentration of phytoplankton is low (average 2 µg chlorophyll-a/l) in Fehmarnbelt we may expect some impact on zooplankton at SS levels above 15 mg/l and high impacts at 50 mg SS/l and above.

Resting eggs of zooplankton that are permanently buried cannot hatch but are viable for several years. Bioturbation and resuspension events may bring buried resting eggs up to sediment surface allowing them to hatch. Long-term sediment accumulation of suspended matter from spilled sediments may thus delay or permanently prevent hatching. How effects of reduced recruitment from resting eggs affect Fehmarnbelt zooplankton populations will depend on the total area with long-term sediment cover of a considerable magnitude (i.e. 20-40 mm) and other sources of recruitment such as advection from non-affected areas.

5.4 Solid substrate

5.4.1 Piers promoting jellyfish populations

Jellyfish has an important role in coastal marine systems within pelagic food webs by acting as competitor and predator for zooplankton, fish larvae of commercially important planktivorous fish species. One effect of massive occurrence of scyphozoan medusae has been addressed to the settlement success of tiny planula larvae on suitable hard substrate and its development into inconspicuous polyp colonies. Single sessile polyps are able to reproduce asexually by budding further polyps as well as by strobilating several ephyrae (young medusae) for years. Increasing polyp populations can be followed by highly increased numbers of ephyrae, causing mass occurrences of jellyfish medusae, which in turn reproduce sexually and release a new generation of planula larvae. The settling success of pelagic planula larvae as well as the survival rate of released ephyrae are expected to be strongly affected by the predation of filter feeding invertebrates such as bivalvia and already established scyphozoans (Gröndahl 1988).

In the Baltic Sea, the moon jelly *Aurelia aurita* is the most abundant scyphozoan species in the Baltic Sea (Appendix C). Although its medusae generation appears widely distributed from the western Baltic Sea up to the Bothnian Gulf and the Gulf of Finland only a few polyp populations are reported from Kiel Bight (Schneider and Behrends 1998), Kerteminde Fjord (Olesen et al. 1994) and Gullmar Fjord (Hernroth and Gröndahl 1983). Strobilation of polyps and releasing of ephyrae mainly takes place between January and March in the western Baltic Sea. (Barz et al. 2006) demonstrated the spatial distribution of *A. aurita* ephyrae and medusae from the strobilation areas in Kiel Bight and Kerteminde Fjord into the central Baltic Sea by advection. Bridge pillars in the Fehmarnbelt may provide additional suitable hard substrate for *A. aurita* polyps and lead to a permanent increase of medusae densities in the western and central Baltic Sea. Enhancement of *A. aurita* populations by artificial hard substrate has been observed in Asia (Lo et al. 2008).

Solid substrate plays an important role in the life cycle of scyphozoan jellyfish because of their alternation between benthic polyp and planktonic medusa stages. Hard substrate in the upper water column is mainly occupied with sessile organisms such as blue mussels and macroalgae. Among this, benthic scyphozoan polyps may be exposed to competition for space, especially during spring and summer. Hard substrate below the pycnocline can be seen as a refuge from competition because scyphozoan polyps have a higher tolerance to hypoxic conditions compared to other epifauna (Ishii 2010).

Most population studies have been carried out using settling plates (Hernroth and Gröndahl 1983, 1985; Gröndahl 1988; Holst and Jarms 2007; Watanabe and Ishii 2001). Overall, artificial substrates such as shells, concrete, wood, polyethylene, glass were at least as attractive as stones and boulders for settling. Besides rocks, natural substrates include *Mytilus* shells, algae, the shells of living or dead barnacles, tunics of ascidians and hydrozoans (Thiel 1962; Hernroth and Gröndahl 1985; Miyake et al. 2002).

In situ settlement experiments carried out in the Fehmarnbelt during the Baseline study (see Appendix C) showed:

- that concrete was slightly more attractive for planula larvae than acrylic glass
- density of polyps was 5-10 times higher above pycnocline than below
- polyps preferred to settle on the down-side of plates, thereby avoiding competition for space from algae
- in the second season (autumn 2010 - spring 2011) average polyp abundance was ca. 2 ind cm^{-2} above pycnocline and, 0.2 ind cm^{-2} below pycnocline.

It is not known, but considered rather unlikely that the establishment of polyps had reached the final (stable) abundance after 2 years of exposure in the Fehmarnbelt. Still, using 2 individuals cm^{-2} as a minimum estimate the potential additional population size of polyps can be estimated from the new area of hard substrate (piers, pylons, scour protection) times the abundance on settling plates.

5.5 Hydrodynamic regime

Calm and stratified water bodies provide an environment where cyanobacteria usually have a competitive advantage over other non-buoyant phytoplankters. Bloom-forming cyanobacteria in the Baltic Sea have gas-filled vesicles inside their cells which enabling them to float. Under low-mixing periods cyanobacteria are able to ascend and accumulate at or near the surface, taking advantage of available light and high temperatures (during summer) for growth (Perovich et al. 2008, Pliński et al. 2007, Stal et al. 2003). Nitrogen fixation constitutes an additional advantage during the nutrient-poor period during summer.

In the Fehmarnbelt area, cyanobacteria on average constitute about 20% of the total phytoplankton biomass in the eastern part (Kadett Rende; Darss Sill), about 12% in the Mecklenburg Bight and, about 2% in the Kiel Bight (Figure 5.2). Between-year contribution is large and can partly be explained by meteorological conditions during summer (large contributions in the calm, sunny and warm summers 1997 and 2006). However, low contributions in warm summer of 2002 and a high contribution in the windy summer of 1998 do question a simple cause-effect relation between cyanobacteria and weather conditions during summer.

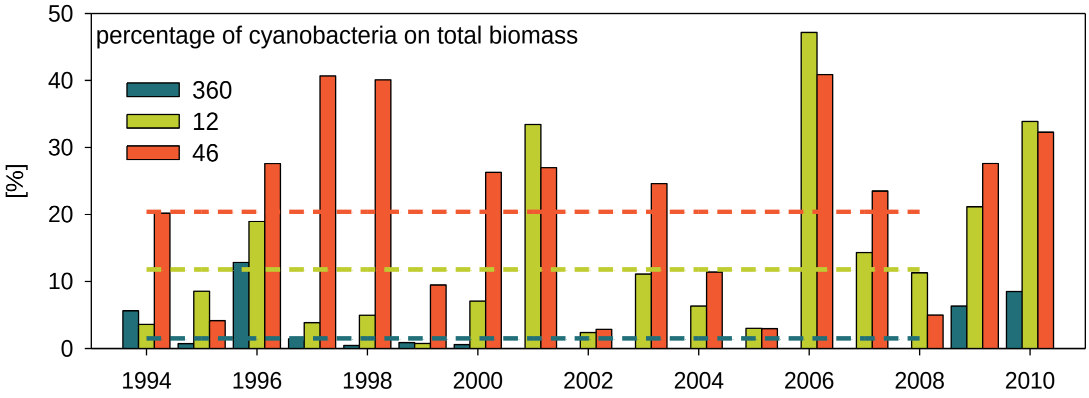


Figure 5.2 *Contribution of cyanobacteria to total phytoplankton biomass during summer in the Fehmarnbelt area. Dashed lines indicate average contribution of cyanobacteria to total biomass. See Figure 3.10 for location of stations.*

Still, based on an overwhelming number of literature references, increase in stratification will promote cyanobacteria and decrease in stratification strength will favour other phytoplankton on expense of cyanobacteria (see review by Perovich et al. 2008).

6 ASSESSMENT OF 0-ALTERNATIVE

In case that construction of a fixed link is abandoned, the existing ferry operation will continue. The impact on local and regional hydrography of ferry operation is negligible (FEHY 2013b,c) and in the absence of other changes in physical and chemical forcing impacts on water quality and plankton will also be negligible by a continued ferry operation. See also section 2.2.8.

7 ASSESSMENT OF IMPACTS OF MAIN TUNNEL ALTERNATIVE

7.1 Toxic substances

During dredging toxic substances and especially heavy metals will be released to the water column along with sediment spill (see Chapter 4). The degree of impairment, severity and significance of release depends on concentrations of substances, and to what extent "release-concentrations" exceed or approach background concentrations and the Environmental Quality Standards (EQS) set to protect aquatic life. If the sum of release-concentration and background concentration is below EQS values, impairment will be negligible and the impacts are insignificant irrespective of duration and area extent of the concentration increase.

Concentration of heavy metals in the Baltic Sea has decreased steadily during the past 2-3 decades (Pohl and Hennings 2009) and recent (2006-2009) concentrations are low, also in the Fehmarnbelt (Table 7.1). Calculated release-concentrations are lower (central estimate) or comparable (max estimate; 99.9-percentile) to the background concentrations.

Table 7.1 Background concentrations of heavy metals in surface waters of Fehmarnbelt and the Baltic Sea (Gotland Deep) and, predicted concentration (central estimate and max estimate) in spill plume resulting from dredging in Fehmarnbelt. Values in brackets (for Cu and Pb) are estimated based on modelled release conc. from Cd and Ni; background conc. from Pohl and Hennings, 2009; Forsberg et al. 2006; Österlund et al. 2012

			Cd	**Cu**	**Hg**	**Ni**	**Pb**	**Zn**
			-----	-----	*µg/l*	-----	-----	-----
Back-ground	Femarnbelt	dissolved	0.01		0.001a)		0.0008	
		particulate	0.0002				0.06	
	Baltic Sea (Gotland Deep)	dissolved	0.009	0.5		0.6	0.006	1.3
		particulate	0.0002				0.004	
Spill	Central estimate	dissolved	0.0033	(<0.001)		0.036	(<0.001)	0.16
	Max estimate		0.029			0.55		2.55

a) total conc.

The calculated release concentrations of the three metals with the highest mobility are below corresponding environmental quality standards (Table 7.2, Figure 7.1). The EQSavg for cadmium set by EU (and adopted by Germany and Denmark) at 0.2 µg Cd/l is about 2 magnitudes higher than the central estimate of release concentration (0.0033 µg Cd/l; (Table 7.2 and 20 times higher than the background concentration (0.01 µg Cd/l, Table 7.1). The predicted max concentration at 0.029 µg Cd/l is about 20 times lower than the max acceptable concentration at 0.45 µg/l set by EU (also adopted by Denmark and Germany).

Danish authorities have set standards for nickel (VKK_{gen} 0.23-3.0 µg Ni/l), which are much lower than the EQS set by EU (20 µg Ni/l) (Figure 7.1, Table 7.2). Still, predicted release concentration of nickel in spill plume (0.036 µg Ni/l) is 6 times lower than the most restrictive Danish value (0.23 µg Ni/l), while the EU EQS is more than 500 times larger (Figure 7.1, Figure 7.2).

For zinc, the predicted median concentrations are 50 times below $VKK_{general}$ and the max predicted concentration (99.9 percentile – 2.55) is ca. 3.5 times below VKK_{max} (Table 7.2).

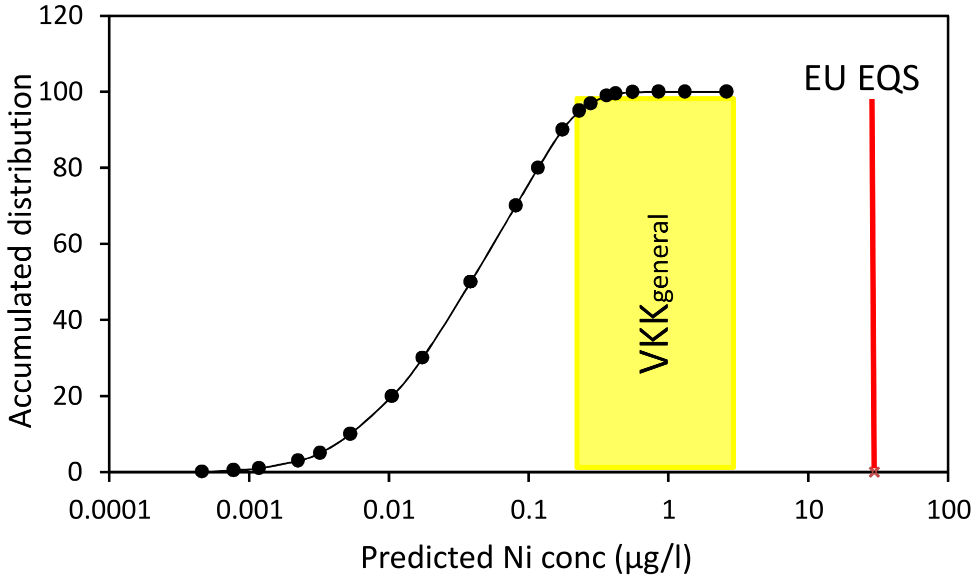


Figure 7.1 *Distribution function for predicted concentration of Ni down-stream dredger. Vertical red line denotes the EU/German EQS for Ni at 20 µg/l; yellow box shows the Danish provisional "general" EQS ranging 0.23-3.0 µg/l (added conc).*

Table 7.2 *Predicted distributions of increases in concentrations of Cd, Ni and Zn (µg/l) for dredging activities in the Fehmarnbelt. EQS (annual, allowable concentration) are environmental quality standards that protect aquatic life. Predicted median concentrations (50 percentile) are compared to $VKK_{general}$ and EQS_{annual}; and 99.9 percentile are compared to VKK_{max}.*

Percentile	**Cd**	**Ni**	**Zn**
50	*0.0033*	*0.036*	*0.16*
99	0.023	0.36	1.4
99.9	**0.029**	**0.55**	**2.55**
EQS (EU), ann avg.	*0.2*	*20*	
EQSmax (EU)	*0.45*		
"EQS", $VKK_{general}$ (DK)		*0.23-3*	*7.8*
"EQS",VKK_{max} (DK)		**6.8**	**8.4**

Persistent organic pollutants (POP's) in sediments have a much lower mobility than heavy metals and accordingly, the release during dredging and sediment spill will be very small. Assuming worst conditions the max concentration of benz(a)pyrene (magnitude of pressure) in spill plume was estimated to 0.0003 µg/l (Section 4.1). This max release concentration is ca 300 times lower than the max allowable EQS set by EU (adopted by Germany and Denmark), see Table 3.9. We do not know the release rate of other POP's from dredged sediments but they are presumably as low as benz(a)pyrene. Therefore, exceedance of EQS values for other POP's resulting from dredging operations is highly unlikely.

Impairment and Significance

Under both average conditions and under worct case conditions there is a large safety margin between predicted release concentrations of heavy metals and persistent organic pollutants in spill plume on the one hand side and, the most conservative environmental quality standards on the other side. After dispersion of the spill plume concentration will decrease further and "safety margin" increase. Therefore, release of toxic substances from dredged sediments is insignificant.

7.2 Suspended sediment

7.2.1 Magnitude of pressure

All magnitudes of pressure are described in Chapter 4.

7.2.2 Degree of impairment, severity and significance

By increasing the concentration of suspended matter in the water column substantially above the background level sediment spill from the Fehmarnbelt dredging can have adverse effects on water quality, on plankton populations, on productivity, and on bathing water quality.

The most important effects of sediment spill are mediated through reductions in light penetration into the water column thereby directly affecting transparency of water, i.e. quantified as Secchi depth. Reduction in Secchi depth affects the aesthetic value of beaches; reduce the primary production and gives rise to secondary effects such as lower oxygen production.

The impacts on water quality and bathing water quality related to reduction in light are predicted using numerical models while other impacts related to sediment spill are based on combining modelling, information from literature and expert judgements (see sections 3.2 and 3.3).

In the following sections that builds on numerical modelling baseline conditions are presented either as time series (Secchi depth at beaches) or as 2-D plots averaged over specific periods or over an entire year. Changes in water quality due to shading effects caused by sediment spill are presented as %-deviations (increase or decrease) from the baseline conditions. For oxygen where accepted numeric guidance values exists impacts are expressed as changes in mean concentrations and as changes in areas *times* days below 4, 2 and 1 mg O_2/l. If relevant, deviations from baseline conditions are shown for several years (October 2014-2019) during construction. Because suspended sediments potentially can have an impact on several (sub-) components it is convenient to evaluate severity of impairment and significance in common sections instead of treating them separately.

7.2.3 Secchi depth and bathing water

In baseline conditions Secchi depth varied spatially in the Fehmarnbelt from 7.0-7.5 m in the central part to 2.3-3.5m in the non-vegetated eastern part of the Rødsand Lagoon (Figure 7.9). Low Secchi depths in Rødsand are caused by regular resuspension of the relative fine sediment. In the western part of Rødsand Lagoon Secchi depth is larger because the dense population of *Zostera* supresses resuspension. Along the Lolland coast and around Fehmarn Secchi depth was higher than in the central part of Fehmarnbelt due to filtration by the large population of blue mussels. A high filtration pressure reduces phytoplankton and chlorophyll-a and, thereby reduces absorption of light.

As result of dredging in 2014-2015, the yearly average Secchi depth becomes reduced with 0 - 45% along the Lolland coast and in Rødsand Lagoon caused by sediment spill and spread and resuspension of sediments originating from dredging works in connection with work harbour and tunnel works. The largest reductions are expected at the Lolland coast near the alignment (estimated maximum at 45%; while maximum for Rødsand Lagoon is estimated at 40%). Reduction in light penetration affects both pelagic primary production (see below) and growth of macroalgae and eelgrass (see FEMA 2013a).

Reduction in Secchi depth in German waters is Minor (up to 15% reduction) and confined to coastal waters near Puttgarten (Figure 7.9).

Effect on Secchi depth of dredging works decreases in 2016 (max. reduction in Secchi depth ca. 32%) and are further reduced in 2017 when almost only Rødsand Lagoon is affected at levels above 10% reduction (Figure 7.3). In 2018 baseline conditions of Secchi depths will be fully restored (not shown).

In German waters Minor reductions in Secchi depth (up to 7%) are predicted west of Puttgarden in 2016 (Figure 7.3).

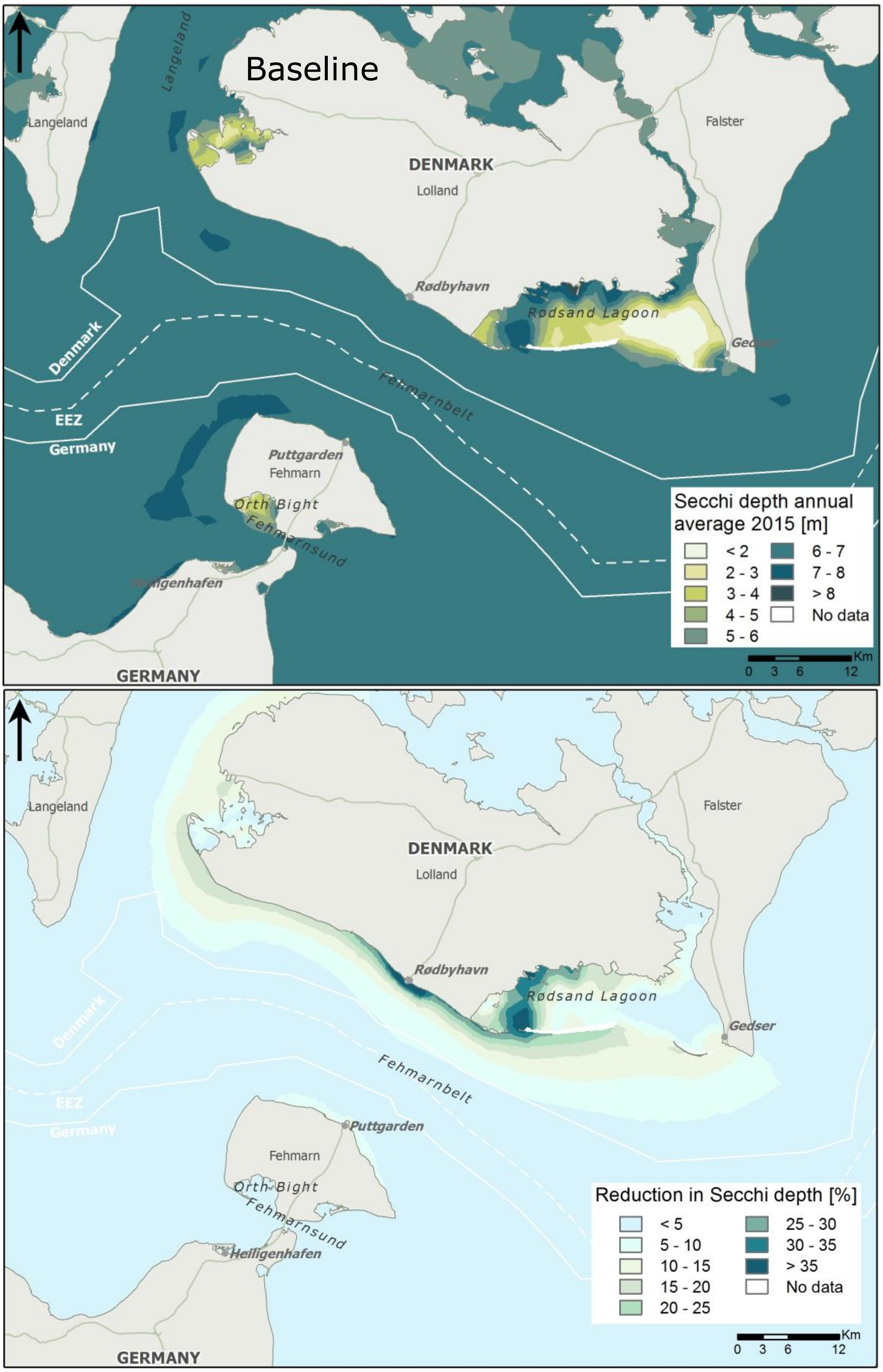


Figure 7.2 Modelled Secchi depth under baseline conditions (upper) and %-reduction in Secchi depth in 2014-2015, when tunnel dredging works will be most intense. Based on average Secchi depths from October 2014 to December 2015.

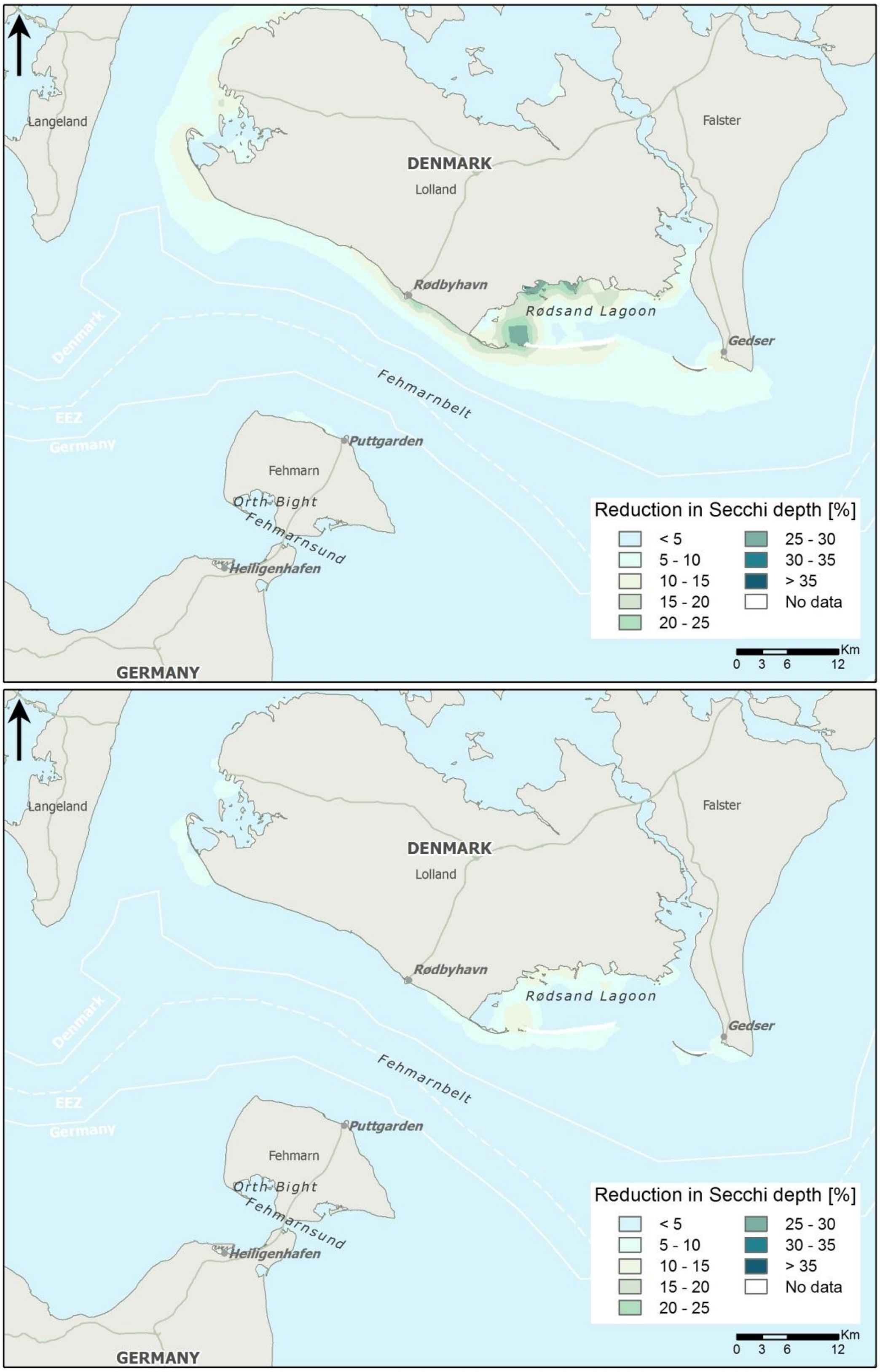


Figure 7.3 Modelled reduction (in percentage) in Secchi depth in 2016 (upper) and in 2017 (lower).

Impairment and Significance

The reductions in Secchi depth corresponds to Minor, Medium, and High degree of impairment in 2014-2015 and 2016 and Minor impairment in 2017 (Table 7.3) with Minor degree of impairment estimated for ca. 82% of the impacted area in 2014-2015. The total impacted area is reduced with more than 50% in 2016 with Minor degree of impairment estimated for ca. 89% of the impacted area. Except for 2014-2015 all impairments occur in Danish waters.

In German waters, reduction in the yearly averaged Secchi depths is confined to 2014-2015 and never exceeds Minor degree of impairment (Table 7.3).

Table 7.3 Degree of impairment on Secchi depth (areas in ha) caused by suspended sediments for the tunnel alternative.

	Total	**DK national + EEZ**	**DE national**	**DE EEZ**
2014-15				
Very high	0	0	0	0
High	1975	1975	0	0
Medium	5952	5952	0	0
Minor	35858	35745	113	0
Total	**43785**	**43672**	**113**	**0**
2016				
Very high	0	0	0	0
High	7	7	0	0
Medium	1926	1926	0	0
Minor	15874	15874	0	0
Total	**17807**	**17807**	**0**	**0**
2017				
Very high	0	0	0	0
High	0	0	0	0
Medium	0	0	0	0
Minor	2022	2022	0	0
Total	**2022**	**2022**	**0**	**0**

Reduction of water transparency along beaches
There are 16 designated beaches along the Fehmarnbelt coasts, 10 on Lolland and 6 on Fehmarn (Figure 7.4). The bathing water quality based on bacteriological status complies with the Bathing Water Directive for all beaches. In the previous Bathing Water Directive (BWD) (76/160/EEC) transparency of water was a mandatory parameter and previously also an integral part of the bathing water quality assessment allowing for no abnormal change in transparency. In the updated directive (2006/7/EC) transparency and other "supporting" physio-chemical parameters are not included. Therefore, for formal reasons reductions in water transparency at beaches along the Lolland coast and around Fehmarn do not violate the new BWD, but perception by bathers and local tourism probably will focus on such an aesthetic characteristic of beaches.

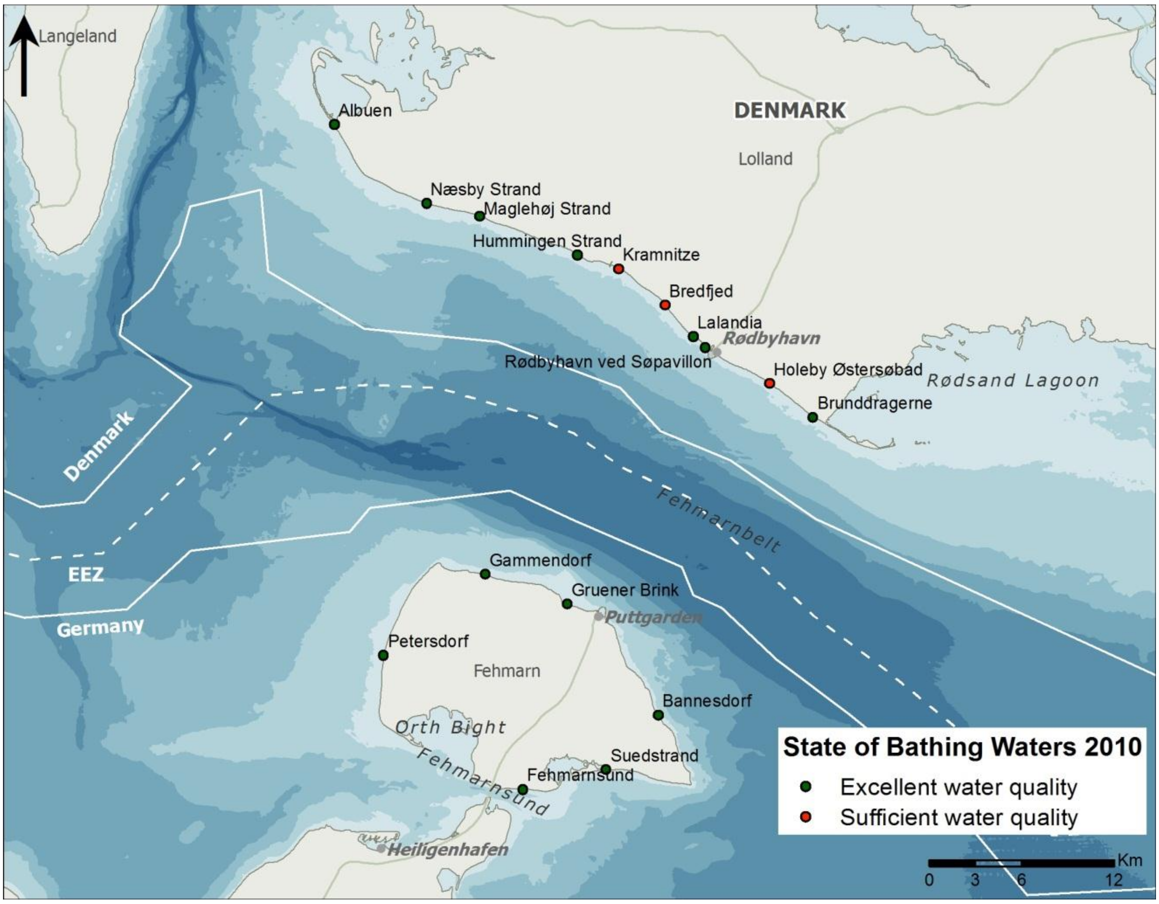


Figure 7.4 Location of 16 designated beaches with obligatory assessment of bathing water quality and, their compliance with the bathing water Directive. Blue indicates that the water quality is compliant with the guide values of the Directive or excellent water quality for 2010. Green indicates that the water quality is compliant with the mandatory values of the Directive or sufficient water quality for 2010 (Ref: WISE database (and to 3.2 http://www.eea.europa.eu/themes/water/interactive//batg)).

When construction work is initiated the two beaches near Rødbyhavn, "Lalandia" and "Rødbyhavn at Søpavillon" will be closed because of land reclamation (Figure 7.5). The remaining 14 beaches will be subject to potential impact from dredging works.

Figure 7.5 Sketch of planned land reclamation on Lolland.

Figure 7.6 shows two examples of temporal variation in Secchi depth for baseline and tunnel-scenario during 2015. A general feature is that reduction in Secchi depth increases towards the work-areas outside Rødbyhavn where sediments are spilled. Reductions in Secchi depths in 2015 varied between 16% and 48% at Lolland beaches, while predicted reductions at German beaches do not exceed 1% (Table 7.4).

Using the degree of impairment diagram (see Table 7.3) the predicted status of beaches can be estimated/designated for the different years under the tunnel construction period (Figure 7.6). One beach on Lolland (Bredfjed) will be minor affected due to reduction in Secchi depth during the bathing season 2015, when dredging works is most intense. At Bredfjed beach Secchi depth in bathing season will be reduced from an average of 9.0-9.1m to 4.7m. In 2016 and later, reduction of Secchi depth on Danish beaches varied between 0.6 and 2% and did not exceed 0.6% on German beaches.

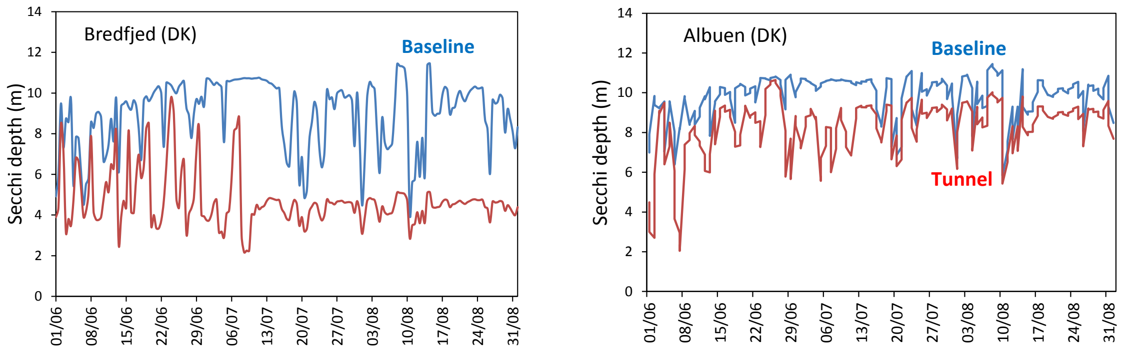


Figure 7.6 Temporal variation during bathing season (1 June- 31 August) in Secchi depth at Bredfjed and Albuen beaches on Lolland in 2015. Baseline and tunnel alternative shown.

Table 7.4 Predicted status of 14 beaches (based of water clarity) in 2015 and 2016. Values in brackets are reductions (%) in Secchi depth compared to baseline

Beaches	2015		2016
Albuen (DK)	negligible	(-16%)	negligible
Næsby Strand (DK)	negligible	(-17%)	negligible
Maglehøj Strand (DK)	negligible	(-20%)	negligible
Hummingen Strand (DK)	negligible	(-18%)	negligible
Kramnitze (DK)	negligible	(-27%)	negligible
Bredfjed (DK)	*minor*	(-48%)	negligible
Holeby (Hyldtofte) Østersøbad (DK)	negligible	(-16%)	negligible
Brunddragerne (DK)	negligible	(-16%)	negligible
Petersdorf (DE)	negligible	(-0.2%)	negligible
Gammendorf (DE)	negligible	(-0.4%)	negligible
Gruener Brink (DE)	negligible	(-0.9%)	negligible
Bannesdorf (DE)	negligible	(-0.7%)	negligible
Suedstrand (DE)	negligible	(-0.5%)	negligible
Fehmarnsund (DE)	negligible	(-0.2%)	negligible

Impairment and significance
One out of eight beaches along Lolland's southern coast that is expected to be active during dredging works will be minor impaired in 2015. Seen in isolation the impairment must be considered significant, but with access to a non-impaired beach within a distance of 4 km one could argue that for bathers the impairment will be non-significant.

7.2.4 Dissolved oxygen

Availability of dissolved oxygen at the sediment-water interface is important for the benthic fauna and for sediment processes such as remineralisation of nutrients. Under baseline condition concentration of oxygen in bottom water during summer and early autumn varied spatially from super-saturation (>8 mg O_2/l) in shallow areas with benthic vegetation (e.g. Rødsand Lagoon) to 2 mgO_2/l in the deep parts of Mecklenburg Bight (Figure 7.7).

Reduction in concentration of oxygen in bottom waters is driven by two mechanisms, which both can be directly related to dredging activity. Release of oxygen demanding substances (e.g. H_2S) will act locally depressing oxygen in water column close to the dredging site, while reduction in benthic oxygen production due to shading from suspended sediments has wider area extension and will be longer-lasting because spilled sediment will undergo regular sedimentation and resuspension. Release of oxygen demand is most critical below pycnocline, while reduction in oxygen production practically only occurs above pycnocline. Above pycnocline, release of oxygen demand at most can reduce bottom water oxygen concentration by ca. 2% (see section 4.4.3). Such reduction will work in concert with reductions in oxygen due to suppression of benthic primary production. In effect, estimated reductions in oxygen shown in Figure 7.7 in worst case should be reduced by an additional 2% locally around the alignment.

As and indirect result of dredging works in 2015 average concentration of bottom water oxygen will be reduced with up to 10% in Rødsand Lagoon. Outside Rødsand Lagoon and along Lolland coast, reductions in oxygen decrease with depth and also decrease westwards along the Lolland coast (Figure 7.7). The reduction is due to suppression of benthic primary production caused by shading from suspended sediments.

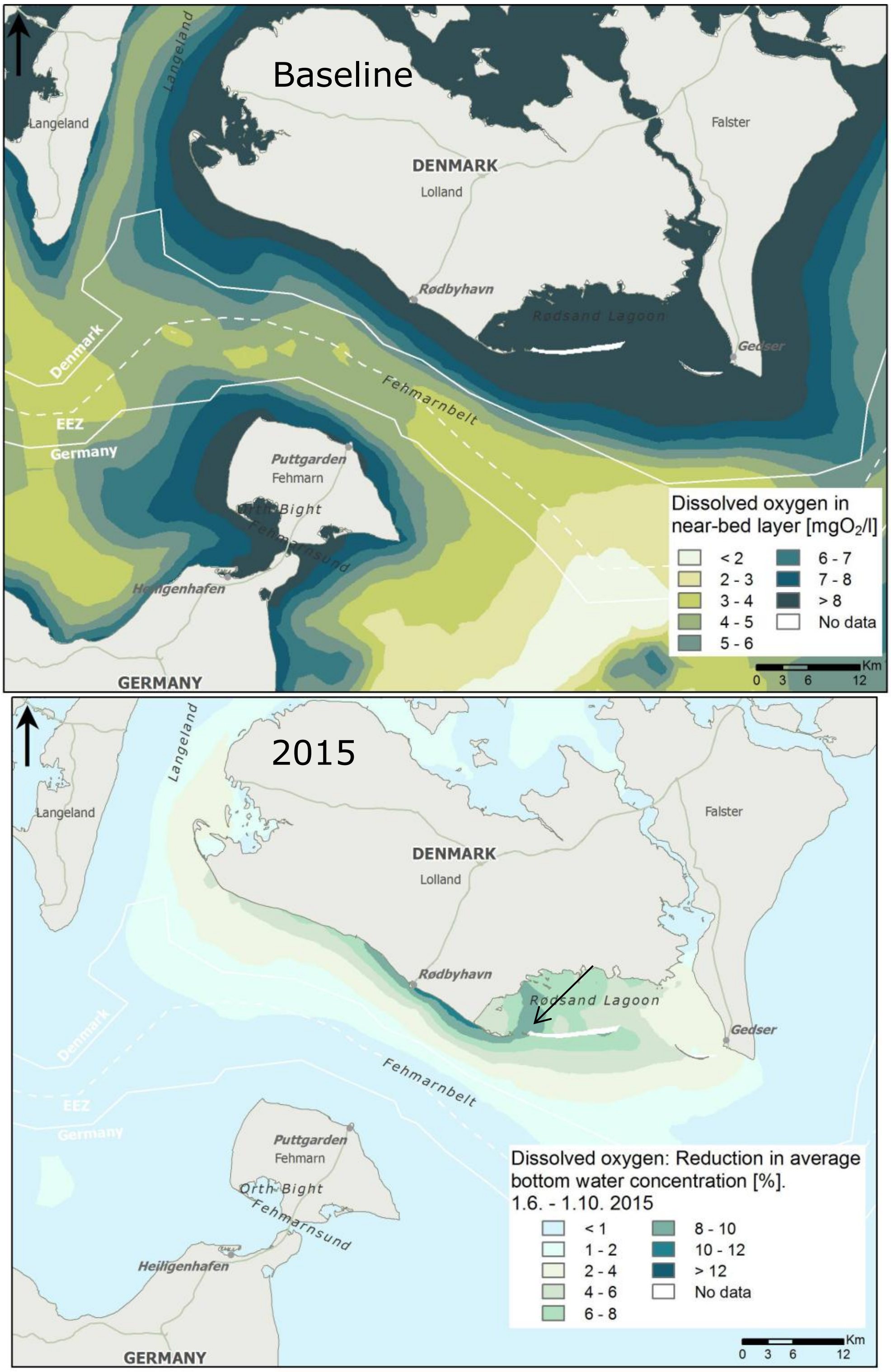


Figure 7.7 *Modelled concentration of dissolved oxygen in bottom under baseline conditions (upper) and %-reduction in oxygen in 2015, when dredging works will be most intense. Based on average oxygen concentration from 1 June to 1 October. Arrow shows position where time-series in Figure 7.8 of bottom water oxygen were extracted.*

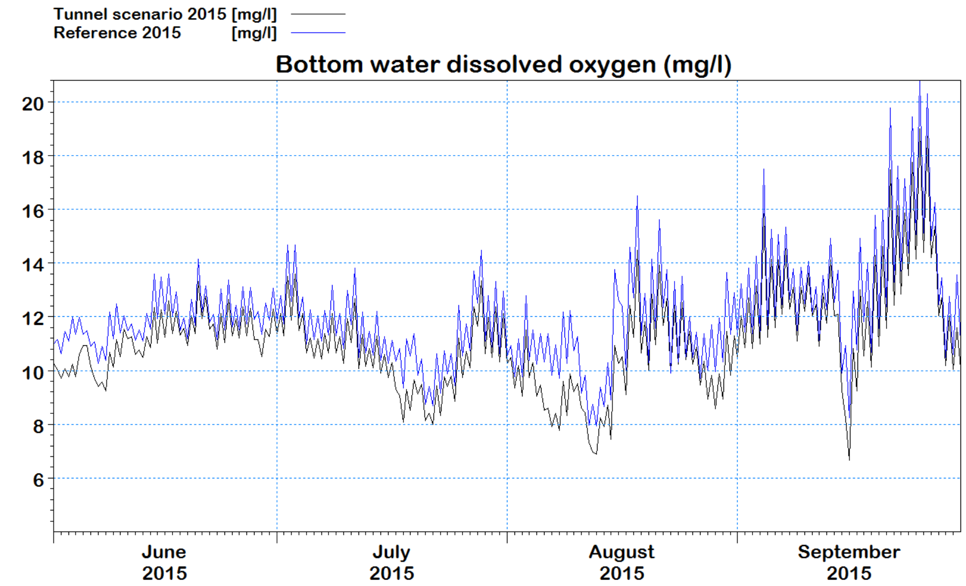


Figure 7.8 *Variation in modelled concentration of dissolved oxygen in bottom water under baseline conditions and in 2015 (tunnel alternative), when dredging works will be most intense.*

Impairment and Significance

In areas with the largest reduction in oxygen concentration such as in Rødsand Lagoon the concentration of oxygen do not fall below 6 mgO_2/l (Figure 7.8. Hence, using an incipient critical level of 5.7 mg O_2/l (see section 3.7) reduction in oxygen levels caused by dredging will not constitute an additional pressure on benthos and therefore, impairment of indirect oxygen reductions are considered to be insignificant.

Release of oxygen demand during dredging (direct effects of dredging) will be small and even under worst conditions this effect would not affect benthic organisms. Therefore, the combined effect of indirect and direct impacts will be very small and negligible.

7.2.5 Chlorophyll-a

Chlorophyll-a is an important light-harvesting pigment that occurs in all algae and therefore chlorophyll-a is a much used surrogate measure of plankton algal biomass. Along with nutrients and oxygen, concentration of chlorophyll-a is an important parameter to characterise the status of water quality.

In the FEMA model chlorophyll-a is a derived variable depending on phytoplankton biomass and the internal concentrations of nitrogen and phosphorus in phytoplankton. Chlorophyll-a therefore only loosely tracks phytoplankton biomass.

In the baseline conditions the concentration of chlorophyll-a in the main production period (March through November) varied in the Fehmarnbelt, from 1.6-2 µg chlorophyll-a/l in the central part, to less than 0.5 µg/l in the Rødsand Lagoon (Figure 7.9). Along the Lolland coast concentration of chlorophyll-al was slightly lower than in the central part of Fehmarnbelt due higher influence of Baltic Sea water (which has a lower chlorophyll-a concentration), but also due to filtration of the population of blue mussels. The highest concentrations were found southeast of Fehmarn caused by influence from the more nutrient-rich Mecklenburg Bight.

During 2015 concentration of chlorophyll-a will be reduced by up to 8-10% in Rødsand Lagoon, while reductions will be lower along the Lolland coast (max reduction 3-4%, Figure 7.9). Reductions in chlorophyll-a is much lower than impacts on the sedentary eelgrass and macroalgae (FEMA 2013a), because plankton is continuously replenished by advective transports mainly from the western Baltic Sea. The water with reduced plankton concentration is advected westwards with minor increases in chlorophyll-a towards the Great Belt. In 2016 and 2017 reductions in chlorophyll-a will gradually decrease (Figure 7.10) and baseline conditions are restored in 2018.

Impairment and significance
The modelled reductions correspond to a low and negligible degree of impairment, because reductions in waters of special importance for plankton (depths > 6m) are below 5% in all years during construction (Figure 7.9 and Figure 7.10). The impact is therefore considered to be insignificant.

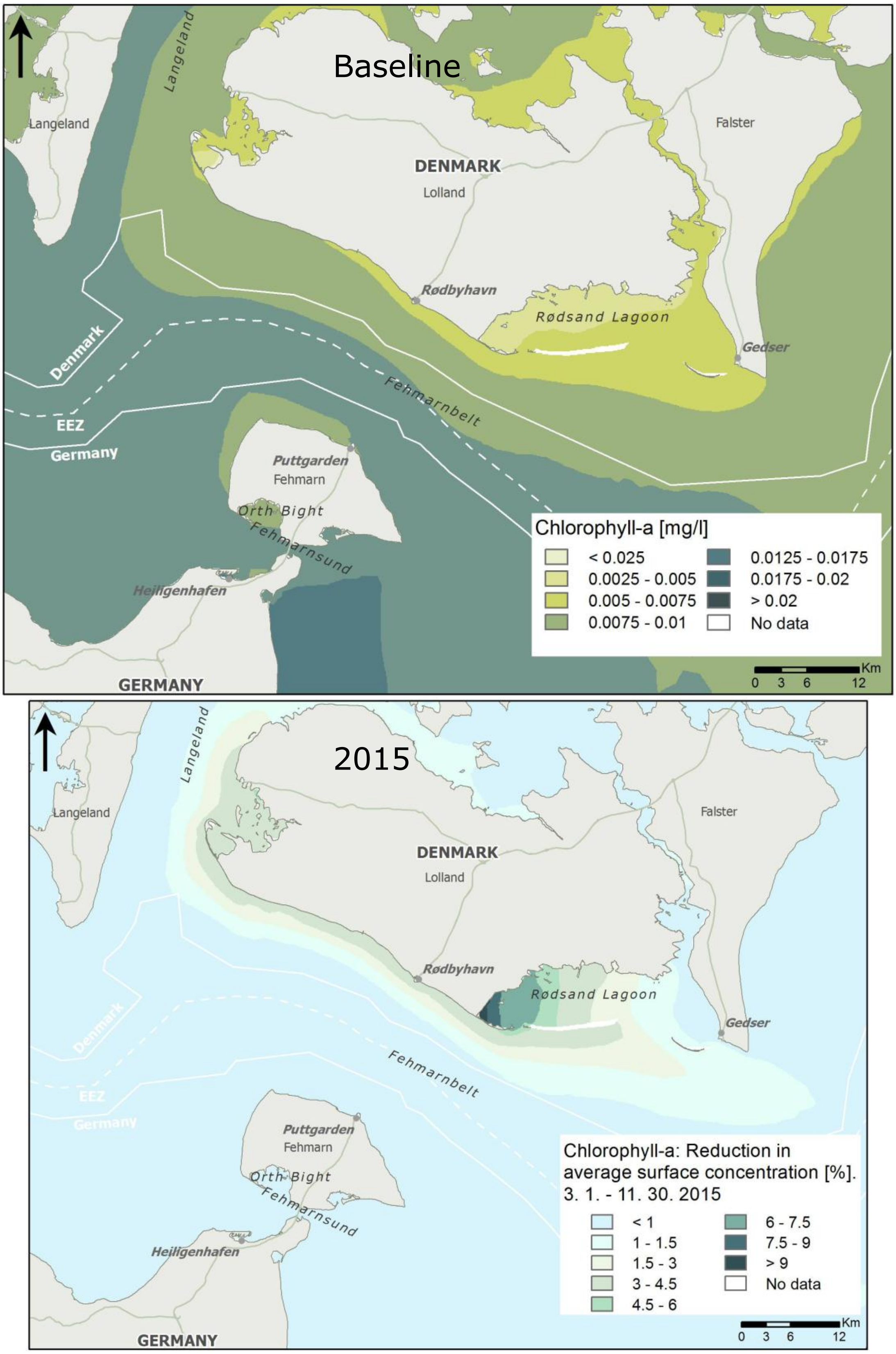


Figure 7.9 Modelled concentration of chlorophyll-a under baseline conditions (upper) and %-reduction in chlorophyll-a in 2015, when dredging works for tunnel will be most intense. Based on average concentrations 1 March-30 Nov.

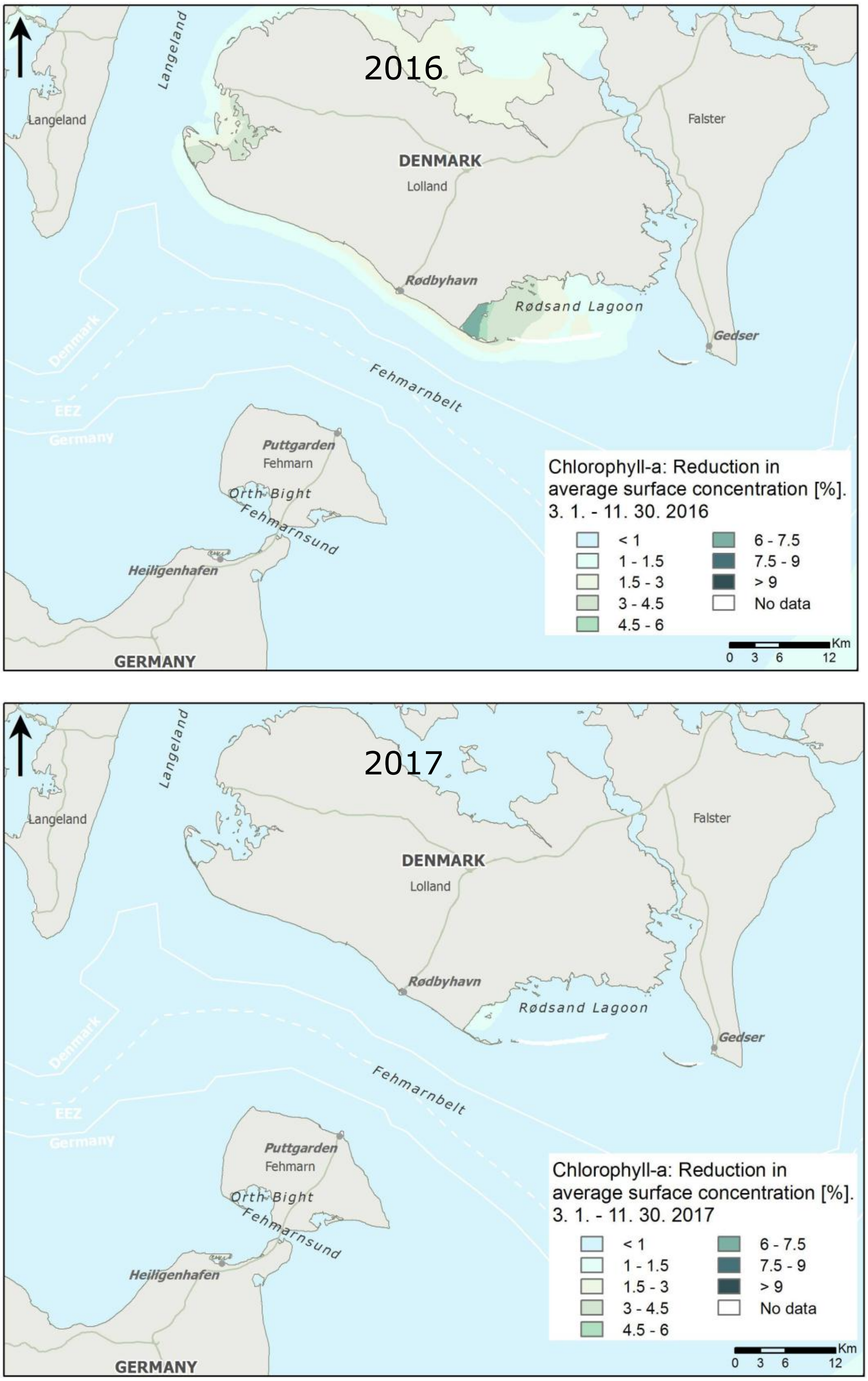
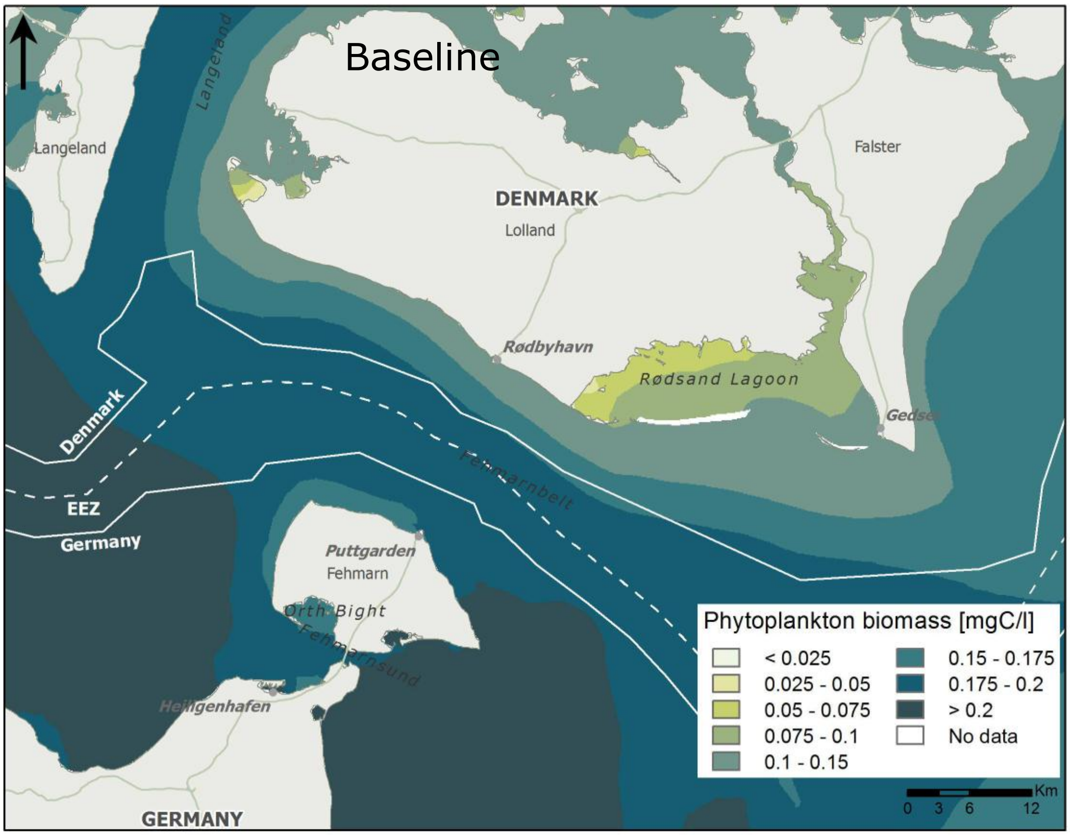


Figure 7.10 Modelled reduction (in percentage) in concentration of chlorophyll-a (compared to baseline) in 2016 (upper) and in 2017 (lower) of the tunnel scenario. Based on average 1 March-30 November.

7.2.6 *Phytoplanton biomass, production and nutrient dynamics*

Phytoplankton constitutes the base of the pelagic food web and any change in phytoplankton production and biomass can have cascading effects on higher trophic levels in the water column (zooplankton, planktivorous fish) or in benthos (filter-feeders at shallow waters, deposit feeders at larger depth).

As for chlorophyll-a the biomass of phytoplankton under baseline condition varied spatially with the lowest concentrations in Rødsand Lagoon and along the Lolland coast while the highest concentrations were modelled in Mecklenburg Bight and towards Kieler Bight (baseline; Figure 7.11). Dredging works carried out in 2015 will lead to reductions in phytoplankton biomass varying between 10% in the western part of Rødsand Lagoon, and 1-4% reductions along the Lolland coast. The largest %-wise reductions are modelled in Rødsand and along the Lolland coast where concentrations are low even under baseline conditions.



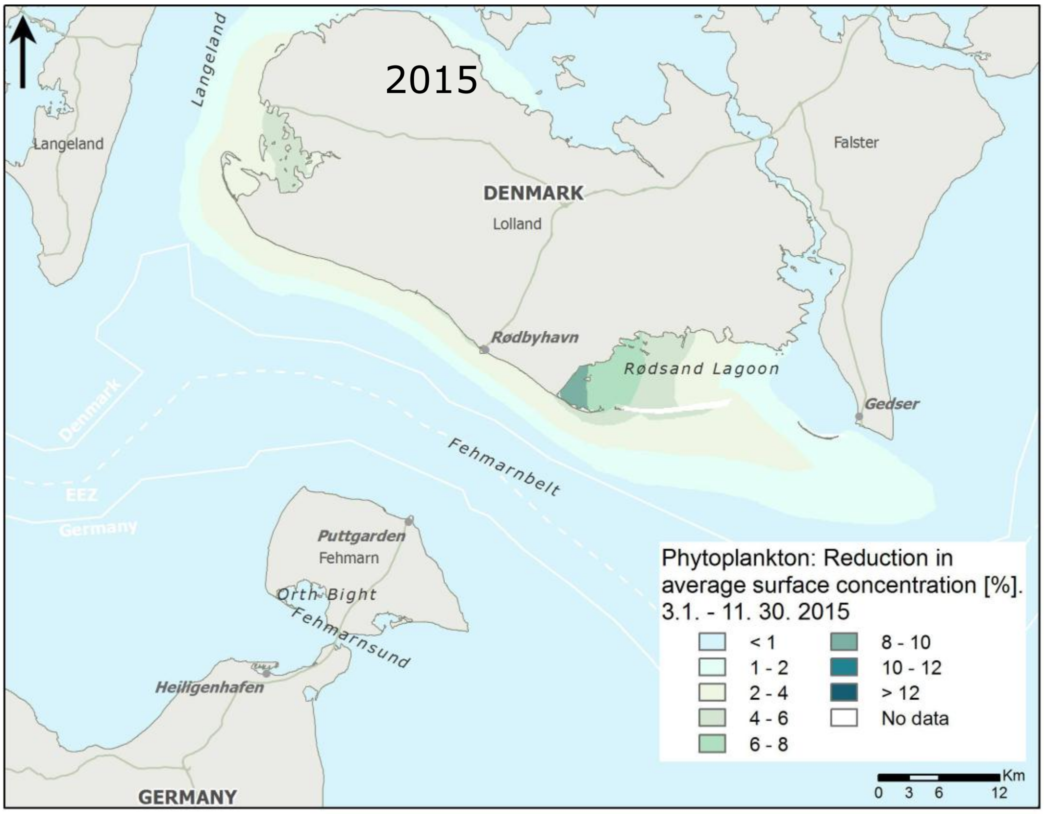
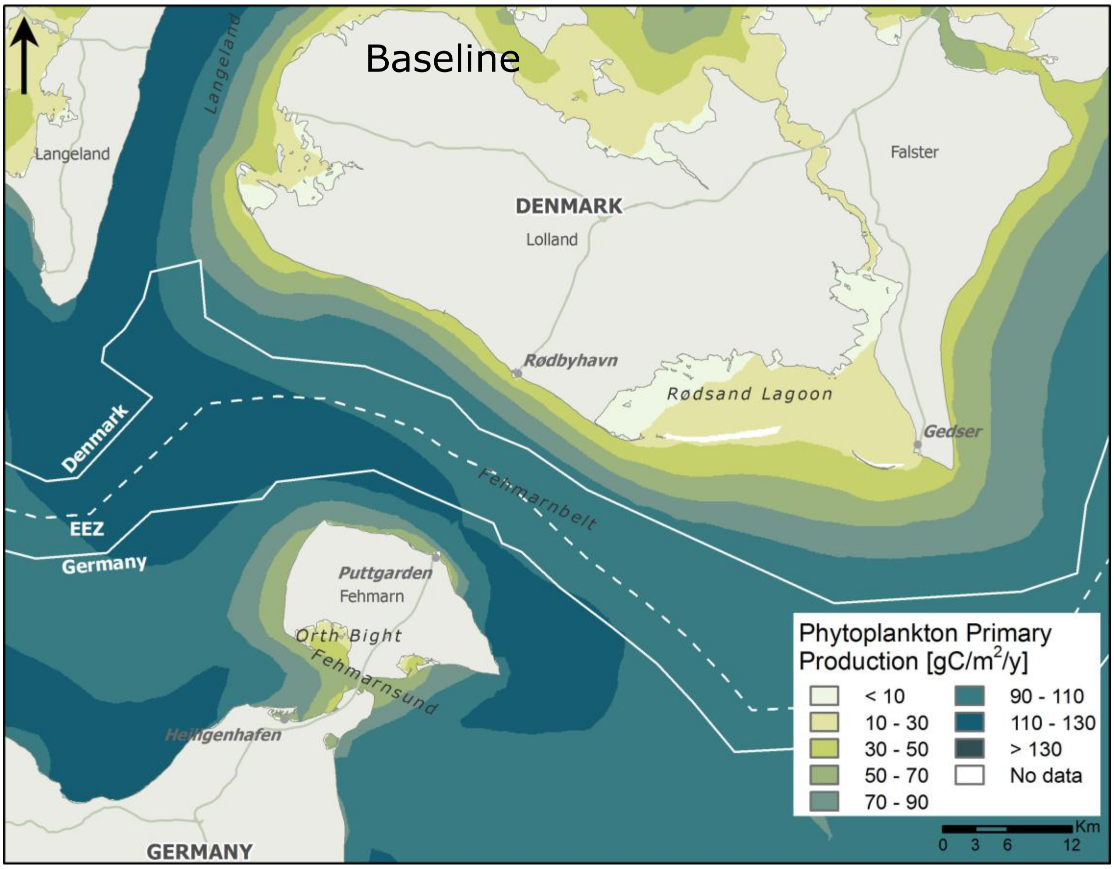


Figure 7.11 Modelled surface concentration of phytoplankton biomass under baseline conditions (upper) and %-reduction in biomass in 2015, when dredging works for tunnel will be most intense. Based on average concentrations 1 March-30 Nov.

Modelled pelagic primary production (PP) varied from 15 to 125 gC/m^2/y in the Fehmarnbelt under baseline condition, lowest in shallow areas dominated by benthic vegetation such as in Rødsand Lagoon, along Lolland and in Orth Bight and, highest in the Southern and Western part of the assessment area (Figure 7.12). The spatial variation is a result of depth (PP increases with depth until ca. 1% surface light ≈ 15 m), phytoplankton concentration (Figure 7.11) and nutrient availability (intrusion of nutrient-rich water from the Great Belt, coastal upwelling East of Fehmarn).

Dredging will lead to reduction in primary production caused by shading from spilled sediments. In 2015 dredging works are predicted to reduce primary production by up to 25% at the Lolland coast near alignment (Figure 7.12). The largest relative reductions occur where production is low. Integrated over the assessment area a reduction of 2-3% in primary production in 2015 can be expected and less the following years.



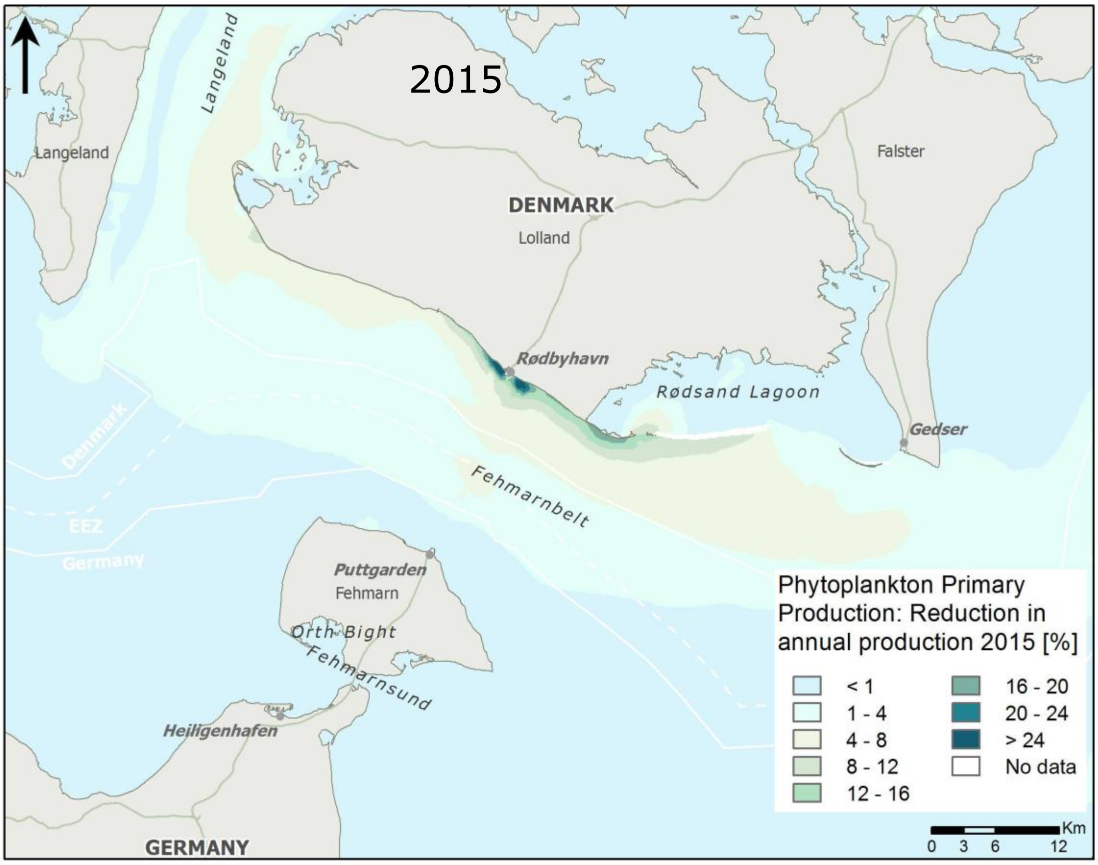
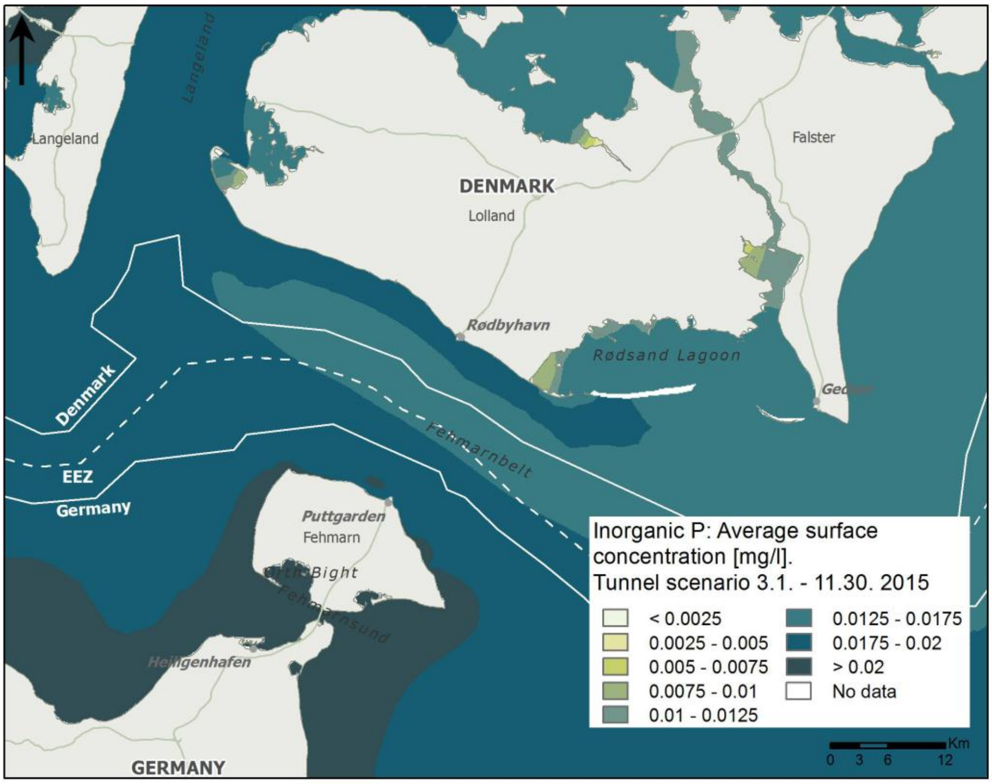


Figure 7.12 Modelled pelagic primary production (per year) under baseline conditions (upper) and %-reduction in primary production in 2015, when dredging works will be most intense.

Nutrient dynamics

Release of nutrients from dredged and spilled sediments will be very low and will not contribute to eutrophication (FEHY 2013d). Therefore, any changes in nutrient concentrations will be coupled to reduction in primary production and uptake in algae.

Based on the ratio between inorganic nitrogen and phosphorus, pelagic primary production is potentially strongly nitrogen limited implying that phosphate is in surplus (FEHY-FEMA 2013). In accordance, concentration of phosphate in surface waters is expected to increase during sediment spill because of lower growth in phytoplankton and benthic vegetation. The largest increase in phosphate is modelled in Rødsand Lagoon and along Lolland coast, in concert with presence of high concentrations of suspended solids and reductions in light availability (see Secchi depth above). The increase in phosphate concentration probably is a consequence of reduced primary production (lower uptake of phosphate in algae) and increased release from less-well oxygenated sediments.



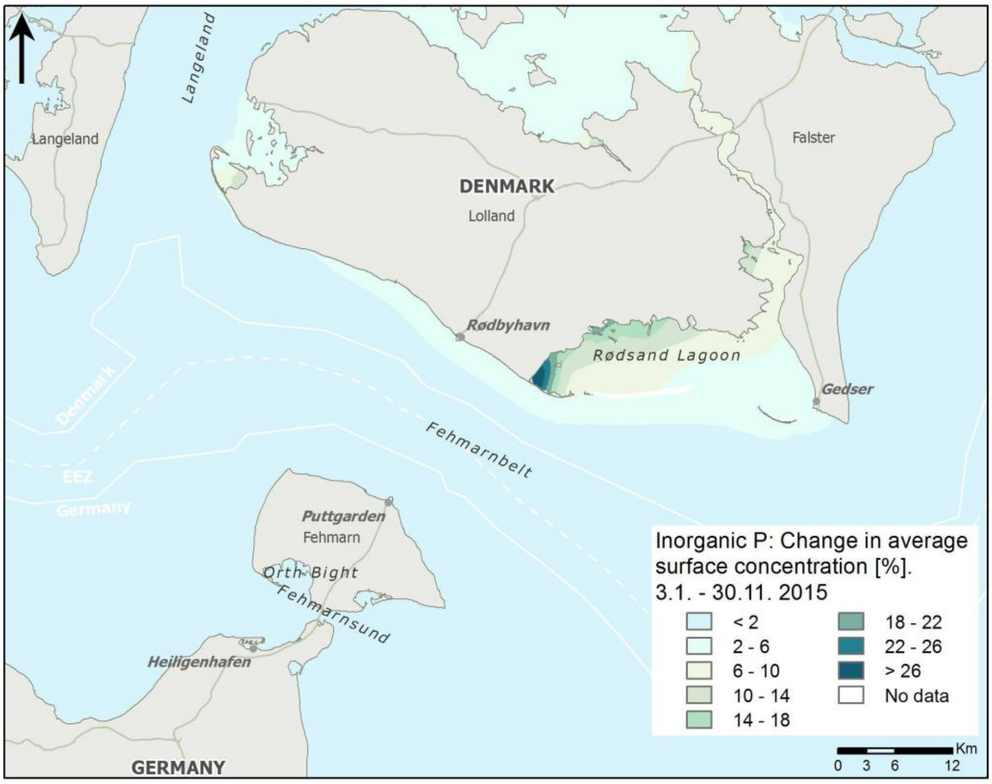


Figure 7.13 Modelled concentration of phosphate-P under baseline conditions (upper) and %-reduction in phosphate-P in 2015, when dredging works for tunnel will be most intense.

Increased sedimentation of phytoplankton

High concentration of suspended sediments (> 10 g/m^3) can lead to increased sedimentation of phytoplankton provided that phytoplankton cells are "sticky" (primarily diatoms) and they occur in high concentrations (> 300 mg/m^3). Such situations only occur during the spring bloom in the Fehmarnbelt.

The criteria for aggregation between phytoplankton and sediment spill and subsequent sedimentation are met along Lolland coast for a 7 day period in early April (Figure 7.14). Assuming that all phytoplankton biomass in these areas aggregate with suspended sediments and settle ca. 14 tons organic carbon will be taken out of the water column and the seabed will receive ca. 14 tons organic carbon in these areas (Table 7.5). Under baseline condition and no sediment spill 8.8 tons organic carbon will sediment in "aggregation" area, but summed over an entire year differences in sedimentation is very small. For the entire assessment area the difference in accumulated sedimentation is much below 0.01%.

Table 7.5 Predicted sedimentation of phytoplankton carbon in areas where aggregation between phytoplankton and suspended sediments are likely for tunnel scenario and baseline.

Tunnel scenario		**Baseline**	
30 March - 7 April	1 Jan – 31 Dec	30 March - 7 April	1 Jan – 31 Dec
13.7 t	411 t	8.8 t	406 t

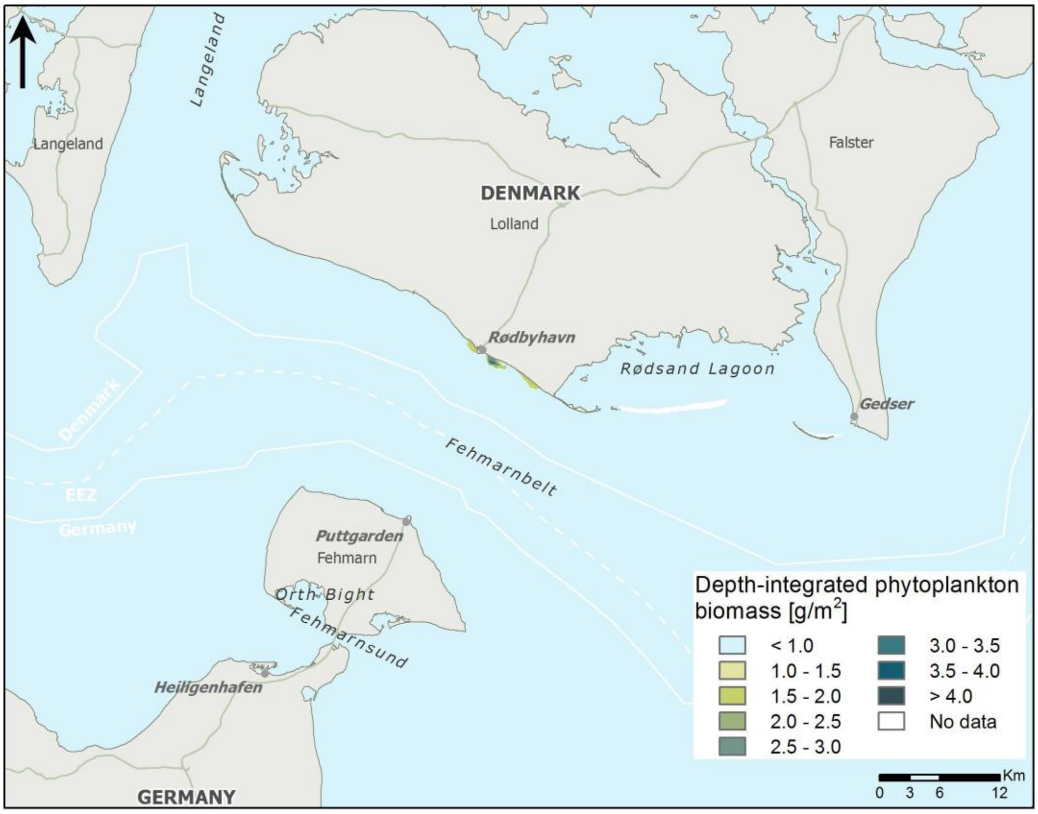


Figure 7.14 Depth-integrated phytoplankton biomass in areas where criteria for aggregation of phytoplankton and suspended sediment are met in the tunnel scenario.

Change in composition of phytoplankton

Light-induced changes in phytoplankton composition invariable will be very small because the average reduction in primary production (mediated by reduction in light availability) in the assessment area at the maximum is 1.2% (2015).

Impairment and significance
Chlorophyll-a, phytoplankton biomass and pelagic primary production are structurally and functionally tightly linked and in accordance, the spatial distribution and the relative reduction in concentration and rates caused by sediment spill show almost identical patterns. Therefore is it logical to apply a 5% reduction level as cut-off for impairment on phytoplankton biomass as applied for chlorophyll-a. The potential for additional sedimentation of phytoplankton are limited to very small areas and will only occur during spring bloom. Accumulated over a year the sedimentation may increase locally with up to 1-2%, but in the entire assessment area increase in sedimentation will not exceed 0.01%. Because reduction in primary production is very low (reflecting a very limited reduction in light availability) changes in phytoplankton composition is highly unlikely.

Inorganic nitrogen is by far the most limiting nutrient in the Fehmarnbelt (FEMA-FEHY 2013). Dredging will not lead to significant changes in concentration of nitrogen. Increases in phosphate concentrations of 2-4% along Lolland coast is equivalent to an absolute increase of 0.0005 mg/l. As phosphorus is in surplus in Fehmarnbelt any minor increase will not affect production and add to eutrophication effects. Therefore, impairment level is low and any effects will be insignificant.

Thus, direct and indirect impairments on nutrients, phytoplankton composition, biomass and primary production are considered as insignificant.

7.2.7 Zooplankton

Indirect effects
Production and biomass of zooplankton depend amongst other on availability of food (primarily phytoplankton). Thus, reduction in food concentration mediated through shading from suspended sediments can lead to reduction in growth and biomass in zooplankton, i.e. an indirect effect.

Baseline condition the biomass of zooplankton varies 10-fold within the model area, lowest in Rødsand Lagoon and highest west of Fehmarn (Figure 7.15). The indirect effect of suspended solids on zooplankton was very low in 2015 where sediment spill was highest, not exceeding 1% reduction in average biomass (Figure 7.15). Reductions larger than 0.1% were confined to Rødsand Lagoon, along Lolland coast and Hyllekrog.

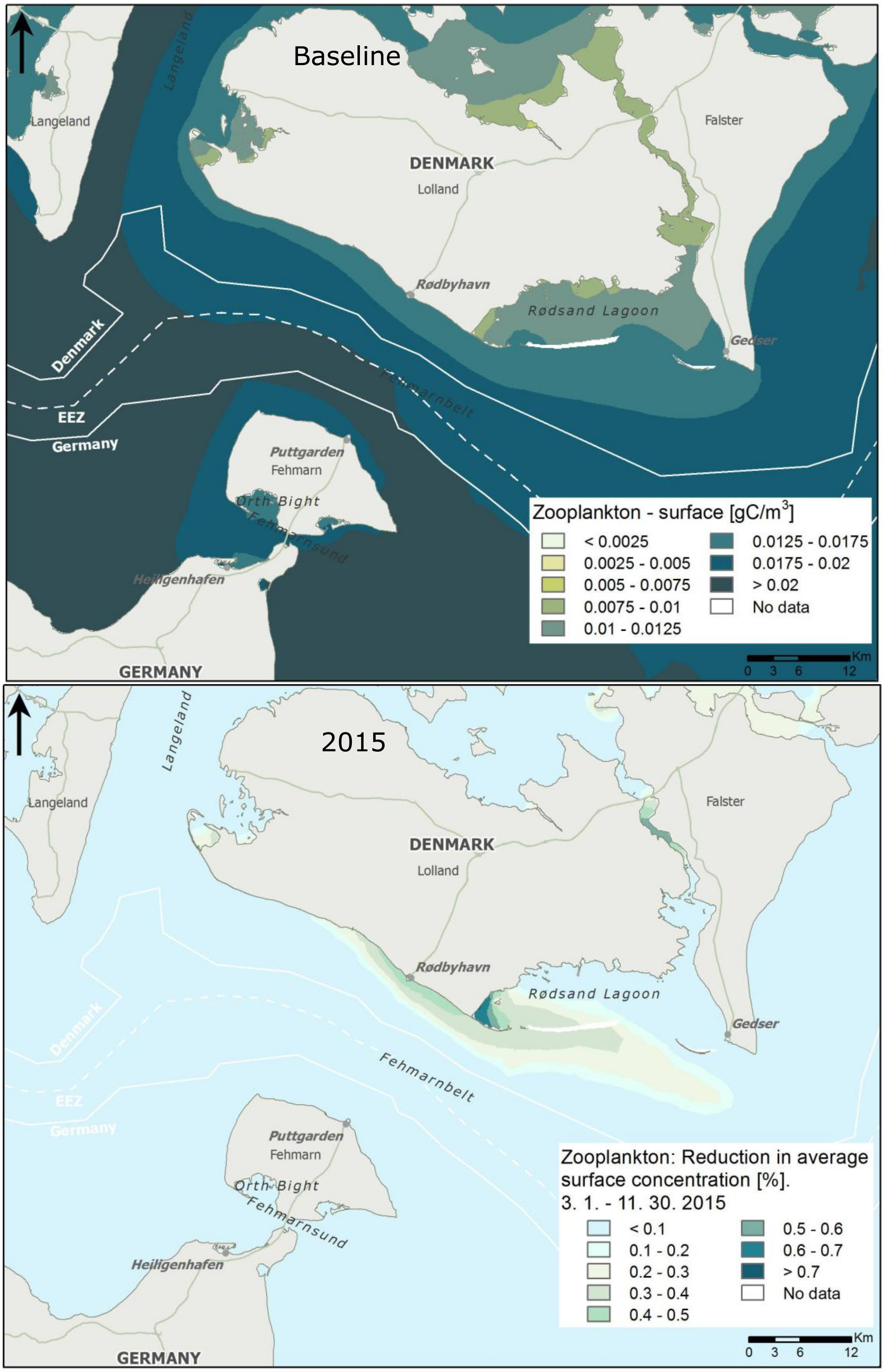


Figure 7.15 *Modelled biomass of zooplankton under baseline conditions (upper) and %-reduction in zooplankton in 2015 (lower panel), when dredging works for tunnel will be most intense.*

Direct effects
Reduction in physiological rates including behaviour, feeding activity and rate of egg production can be affected in some species (cladocerans) at suspended sediment concentrations above 10-20 mg/l, while copepods that dominate the biomass in Fehmarnbelt (FEHY-FEMA 2013) are much less sensitive (50-100 mg/l). Recognising that the water column concentration of suspended sediments in general is low (< 2 mg/l) in Fehmarnbelt, except in coastal waters along Lolland and in Rødsand (where concentration of zooplankton is low, see Figure 7.16) the direct impact of suspended solids on zooplankton will be very low.

Impact impairment and significance
Significance of impairment due to indirect effects on zooplankton of suspended sediments is insignificant, because reductions in biomass in all areas is below 1% of the baseline condition and, summed over entire model area reductions is below 0.1%. Analogously, direct impacts will be very low because concentration of suspended sediment is low in those areas where zooplankton biomass is high.

The impact is assessed to be insignificant.

7.3 Sedimentation

Sedimentation only affects zooplankton by burial of resting eggs.

Recruitment of zooplankton (especially copepods) can be impaired if resting eggs in sediments are covered with 20-40 mm sediment for extended periods. Resting eggs produced in late autumn that settle in the tunnel trench will likely not hatch because of burial under several cm of fine sand (Figure 4.4). Likewise, permanent burial of resting eggs will take place in the western part of Rødsand lagoon (Figure 4.5). The total area affected is 760,000 m^2 (tunnel trench: 160,000 m^2, Rødsand: 600,000 m^2). Compared to the total area of assessment (402,282 ha) the affected area is insignificant at 0.02%. In addition, when resting eggs are produced in autumn the biomass of zooplankton is very low in Rødsand, indicating a very low production of resting eggs (Figure 7.16). Also, given the large exchange with the adjacent areas minor "deficits" in recruitment will be compensated by imports from Great Belt and the Western Baltic Sea.

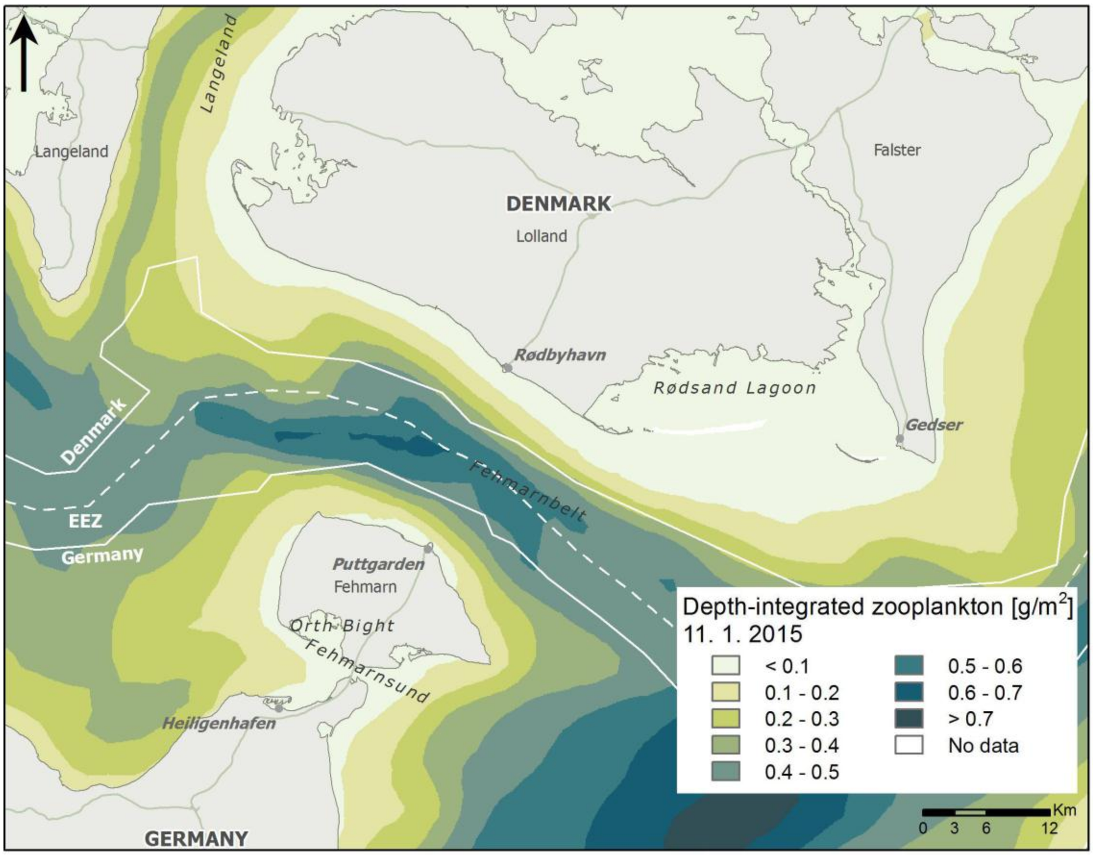


Figure 7.16 Modelled depth-integrated biomass of zooplankton on 1 November.

Impairment and significance
Impairment of recruitment to maintain the zooplankton communities in Fehmarnbelt by hatching of resting eggs will be insignificant, because of comparable small areas affected by sediment accumulation and, because production of resting eggs in part of these areas will be very small.

7.4 Permanent impacts including loss of habitats

Permanent impacts of the tunnel solution relate to loss of pelagic habitats for plankton (see section 4.5). The volume lost constitutes ca. 0.03% of total pelagic volume (0-20 m) in Fehmarnbelt and adjacent waters. Such low proportion, along with the fact that loss mainly is confined to waters where importance for plankton is "general" lead to conclude that impairment caused by permanent impacts is negligible.

7.5 Aggregation of impacts

The assessment results for the relevant pressures are compared and briefly discussed. The overall assessment are summarised in Table 7.6.

Table 7.6 *Overview of effects of the tunnel on water quality and plankton. Negligible = pressure very low or impairment negligible. Gray cells: not relevant.*

	-------- Temporary --------			---------- Permanent ---------		
Component/ sub-component	**Suspended sediment**	**Sedi-menta-tion**	**Toxic substan-ces**	**Solid sub-strate**	**Lost habitat**	**Hydrogra-phical re-gime**
Water quality:						
Secchi depth	Locations of minor to high degree of impair-ment					
Bathing water	Location of minor im-pairment					
Inorganic nutri-ents	Negligible					
Oxygen concen-trations	Negligible					
Plankton:						
Production, con-centration & composition	Negligible		Negligible		Negligible	
Plankton, facili-tated sedimenta-tion	Negligible					
Zooplankton	Negligible	Negligible			Negligible	

Most of the pressures have a negligible impact on water quality and on plankton. Secchi depth is affected by minor to high degree of impairment in the first year of the tunnel construction. The reduction in water transparency reduces the quality of the bathing water at one beach at the Lolland coast. It is assessed to be impaired to minor degree the first year of dredging.

7.6 Cumulative impacts

Cumulative impacts are not expected because comparable projects are not planned in the Fehmarnbelt area.

7.7 Climate change

The climate change up to year 2080-2100 has been evaluated at a workshop at the start of the Fehmarnbelt workshop. The outcome was the following main predictions:

- Air temperature will increase up to 4°C in the area

- The extreme wind speed (50 year return period) may increase by 3m/s or 10%. For more typical wind speeds there are no indications of significant changes

- The ocean water level may rise up to 1m, which will propagate into Fehmarnbelt and the Baltic Sea

The isolated impact of the tunnel in such a new climate setting is evaluated as being similar to the estimated impacts for the present climate setting.

7.8 Transboundary impacts

Transboundary impacts on water quality and plankton have not been identified for the tunnel alternative (see FEHY 2013c).

7.9 Mitigation and compensation measures

Due to the reduced Secchi-depth, bathing water will be impacted at one beach close to the alignment on Lolland. This impact may partly be mitigated by increasing bathers' awarenes of nearby beaches that are not affected, e.g. by dedicated signposting and improving the facilities, including parking etc.

7.10 Decommissioning

Decommissioning of the tunnel alternative is foreseen to take place in 2140, when the Fixed Link has been in operation for the design lifetime of 120 years.

The overall plan is that the main elements of the tunnel, relevant for the marine area will be decommissioned as follows:

The tunnel elements will remain below the seabed. The tunnel tubes are assumed filled with inert or mineral waste etc. and the tunnel tubes will be sealed. There will be no impact on the marine environment and thus no impact on water quality or plankton.

The reclamation areas will remain in place and will not be decommissioned. Therefore, no impact on the marine environment including on water quality or plankton is expected.

8 ASSESSMENT OF IMPACTS OF MAIN BRIDGE ALTERNATIVE

8.1 Toxic substances

The dredging intensity and sediment spill at level of a dredger will be comparable for both tunnel and bridge works. In effect, pressures, impairments and significance of toxic substances releted to the bridge alternative will be insignificant.

8.2 Suspended sediments and sedimentation during construction

Dredging works for the bridge is much less extensive than for the tunnel, approximately one tenth in terms of sediment spill. Impacts related to dredging works, i.e. extra suspended sediment and sedimentation are much lower than for the tunnel solution. In effect, impairments on factors and sub-factors will be very small in terms of degrees and area extension and accordingly, impairments are negligible. Examples of impact levels on Secchi depth, chlorophyll-a, near-bed oxygen, phytoplankton production and biomass, and zooplankton are shown in Appendix B.

8.3 Permanent hydrographic regime impacts

Detailed predictions of permanent change in hydrographic regime including temperature, salinity, current speed, wave climate were carried out using two different numerical models (MIKE 3 and GETM, see 3.9.2). In general, changes in water quality (Secchi dish, chlorophyll-a) and plankton sub-factors were small (FEHY 2013c) and insignificant with negligible impacts.

Two issues related to the bridge structure were noticeable; change in vertical mixing intensity resulting in a weakening of stratification east of the bridge and, a strengthening of stratification in Mecklenburg Bight (Figure 4.10, Figure 4.11).

The increase in vertical mixing has a positive effect on oxygen in bottom water, with local increases between 0.1-0.2 mg O_2/l covering an area of ca. 150-200 km^2 east of the alignment (Figure 8.1). Interestingly, bottom water in Mecklenburg Bight is not likely to be affected negatively by the stronger stratification, as bottom water oxygen is either unaffected or slightly increased. Therefore, the advection of oxygen-enriched bottom water east of the alignment more than counteracts the increase in stratification.

In spite of the modest increase in oxygen under average (model) conditions, the down-mixing of oxygen can provide an important supply to the benthic communities during periods with critical low oxygen levels as in 2010. Because the supply of oxygen is permanent the effect is considered being significant (positive).

Increased strength of stratification especially during summer will favour cyanobacteria compared to other phytoplankton groups (see section 5.5), but it is uncertain to what extent a small increase of 0.12-0.20 kg/m^3 in density difference between surface and bottom water (≈ 0.12-0.2 psu) will affect the risk for cyanobacteria blooms. Other factors may be equally important as indicated by the rather poor linkages between meteorological conditions during summer and concentration of cyanobacteria in the Fehmarnbelt area (see section 5.5).

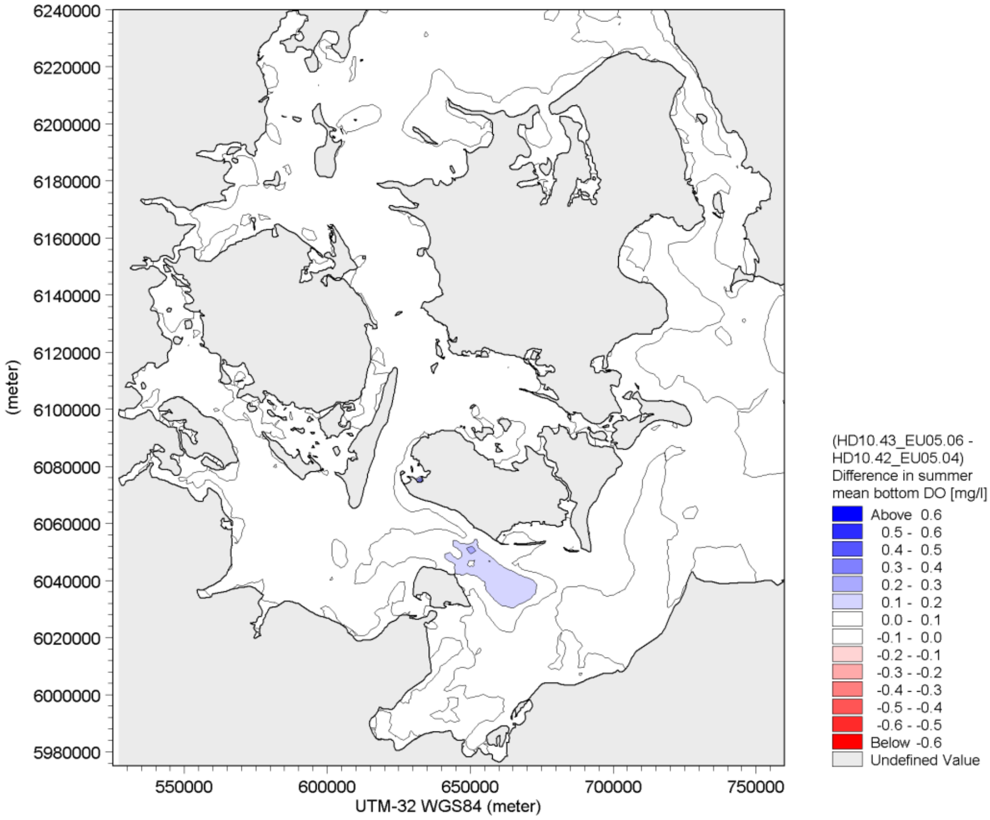


Figure 8.1 Predicted change in bottom water oxygen during summer using MIKE 3 local model for "bridge+ferry" case, from FEHY (2013b).

8.4 Additional solid substrate

Bridge piers, pylons and scour protection will increase the area of solid substrate and thereby favour populations of epibenthic invertebrates provided they are sub-state limited. Blue mussels will populate the solid substrate and filter phytoplankton advected between pylons and piers, but their impairment on the phytoplankton bi-omass was negligible (see 4.4.2).

Other issues of potential larger importance include effects on jellyfish.

8.4.1 Solid substrate promoting jellyfish in the western Baltic Sea

Polyps of *Aurelia aurita* will populate the additional hard substrate with a minimum abundance of 20,000 individuals per m^{-2} above the pycnocline and a 10 times lower abundance below pycnocline.

The additional area of solid substrate, i.e. bridge piers and pylons, from 3 m below MSL and to 20 m is 254,000 m^2 (25.4 ha) (FEMA 2013b). This area has to be com-pared to the existing area of solid substrate suitable for polyps. Besides stones and perennial macroalgae shells of blue mussels – living or dead - are such substrate. Blue mussels dominate the benthic biomass along Lolland coast and around Feh-marn in three communities: the *Mytilus* community, the *Bathyporeia* community and in the *Gammarus* community (FEMA 2013b) in an area totalling 1,200,000 ha.

Theoretically, polyps cannot establish on shells of young mussels (i.e. mussels with a shell length less than 35 mm) because young mussels continuously clean their shells with the foot (Theisen 1972). Hence, shells of mussels with shell length less than 35 mm cannot be considered as an available substrate for *Aurelia* polyps.

Based on measurements of 20 mussels (35-75 mm length) the outer (curved) shell area of a mussel with a shell length L was described by:

$$\text{Area (cm}^2\text{)} = 0.62 * L^2,\ r^2 = 0.93,$$

In calculating the area of available solid substrate it is assumed that only one of the two shells will be exposed and available to settlement. The total shell area representing living mussels were calculated based on size-abundance data from stations located within the three benthic communities dominated by blue mussels (FEMA 2013b).

Table 8.1 shows the calculated area of shell substrate within the depth range 3-20m where mussels occur.

Table 8.1 Calculated solid substrate composed of shells of blue mussels larger than 35 mm. Only one shell from each individual is included.

Depth range	**Shell area**	**Community area**	**Solid Substrate**
	m^2/m^2	**m^2**	**ha**
3-14 m	0.12	$120 *10^7$	14,400
14-20 m	0.005	$160 *10^7$	800
total 3-20m			15,200

The calculated area of solid substrate in terms of mussel shells at 15,200 ha is 600 times larger than the solid substrate of the bridge structures and accordingly, the additional recruitment of jellyfish caused by the bridge structures will be insignificant.

8.5 Aggregation of impacts

The assessment results for the relevant pressures are compared and briefly discussed. The overall assessment are summarised in Table 8.2.

Table 8.2 Overview of effects of the bridge on water quality and plankton. Negligible = pressure very low or impairment negligible. Gray cells: not relevant. (s+): significant change (positive).

Component/ sub-component	Suspend-ed sedi-ment	Sedi-menta-tion	Toxic subst.	Solid sub-strate	Lost habitat	Hydro-graphical regime
	-------- Temporary --------			---------- Permanent ---------		
Water quality:						
Secchi depth	Negligible					
Bathing water	Negligible					
Nutrients	Negligible					
Oxygen	Negligible			Negligible		Minor (s+)
Plankton:						
Production, con-centration and and composition	Negligible		Negligible	Negligible	Negligible	Minor - negligible
Facilitated sedi-mentation	Negligible					
Zooplankton	Negligible	Negligible	Negligible		Negligible	
Jellyfish recruit-ment				Negligible		

All pressures except two have a negligible impact on water quality and on plankton. Bridge structures increase vertical mixing in the Fehmarnbelt leading to an increase in bottom water oxygen concentrations (positive effect). Also related to bridge structures, strength of stratification is slightly increased in Mecklenburg Bight with possible increase in risk of cyanobacteria blooms.

8.6 Cumulative impacts

There are no cumulative impacts for water quality and plankton.

8.7 Climate change

The climate change up to year 2080-2100 has been evaluated at a workshop at the start of the Fehmarnbelt workshop. The outcome was the following main predictions:

- Air temperature will increase up to 4°C in the area
- The extreme wind speed (50 year return period) may increase by 3m/s or 10%. For more typical wind speeds there are no indications of significant changes
- The ocean water level may rise up to 1m, which will propagate into Fehmarnbelt and the Baltic Sea

An increase in temperature will favour heterotrophic processes (such as filtration rates in mussels) compared to autotrophic processes (i.e. phytoplankton growth)

(Dippner et al. 2008). In effects, filtration pressure on phytoplankton from mussels on piers and pylons probably will increase by 5-10%. Such small increase will not increase this impairment above negligible. For other impacts related to the bridge they are not expected to change in a future climate setting.

8.8 Transboundary impacts

Transboundary impacts on water quality and plankton was examined using the FEHY regional model (FEHY 2013b). Changes in water quality and plankton biomass in the Baltic Sea were very small and, when compared to the natural year-to-year variation impairments were negligible.

8.9 Mitigation and compensation measures

Compensation is a legal requirement, if protected habitats/species are lost or impaired significantly. Water quality and plankton is not significantly impaired due to building of the bridge. Thus, compensation and mitigation of water quality and plankton is not necessary for the bridge alternative.

8.10 Decommissioning

Decommissioning of the bridge alternative is foreseen to take place in 2140, when the Fixed Link has been in operation for the design lifetime of 120 years.

There is an overall plan for all main elements of the bridge. At sea all parts of the construction will be removed, leaving only the pile inclusions, which are located under the seabed. This section describes the decommissioning which is in relation to the marine area and hence the water quality and plankton.

Dismantling of the bridge superstructure will happen at sea and structures will then be transported to the shore.

All elements of pylons and piers will be cut in-situ into manageable sizes and then transported to shore.

The pylon caissons will be removed by in-situ demolition of the plinth, de-ballasting and de-floating of the caissons. Caissons are transported to shore. The pier caissons are removed by removal of internal ballast material, removal of scour protection and backfill material around the caissons and then transported to shore.

Ship collision structures are removed by a reversed construction.

During the decommissioning of the bridge, water quality and plankton are not expected to be impaired.

The permanent effects described in this EIA on water quality like reduced oxygen ad increased strength of stratification in the Mecklenburg Bight will disappear.

Reversed construction, in-situ demolition and cutting procedures can have a minor near zone effect, but the impact is regarded as negligible.

9 COMPARISON OF BRIDGE AND TUNNEL ALTERNATIVES

The assessment results are compared and briefly discussed. The alternatives are classified pressure-specific in terms of preferable alternative (Table 9.1). Three classes are used for the classification:

- ++ preferred alternative (the alternatives differ in terms of significance)
- + slightly preferred alternative (the alternatives differ respectively in terms of overall affected area and severity levels)
- 0 no difference between alternatives

Suspended sediment
Suspended sediment causes a reduction in Secchi depth during construction of the <u>tunnel</u>. The impact is mainly restricted to the first three years of the construction period. This impairment also impacts bathing water at one beach in the first year of the tunnel construction.

For all other components within water quality and plankton the impacts are negligible.

The pressure has no or negligible impacts on water quality and plankton for the <u>bridge</u> alternative, thus the bridge alternative is the preferred solution considering suspended sediment.

Sedimentation
The pressure sedimentation has no or negligible impacts on water quality and plankton for the tunnel and the bridge alternatives.

Toxic substances
The pressure has no or negligible impacts on water quality and plankton for the tunnel and the bridge alternatives.

Solid substrate
The pressure has no or negligible impacts on water quality and plankton for the tunnel and the bridge alternatives.

Lost habitats
Pelagic habitat will be lost in either link solution, but volumes involved are so small that impairment is negligible

Hydrographical regime
The structures of the <u>bridge</u> lead to seasonal (summer) increases in bottom water oxygen in the Fehmarnbelt because of increased vertical mixing, and to slightly stronger stratification of the water column during summer in Mecklenburg Bight. The latter will slightly increase local risks for cyanobacteria blooms. This impairment cannot be graded but is small. Even though the bridge structures introduces a change in oxygen, the change is regarded as positive for environment and are evaluated to override "negative" environmental effects due to increased strength of stratification.

Table 9.1 Results for the comparison between alternatives (++ preferred, + slightly preferred, 0 no difference).

	Tunnel	Bridge
Suspended sediments		++
Sedimentation	0	0
Toxic substances	0	0
Solid substrate	0	0
Lost habitats	0	0
Hydrographical regime	0	+
Total	0	++

For water quality and plankton the bridge is the preferred solution.

10 PROTECTED SPECIES

There are no protected plankton species.

11 CONSEQUENCES TO IMPLEMENTATION OF WFD AND MSFD

Dredging works and especially for the tunnel alternative will lead to significant reductions in Secchi depth that is a water quality component to support assessment of biological components such as phytoplankton and benthic vegetation. The reduction in Secchi depth will gradually diminish during the construction period and return to baseline condition in 2018.

12 KNOWLEDGE GAPS

During the impact assessment several knowledge gabs were identified resulting in uncertain or impossible assessment of a pressure.

They are in random order:

12.1 Intestitinal bacteria in dredged sediments

The Bathing Water Directive sets standards for maximum concentration of *E. coli* bacteria and enterococci in bathing waters, while sediments as source for indicator bacteria are not considered. A recent review showed that faecal bacteria are nearly ubiquitous in beach sands (Halliday and Gast 2011). Results from previous dredge works are meagre, but *E. coli* has been demonstrated in dredged harbour material in several EIA's from UK. Without knowing the content of intestitinal bacteria in Fehmarnbelt sediments, the risk of introducing intestinal bacteria (not necessarily from humans) into the water column along with suspended solids that can spread to beaches, cannot be quantified.

12.2 Suspended sediments

When sediment spill was modelled the effect of the filtering effect of the filter-feeding bivalves (primarily blue mussels) in shallow waters was not included (impossible for computational reasons). Theoretically, the mussel population has a capacity to filter the entire sediment spill produced during the construction period. To what extend this will happen is not known with precision, but the mussel population will remove a significant part of the suspended solids. If ingested and defaecated small particles (< 50 µm) will be turned into much larger particles (mm - scale) with a 10-20 times larger critical sheer stress for resuspension. Therefore, sediment spill is not "removed" but aggregated into larger particles that do not resuspend.

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A P P E N D I C E S

A P P E N D I X   A

FEMA ecological model, short description and calibration

APPENDIX A: FEMA ECOLOGICAL MODEL, SHORT DESCRIPTION AND CALIBRATION

The ecological model for the flora and fauna impact assessment is based on a three-dimensional ecological model, which has been refined, set up, calibrated and validated in connection with the Fehmarnbelt baseline study on hydrography and marine biology.

The eutrophication model describes the relationships and interaction between nutrients and primary producers, as well as the interrelationship and inter-specific competition between three distinct groups of producers; pelagic phytoplankton, benthic macroalgae (three different groups) and rooted vegetation (eelgrass).

Effect of light on growth rate is described by saturation functions, where light requirements differ between the three defined groups. Besides light, nutrient requirements, source of nutrients (water, sediment and water) and substrate are the main factors that differentiate the groups in the model.

Modelling of sediment spill impacts
Sediment spill has been modelled separately; see (FEHY 2012a).The coupling between sediment spill and light reduction is explicitly handled by the FEMA eutrophication model. Four different fractions of suspended material, with different optical properties determined from laboratory experiments, were transferred dynamically to the three-dimensional eutrophication model during the actual model execution (see Appendix D). The imported spill data were used as the basis for the calculation of the impact on light penetration through the water column, resulting in changes in the primary production in the model - in each grid point, model layer and time step – for specific prediction of the time-integrated effect on benthic communities from the dredging works.

Sensitivity testing was an important process in model establishment that allowed for the identification of specific model characteristics relevant for the modelling of dredging impacts.

Two sensitivity tests were carried out based on the final ecological model. The sensitivity tests were carried out for the following adjustments in the model set-up and its key parameters:

1. Sensitivity testing of the model's response to sediment spill based on low, medium and high suspended sediment concentration pressures (impacts during dredging works)

2. Sensitivity testing of the model's ability to predict biological recovery following the completion of suspended sediment concentration pressures (impacts after completion of the dredging work)

The conducted sensitivity tests for model responses to suspended sediment concentration pressures, and for the biological recovery after completion of suspended sediment concentration pressures, were found to be in line with what is intuitively expected, both with respect to differences between the three biological components (macroalgae, eelgrass and blue mussel), differences found for increasing suspended sediment concentration pressures, and differences identified at different water depths (see FEMA 2012b).

Calibration of FEMA models for water quality and ecology

Water quality and ecosystem models are simplifications of the real nature, but all modellers strive to reduce the gap between the simulated and the observed WQ values to zero. However, a complete reduction is impossible owing to the uncertainties inherent in any modelling procedure. Also models shall perform equally well for all areas in the model domain which invariable include some compromise in performance.

The FEMA model was calibrated against water quality data (inorganic nutrients, chlorophyll-a, dissolved oxygen in bottom waters) collected at several stations within the model area (Fig A.1). Unfortunately, monitoring data was not available from the Fehmarnbelt with sufficient frequency for the calibration year (2005). Therefore, data from Great Belt, Southern Kattegat, Øresund and area South of Fuen provided the main bulk of data for calibration.

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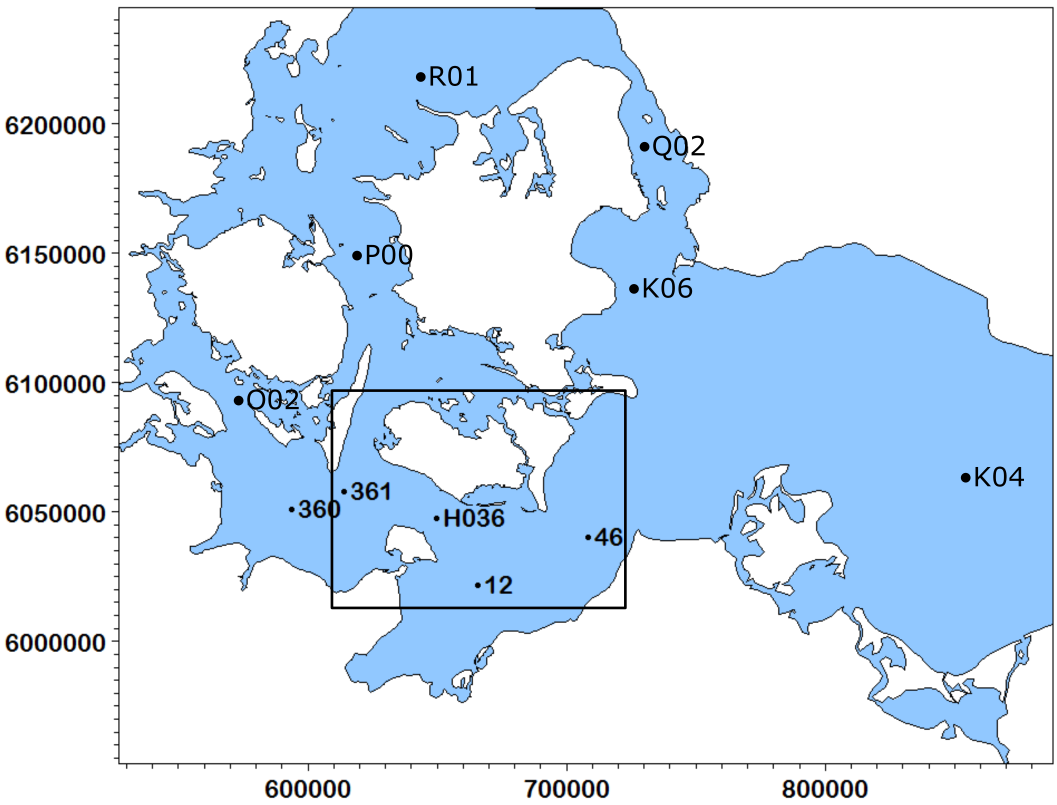


Figure A.1 Modelled area for the local FEMA ecological model; Station data from K04, K06, Q02, R01, P00, Q02 were used for calibration (2005) and, station data from360, 361, H036, 12 and 46 were used for validation (2009-2010 Baseline).

Figures A.2 and A.3 show examples of the model calibration from the northern Great Belt station P00). Overall, the model is able to simulate the inorganic nutrients through seasons, the model generally track the seasonal variation in chlorophyll, but with the exception of a 2½ weeks' delay of the spring bloom. The simulated levels of chlorophyll are satisfactory except that the model underestimate chlorophyll in January and February. The model tracks the autumn reduction in bottom water oxygen, but do simulate the summer concentration to a lower level than measured.

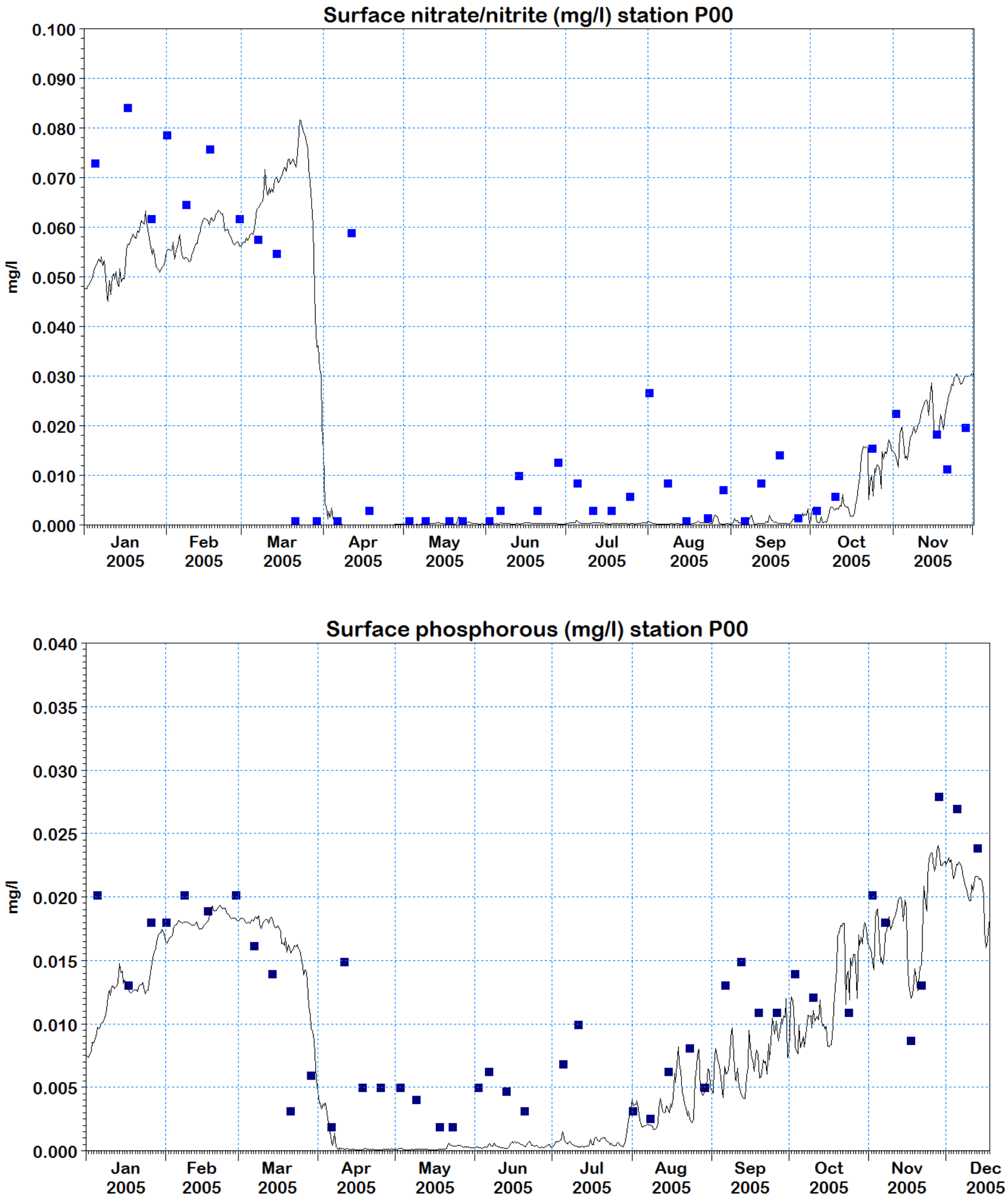


Figure A. 2 Modelled (line) and measured (squares) concentration of nitrite + nitrate (upper panel) and phosphate (lower panel) in surface water in Great Belt in 2005.

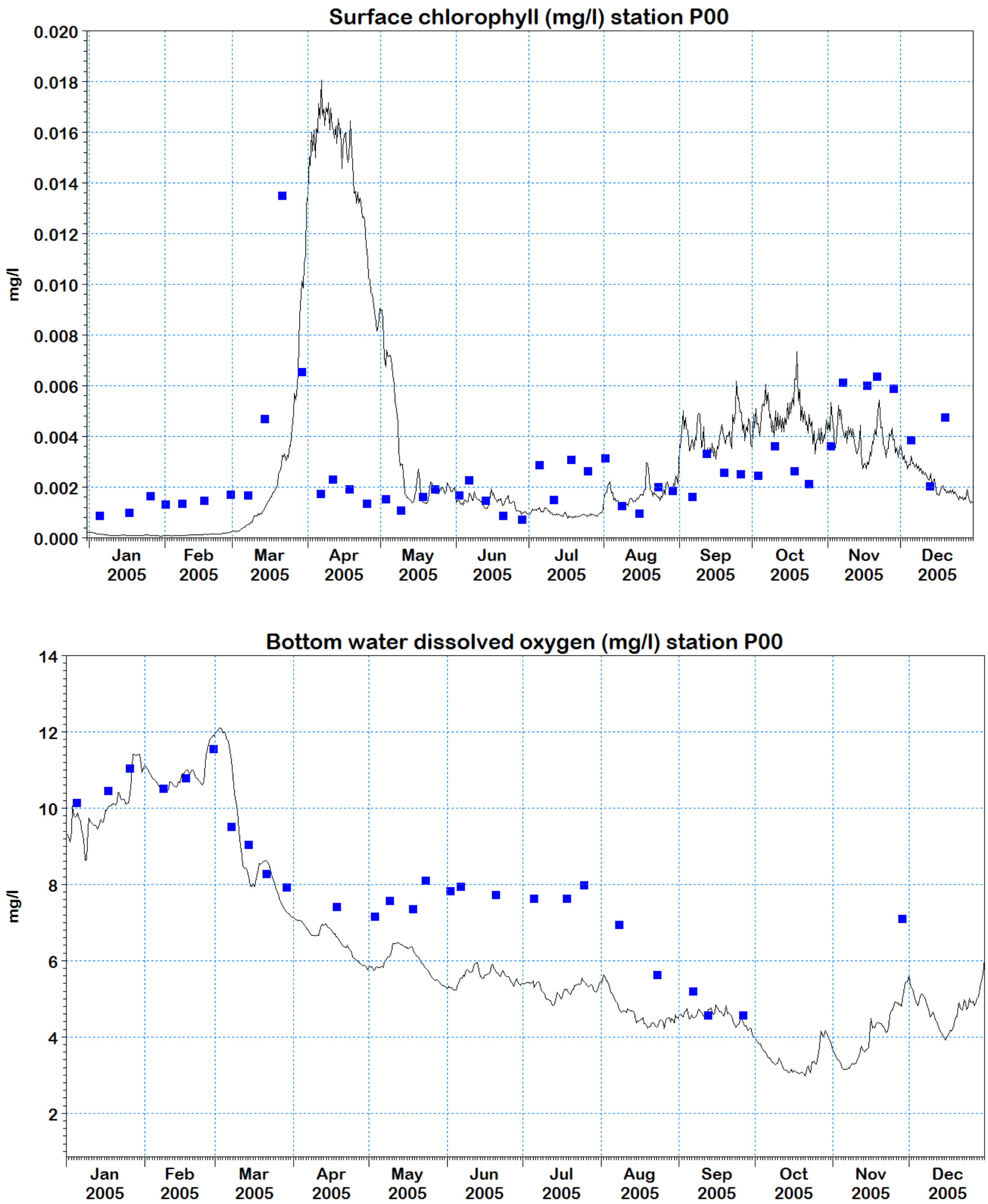


Figure A. 3 Modelled (line) and measured (squares) concentration of chlorophyll in surface water (upper panel) and dissolved oxygen in bottom water (lower panel) in Great Belt in 2005.

Validation of FEMA models for water quality and ecology

The FEMA model was validated against water quality data (inorganic nutrients, chlorophyll-a, dissolved oxygen in bottom waters) collected at 5 stations within the model area (Fig A1). Samples (1, 10, 15, 20m) were collected at monthly – bi-monthly intervals (weather permitting) as part of the Baseline study. Modelled WQ data was available from 1 January 2009 to 1 June 2010. Examples from station H036 are shown in Figures A.4-A.5.

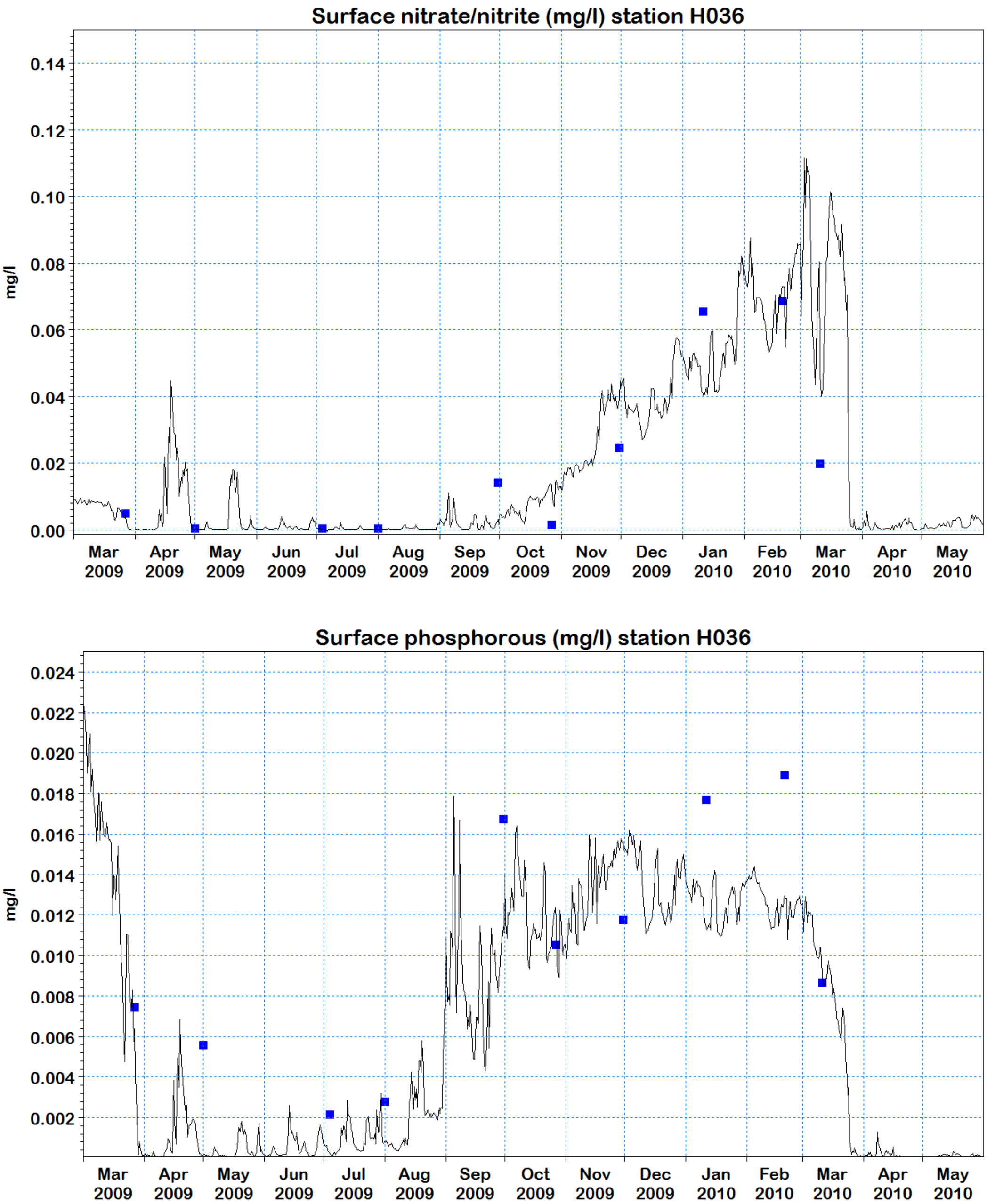
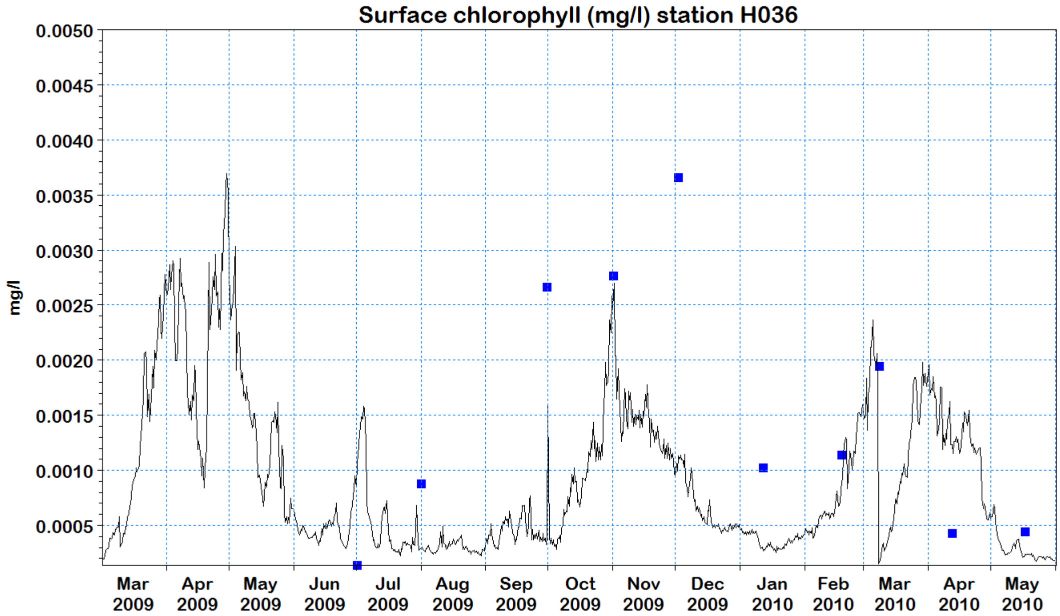


Figure A.4 Modelled (line) and measured (squares) concentration of nitrite + nitrate (upper panel) and phosphate (lower panel) in surface water in central Fehmarnbelt 2009-2010.



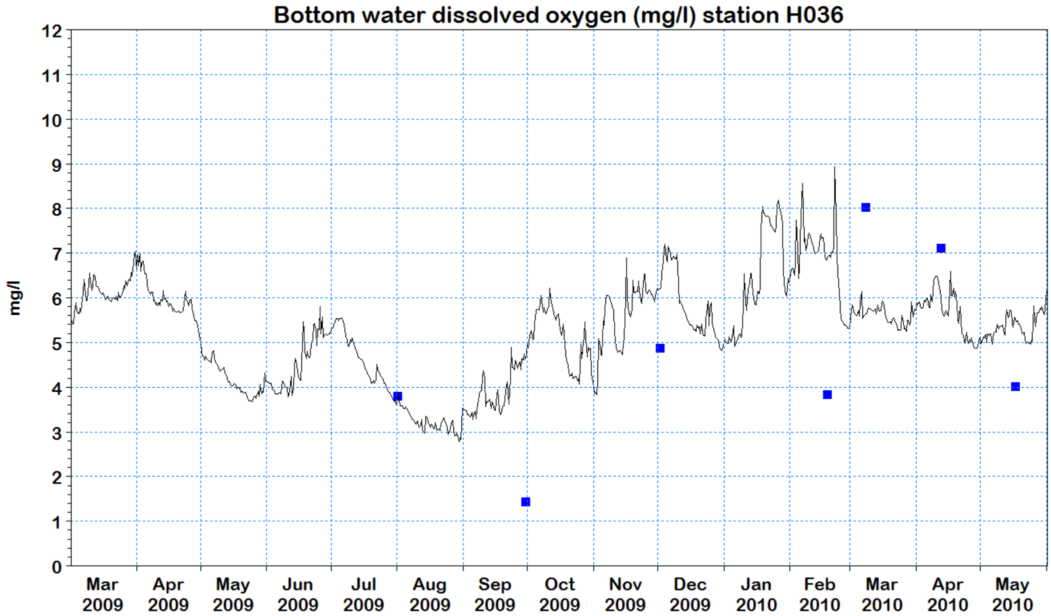


Figure A.5 Modelled (line) and measured (squares) concentration of chlorophyll in surface water (upper panel) and dissolved oxygen in bottom water (lower panel) in central Fehmarnbelt 2009-2010 (including May 2010).

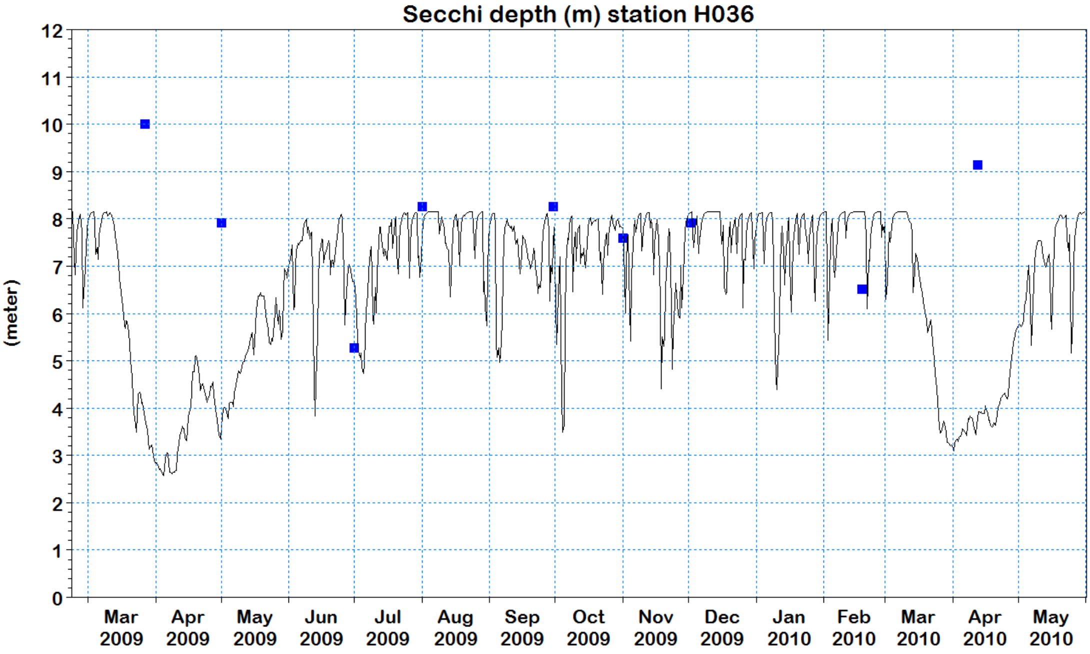


Figure A.6 Modelled (line) and measured (squares) Secchi depth in central Fehmarnbelt 2009-2010 (including May 2010).

A P P E N D I X   B

2-D impact maps of dredging-related impacts for main bridge alternative

AppendiX B: 2-D impact Maps of dredging-related impacts for main bridge alternative

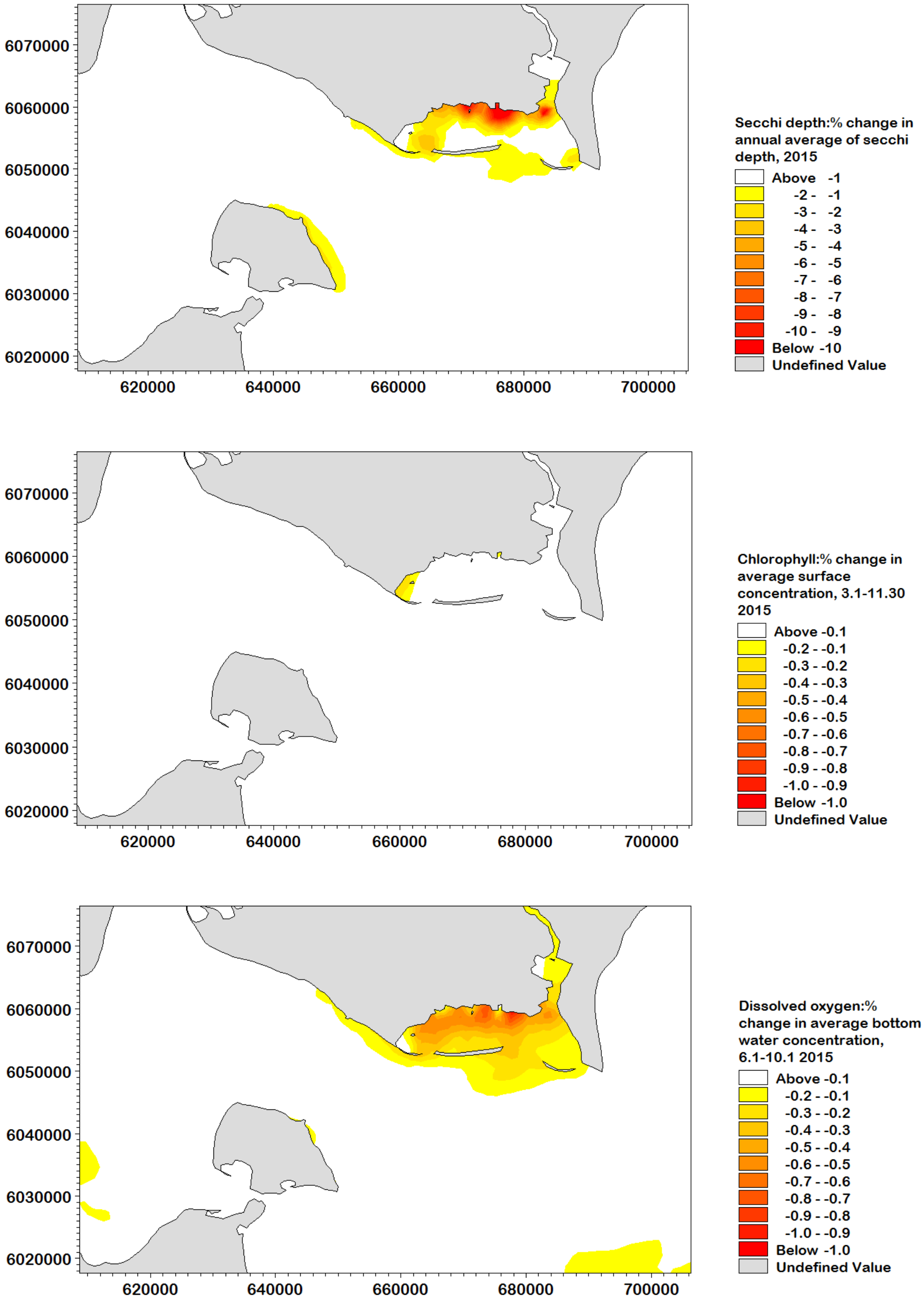


Figure B. 1 %-change in Secchi depth, chlorophyll and dissolved oxygen at bottom averaged over relevant periods (see legend) in 2015.

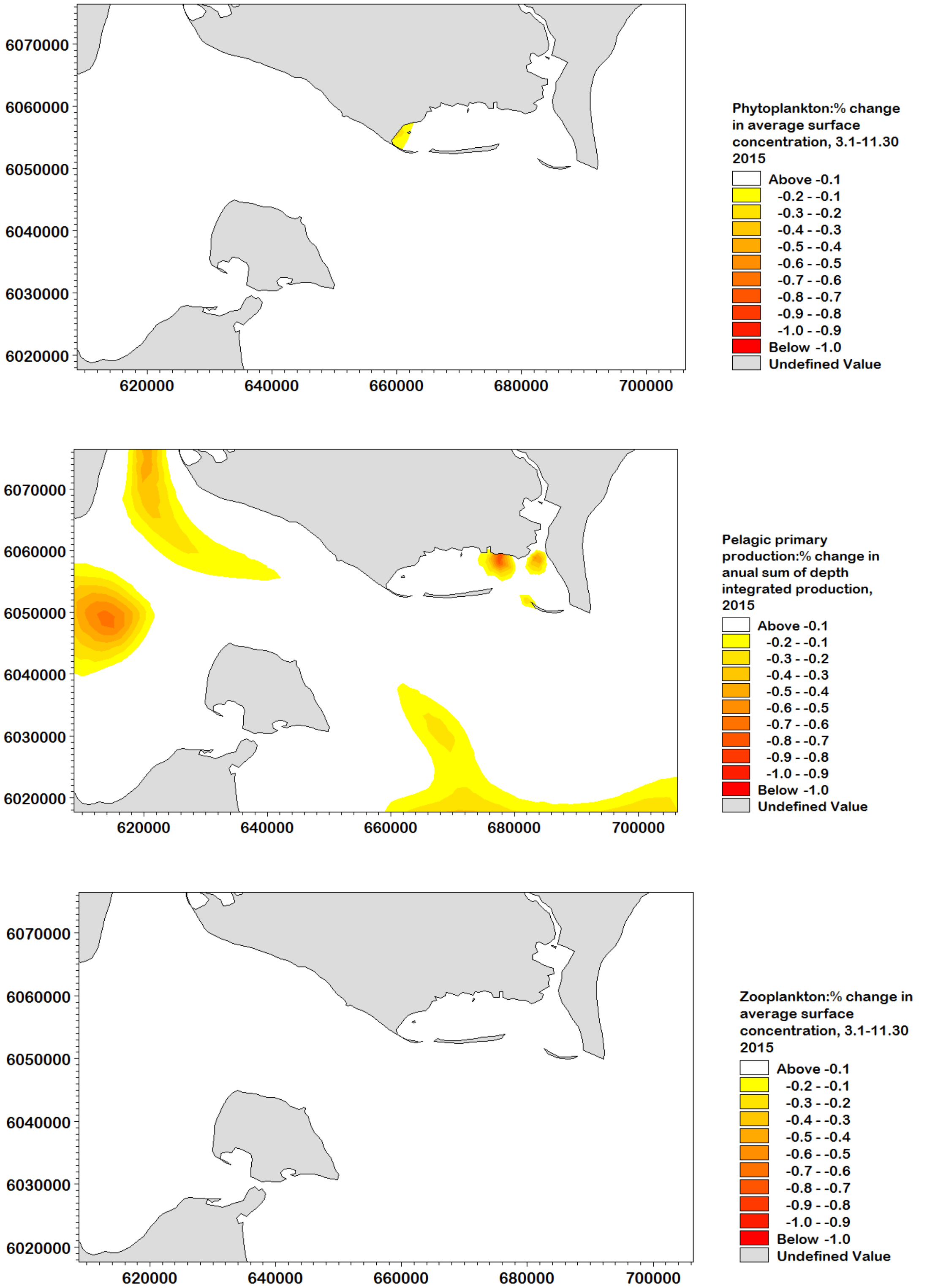


Figure B.2 %-change in phytoplankton biomass, primary production and zooplankton biomass averaged over relevant periods (see legend) in 2015.

A P P E N D I X   C

Investigation of the development of jellyfish polyp populations on additional solid substrate introduced into the Fehmarnbelt area

APPENDIX C: INVESTIGATION OF THE DEVELOPMENT OF JELLYFISH POLYP POPULATIONS ON ADDITIONAL SOLID SUBSTRATE INTRODUCED INTO THE FEHMARNBELT AREA

Introduction

Solid substrate plays an important role in the life cycle of scyphozoan jellyfish because of their alternation between benthic polyp and planktonic medusa stages. Hard substrate positioned in the upper water column is mainly occupied by sessile organisms, which compete for space. The layer close to the bottom can be characterized by summer hypoxia, which favors abundant settlement and high survival of scyphozoan polyps due to their tolerance of hypoxic conditions (Ishii 2010). On the other hand food availability might limit the distribution of the polyps close to the bottom.

Natural substrata reported for polyp settlements of *Aurelia* sp. are *Mytilus* shells, algae, the shells of living or dead barnacles, tunics of ascidians, hydrozoans and bare rocks (Hernroth and Gröndahl 1983; Miyake et al. 2002). Scyphozoan polyps have been observed on horizontal undersurface of artificial hard substrate of floating piers attached specifically to *Mytilus* shells, solitary ascidians, calcareous polychaete tubes, muddy amphipod tubes and to gap space where fouling animals peeled off the substrata (Miyake et al. 2002).

The fact that jellyfish polyps colonize hard substrates in the littoral but do not colonize soft substrates indicate that artificially introduced hard substrate can increase suitable settling surfaces for scyphozoan polyps (Holst and Jarms 2007). Due to anthropogenic activities, the input of those materials into the seas is rising daily.

The objective of this investigation was to determine the potential establishment and development of polyp colonies of *A. aurita* and *C. capillata* if introducing artificial hard substrate such as bridge piers, pylons and scour protection in the Fehmarnbelt area.

Methods

Sampling for jellyfish polyps was included in the monthly FEMA/FEHY baseline investigation with JHC "Miljø". Locations of stations in the investigation area are shown in Figure C.1.

For assessing the population development of the scyphozoan polyps (*A. aurita, C. capillata*) on artificial hard substrate in Fehmarnbelt and Mecklenburg Bight several settlement plates were mounted on moorings (Figure C.2). The moorings were deployed in July 2009 at 6 stations (Figure C.1, Table C.1). The moorings were placed close to the main stations MS01, MS02, MS03 located very close to stations H033, H037 and between H066 and H067 in Figure C.1. Additional settlement moorings were placed in shallow areas next to the near shore stations NS06 and NS10. Additionally, settlement plates were placed in the area of the artificial reef Nienhagen in Mecklenburg Bight.

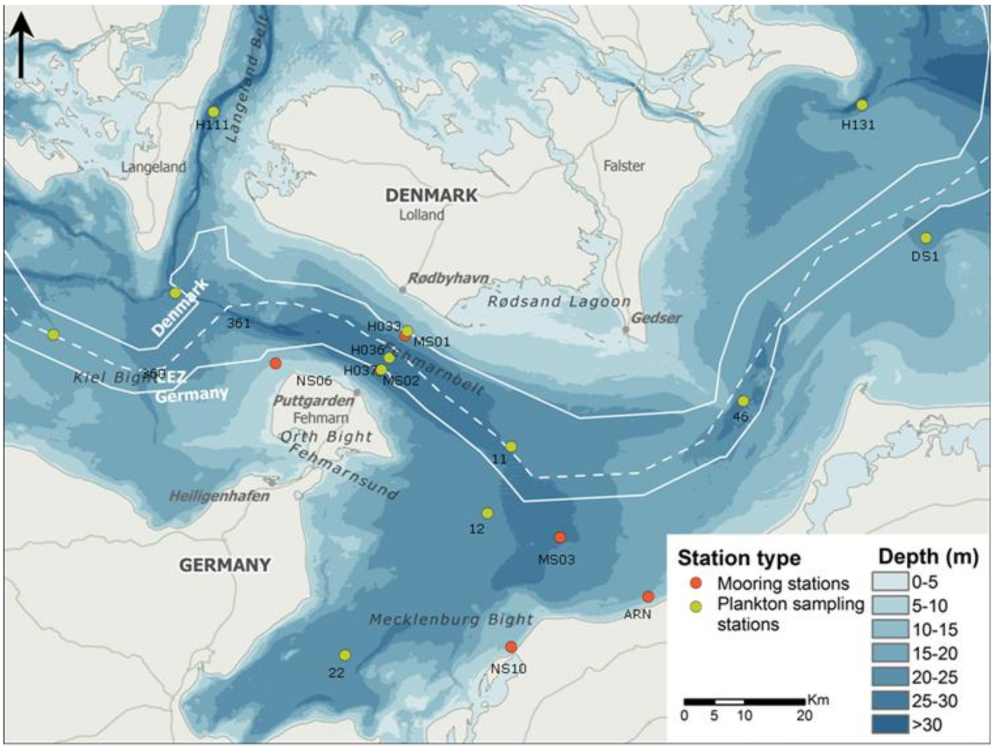


Figure C.1 Map of gelatinous plankton sampling stations (green dots) and polyp settlement moorings (red dots).

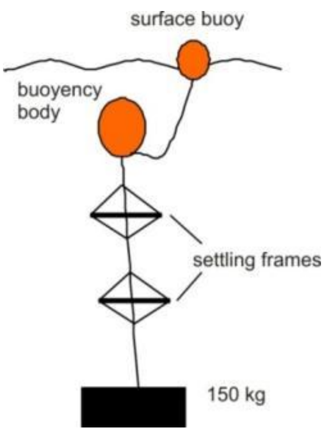


Figure C.2 Polyp settlement mooring system.

Table C.1 Coordinates and water depths of polyp settlement stations (MS: FEHY main stations, NS: near shore stations, ARN: artificial reef Nienhagen Germany)

Station	Latitude	Longitude	Depth (m)	Depth (m) of bottom settling frame	Depth (m) of surface settling frame
MS01	54,585	11,355	19.6	16.5	7
MS02	54,534	11,288	28.0	16	7
MS03	54,275	11,733	25.0	22	7
NS06	54,549	11,020	9.2		8
NS10	54,114	11,598	10.4		7
ARN	54,181	11,952	11.9		3

The moorings consisted of 12-16 removable settlement plates attached to a frame construction of about 1 m diameter (Figure C.3). Areas with a water depth more than 15 m allowed sampling at two different water layers: above and below the pycnocline. At these stations two frames were placed above each other on the same mooring. Hence, depending on the water depth, one or two frames with settlement plates were placed for attracting of polyps.

The "bottom" frame contained additionally to the three horizontal arms one arm in vertical direction. In the following, shallow area moorings with one frame will be referred only to "surface" and deeper areas moorings with two frames will be referred according to their position with "surface" or "bottom" (Table C.1). Each frame contained the same number of plates made either of acrylic glass or concrete, each with a settlement area of 170 cm^2 (15x5x0.5 cm). The plates were placed in the frame and accessible for polyps from all sites.

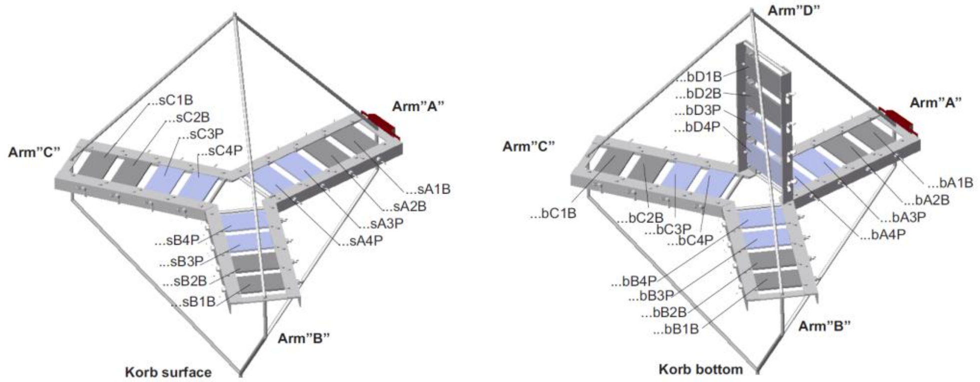


Figure C.3 Polyp settlement mooring surface B. Polyp settlement mooring bottom

The scyphozoan planula larvae were expected to attach upside down and develop into polyps. Sampling of the polyp settlement plates were done regularly during cruises between October 2009 and April 2010 as well as between October 2010 and March 2011 (Table C.2). The samples were obtained by taking the whole mooring out of the water by a crane. Settlement frames "surface" and "bottom" respectively were immediately submersed in individual water tanks on board of JHC Miljø. The plates were placed in seawater containers and stored cool for later analyses. Prior to re-deploying the moorings new settlement plates were installed.

The mooring at the artificial reef Nienhagen (ARN, Figure C.1) was sampled by German scientific divers in cooperation with the University of Rostock, using the research vessel "Gadus".

The attached polyps on all sides of the plates were counted alive in the laboratory or on board under a dissection microscope.

The polyps counted on the settlement plates were used to calculate the natural polyp settlement abundance (polyps cm^{-2}) in Fehmarnbelt and Mecklenburg Bight. During the time of strobilation polyp sampling was analysed seasonally from autumn 2009 until spring 2011. The amount of polyps was calculated to describe differences in polyp abundances below and above the pycnocline as well as between Fehmarnbelt and Mecklenburg Bight. Abundances were calculated as means of the areas of settlement plates. Due to the patched distribution of polyps on the settling plates, maximal polyp densities were also calculated for the area of single settlement patches. All calculations were done for the different exposure sides of the plates. Furthermore, a preference for the substrate was analysed for acryl glass and concrete.

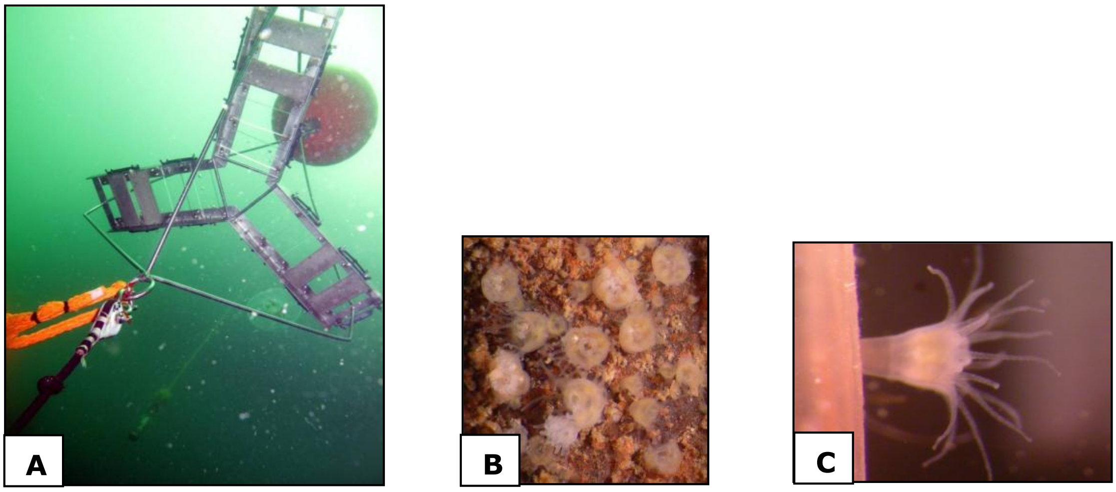


Figure C.4 Durface mooring (Photo: S. Kube), B. polyps on hard substrate, C. single polyp attached on a settling plate (site view) (Photo C. Augustin).

Table C.2 Sampling of polyp settlement moorings during strobilation seasons 2009/10 and 2010/11.

			2009/10							**2010/11**						
			autumn		**winter**			**spring**		**autumn**		**winter**			spring	
month	**subarea**	**remarks**	**O**	**N**	**D**	**J**	**F**	**M**	**A**	**O**	**N**	**D**	**J**	**F**	**M**	**A**
MS01	FB		X		X				X		X		X			X*
MS02	FB	lost Oct 2009														
MS03	MB			X				X	X		X		X		X	
NS06	FB	lost Oct 2009	X													
NS10	MB		X						X		X		X		X	
ARN	MB			X	X				X		X				X	

*sampled after the end of baseline investigation

Results

The settlement plates were covered with different benthic and epibenthic species such as blue mussels, bryozoans and barnacles. Scyphozoan polyps were found within the assemblage of epibenthos. Polyps were determined as *A. aurita* by strobilation experiments in aquaria and subsequent determination of strobilated ephyrae.

In the first year single polyps were observed randomly distributed across the settlement plates (20-60 polyps per plate), while in the second year the polyps were distributed in patches (up to 600 polyps per plate). The patchy distribution indicated the growing of established colonies by asexual reproduction of polyps. The highest polyp abundance within a patch was 34 polyps cm^{-2}. Reproducing polyps showed 2 to 5 strobila.

Temporal variation of polyp abundance

The mean abundance calculated for the total area of the whole plates varied between 0.01 to 0.14 polyps cm^{-2} in the first year of exposure (autumn 2009 to spring 2010) and between 0.08 to 1.66 polyps cm^{-2} in the second year of exposure (autumn 2010 to spring 2011, Figure C.5). The tenfold interannual increase could be explained by two processes: (1) the settlement of new planula larvae in summer 2010 and, (2) asexual reproduction of polyps during 2010. Most likely, the latter process played the major role for the increase of the interannual abundance. The settlement of planula larvae would have caused a rather randomly polyp distribution pattern, whereas the reproduction of the sedentary polyps would cause the patchy distribution discovered on the plates. Comparing the polyp distribution within the strobilation seasons 2009/10 and 2010/11, there was no clear pattern of polyp abundances observed (Figure C.6).

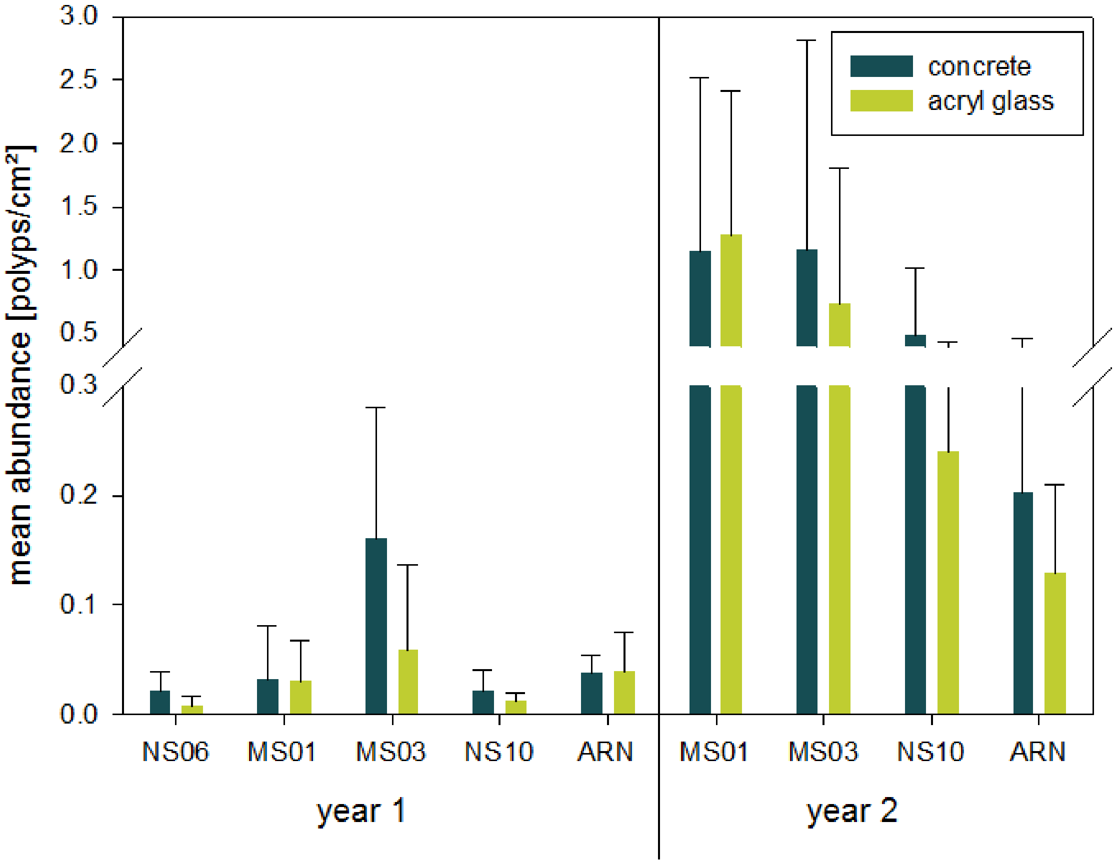


Figure C. 5 Abundance of Aurelia aurita polyps on settling plates made of concrete (blue) and acryl glass (green), respectively (means ± sd of all samplings per strobilation season above and below halocline);. year 1: strobilation season 2009/10, year 2: strobilation season 2010/11.

Spatial variation of polyp abundance

Horizontal distribution

With a mean abundance of 0.15 polyps cm^{-2}, station MS03 in the Mecklenburg Bight showed the highest polyp abundance of the first strobilation season 2009/10 (Figure C 5). All other stations showed low mean abundances of less than 0.05 polyps cm^{-2}. In the second year the highest polyp abundances were observed at stations MS01 (Fehmarnbelt) and MS03 (Mecklenburg Bight) with about 1.0 polyps cm^{-2} (Figure C.5). Mean abundances were lowest at stations ARN (0.2 polyps cm^{-2}).

Vertical distribution

Vertical distribution patterns were compared for stations MS01 and MS03, which were equipped with settlement frames in two different depths (Figure C.3, Table C.1). Scyphozoan polyps clearly preferred the settlement plates placed above the pycnocline (mean of all samplings 2 - 4 polyps cm^{-2}) compared to the plates below the pycnocline (mean of all samplings <1 polyp cm^{-2}, Figure C.7). This might be caused by favourable environmental conditions in the surface water layer such as a higher food supply and higher oxygen abundances.

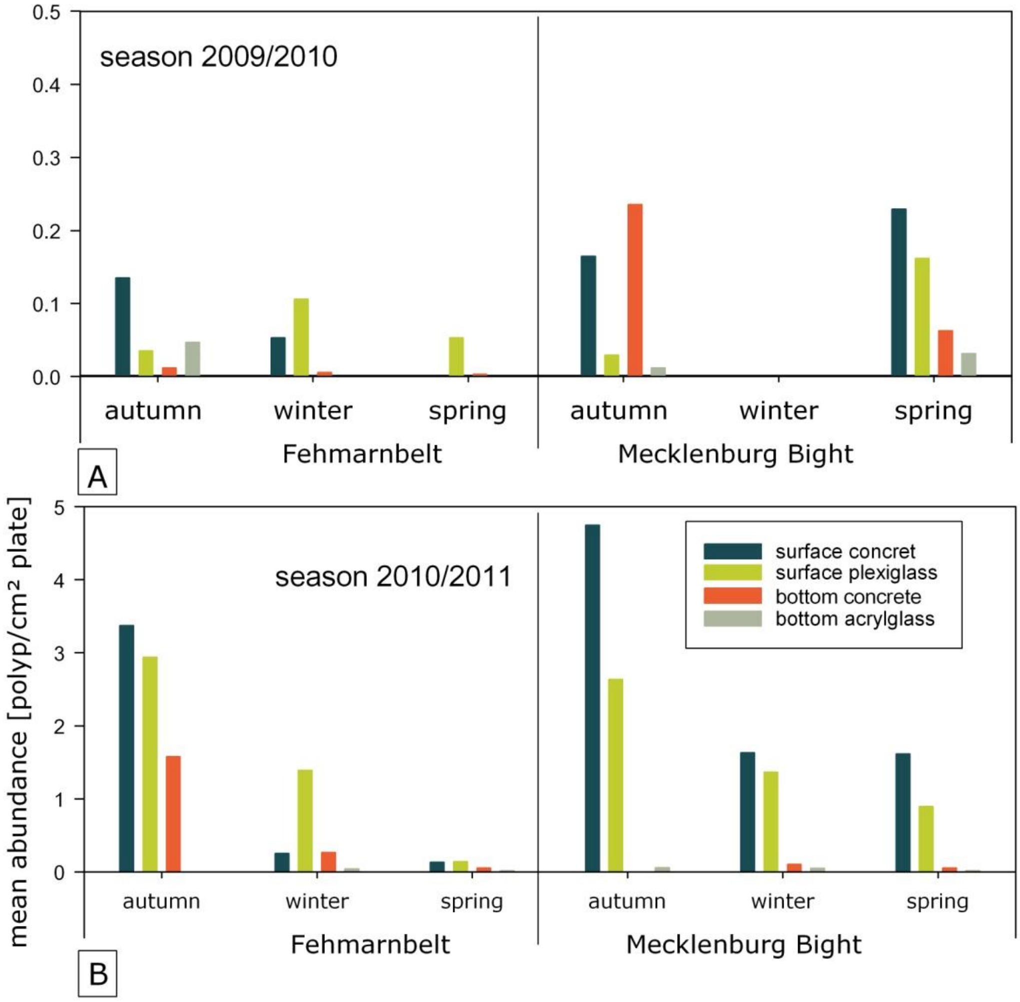


Figure C.6 Abundance of Aurelia aurita polyps as mean for plates and seasons according to their position in the water column (above or below halocline) and material (concrete or acryl glass). A. strobilation season 2009/10, B. strobilation season 2010/11.

The impact of the orientation of the exposure side

Comparing the abundances of the polyps settled on each side of the settlement plates highest abundances were found on the underside (Figure C.7). Polyps were attached upside down, which is known to be the preferred orientation (Holst and Jarms 2007). High abundances of the underside of plates have previously been reported for polyps of *Aurelia labiata* (Hoover and Purcell 2009).

Abundances on sides oriented vertically were much lower with a preference of the "short" side compared to the "long" side of the settling plate. This difference might have been due to the construction of the settling frames. The plates were attached to the frame on their short side. It is thus likely that polyps on the short sides of the plates were better protected for being overgrown by blue mussels or other epibenthic organisms. Hardly any polyps were observed on the upper sides of the sampling plates (Figure C.7).

The effect of the substrate type

There were no significant differences in the mean abundance of settled polyps on the different settling plate materials, concrete or acryl glass, based on the comparison of samples from all areas (Figure C.5) and from two vertical positions of the exposed settlement plates (Figure C.6).

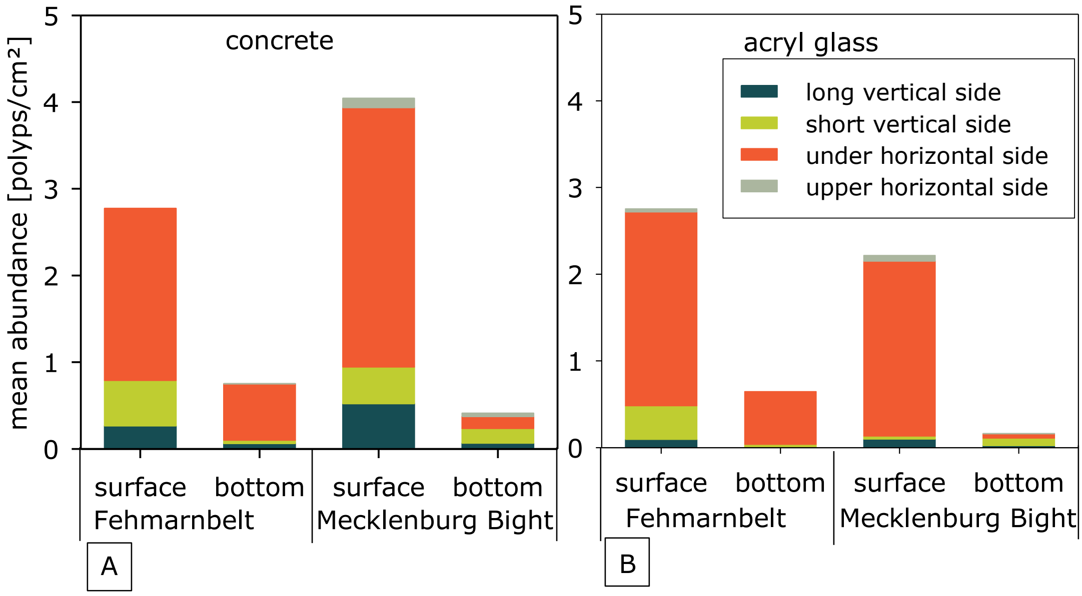


Figure C.7 Mean abundance of Aurelia aurita polyps below and above the halocline (surface and bottom) in Fehmarnbelt (station MS01) and Mecklenburg Bight (station MS03). Colours indicate the different exposure sides of the settling plates. A: concrete plates, B: acryl glass plates

Conclusion

Jellyfish polyps and different other benthic and epibenthic species were found to settle on the artificial settling plates moored in the Fehmarnbelt area. The polyp colonies preferred to settle on the down-side of plates and concrete was a slightly more attractive settling material than acrylic glass. The mean abundance calculated for the total area of the whole plates varied between 0.01 to 0.14 polyps cm^{-2} in the first year of exposure (autumn 2009 to spring 2010) and between 0.08 to 1.66 polyps cm^{-2} in the second year of exposure (autumn 2010 to spring 2011). Most polyps were found above the pycnocline; approx. 2 ind cm^{-2} above the pycnocline and 0.2 ind cm^{-2} below the pycnocline. The establishment of polyps had probably not reached the final (stable) abundance after 2 years of exposure in the Fehmarnbelt area. Nevertheless, 2 ind cm^{-2} can be used for estimating the potential additional population size of polyps on new areas of hard substrate (piers, pylons, scour protection) introduced into the Fehmarnbelt area.

A P P E N D I X   D

Light attenuation of Fehmarnbelt suspended sediments

APPENDIX D: LIGHT ATTENUATION OF FEHMARNBELT SUSPENDED SEDIMENTS

Background

Suspended solids such as spilled sediments from dredging operations add to light attenuation in the water column thereby reducing light intensity reaching the seabed and thus affecting the benthic vegetation. Suspended solids differ in their optical properties, where the organic content, size distribution and shape of particles are important for the mass-specific light attenuation (Baker and Lavelle 1984; Bowers and Binding 2006; Woźniak et al. 2010).

The attenuation of light is the combined effects of two processes in the water column, namely the scattering of light and absorption of light. The scatter of light scales to cross-sectional area of particles (living and dead, inorganic), while the mass-specific scatter (b*) including a diffraction effect can be described by:

$$b^* = \frac{3}{D\rho_P}$$

where D is the diameter of a (spherical) particle and ρ_P is the density of the particle (Bowers and Binding 2006). Besides area, surface properties of particles such as their refractive index are important for the mass-specific scatter (Babin et al. 2003). In the real aquatic environment suspended particles are not perfect spheres and the projected area of a natural inorganic particle can easily be an order of magnitude higher than a sphere of similar mass leading to higher mass-specific scatter (Peng and Effler 2007).

Although scattering does not "remove" photons from the water column, scattering is considered a light extinction phenomenon because it increases the path length of photons and thus the probability of photons being absorbed by the absorbing components in the water column.

Several constituents in natural waters can absorb light. Ranged in decreasing order chlorophyll pigments and other light harvesting pigments in planktonic algae have the highest the mass-specific absorption coefficients, followed by organic matter (living, dead and dissolved), inorganic particles and water itself. If follows from the above, that all particles contribute both to scatter and absorption, but that absorption dominates in organic particles (especially in phytoplankton due to light-harvesting pigments), while scatter dominates in inorganic particles.

The combined effect of scatter and absorption of suspended particles on light attenuation varies between and within coastal areas, shelf and off-shore seas, both as a function of differences in concentrations of chlorophyll-a, detritus, inorganic suspended solids and dissolved organic matter, but also caused by variation in the optical properties of suspended particles. In the scientific literature *in situ* mass-specific light attenuation coefficient of suspended solids (primarily inorganic) has been found to vary between 0.04 and > 0.5 $m^2\ g^{-1}$ (Bowers et al. 2009, Campbell & Spinrad 1987, Devlin et al 2008, Dixon & Kirkpatrick 1995, Gallegos 2001, William et al 2002, Lund-Hansen et al. 2010), with highest values in waters dominated by small-sized particles and/or with some contribution of organic matter in the particulate pool (Hill et al 2011). With such a large range in mass-specific light attenuation coefficients "standard" coefficients cannot be applied universally to any dredging situation, as it can lead to serious bias in prediction in effects. Instead, one should use site-specific attenuation coefficients when predicting effects of sediment spills from dredging works. To this end, three experiments with different sediment types from the alignment were carried out.

Method

Light attenuation of suspended sediment from two different stations and 3 depth strata was quantified measuring light transmission in laboratory experiments. Briefly, a 50 W water-proof halogen lamp (beam angle 30-35 degrees) fitted with a BG-34 filter (mimicking the spectrum of natural light) was used as light source, and a Licor LI-192 (π) Underwater Quantum Sensor placed at a distance of 37 cm was used to quantify light intensities. Both the lamp and Licor sensor were fixed to a common bar. The light transmission over time (7-8 sample times over 24 h) was measured in a 100 L circular black-walled container filled with 'artificial' seawater (20 ‰ NaCl) and added suspended sediment. Light intensities were recorded on a LI-1000 Data Logger. Position of sediment samples used in experiments and brief characteristics of whole sediment are shown in Table D.1.

Table D.1. Position where sediment sample was taken, sediment depth interval and los on ignition and organic carbon in sediment sample.

Sample	Latitude	Longitude	Sediment depth interval	LOI / DOC (% of DW)
A002-1	54.50950	11.2500	0-30 cm	3.15 / 0.95
A006-1	54.55833	11.30617	0-30 cm	2.96 / 1.07
A006-3	54.55833	11.30617	70-100 cm	1.41 / -

Preparation of sediments
Using a syringe with a cut end subsamples of sediment (ca. 10 ml) from selected sediment strata were transferred to 2 L Erlenmeyer flasks filled with 20 psu 'artificial' seawater and placed on a magnetic stirrer for 12 h. Prior to experiments the stirrer was stopped and larger particles (i.e. fine sand) allowed to settle for 30 min.

Experiment
A subsample (≈ 20%) of the supernatant was added to the 100 L experimental container to reach a final concentration of suspended sediments between 10 and 20 mg/l. After thorough mixing, measurements of light transmission were initiated after 10 min and continued at increasing time intervals until 12-24 h after start. During this period the suspension in the container was left unmixed. Light transmission measured prior to adding suspended solids provided data on 'background' transmission related to artificial seawater only. At termination of an experiment the entire volume of water was filtered through a 1 µm in-line filter connected serially to a peristaltic pump, and the light transmission through filtered water was measured after thorough cleaning of the experimental container. The light transmission through filtered experimental water was carried out to quantify the attenuation due to dissolved organic matter and colloid material originating from the sediments (e.g. from pore water).

Particle sizes
Along with the light transmission measurements water samples were taken at the depth-level of light beam (3 positions sampled simultaneously using a peristaltic pump) and the particle size distribution (264 bins) was measured using an electronic particle counter (Coulter Counter Multiseizer, fitted with a 70 µm tube (measur-

ing range: diameter 1.3 µm – 42 µm), see below. At the start of sampling and at the end 1 L samples from the "light beam" height in the container were filtered onto combusted and pre-weighed 47 mm GF/C filters for determination of suspended solid concentration and Loss on Ignition.

During experiments water was sampled and analysed using an electronic particle counter to quantify concentrations of different sized particles. During the course of an experiment the number and the volume of particles decreased, especially the larger particles, while the reduction in concentration of small particles below 2-3 µm was much less (Figure D.1) due to a lower settling velocity.

After subtraction of the attenuation value from filtered experimental water the attenuation of light due to particles was related to concentration of suspended solids described by the total volume (summed for all size classes of particles) and the cross-sectional area of particles (summed for all size classes) assuming that particles were present as spheres. The attenuation by dissolved organic matter (in post-filtered experimental water) was related to the initial particle volume and initial dry weight of suspended matter in the individual experiments.

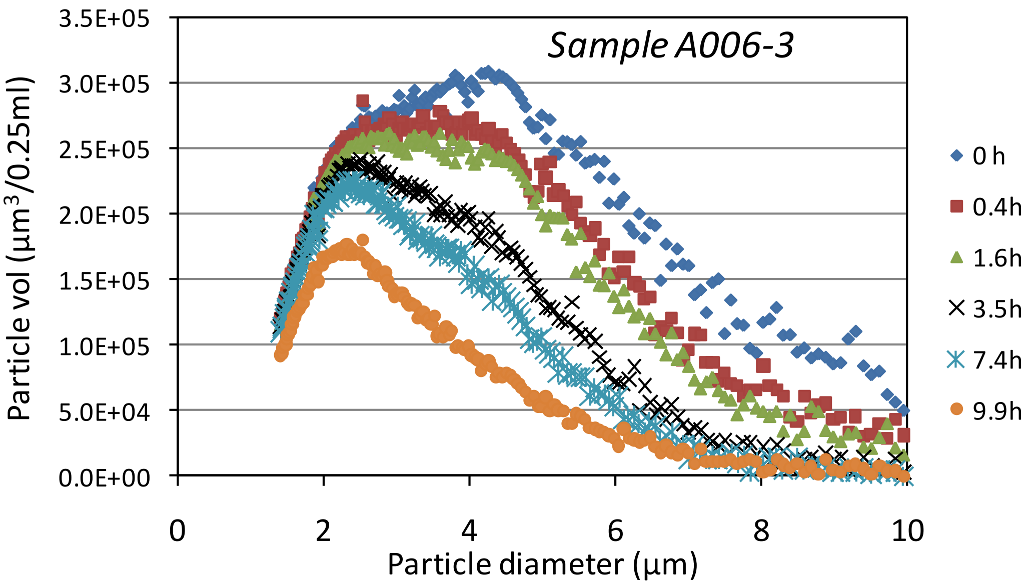


Figure D.1 Size distribution of particle volume over incubation time (0-9.9h) in light attenuation experiments using suspended sediments from sample A006-3 (Station A006; 90-120 cm depth). Concentration of 6 µm particles is reduced by 90% after 9.9h at depth of light path, while 1.5 µm particles are reduced by 12-15% only, due to a lower settling velocity.

Results

Light attenuation varied between sediment samples due to differences in concentration and size distribution and, caused by variation over time, due to differential settling of different sized particles (Figure D.1).

For the individual experiments with the same sediment type the light attenuation coefficient scaled almost linearly to the total particle volume, but relations differed markedly between experiments with different sediment types, because of different size distribution between sediments from station A002 and station A006 (Fig. D.2 upper).

In contrast, if attenuation was plotted against total cross-sectional area of particles all samples fitted to a common line irrespective of differences in size distribution (Figure D.2, lower panel). The fact that light attenuation scaled linearly to cross-sectional surface area of particles is a strong indication that attenuation primarily is due to light scattering rather than absorption.

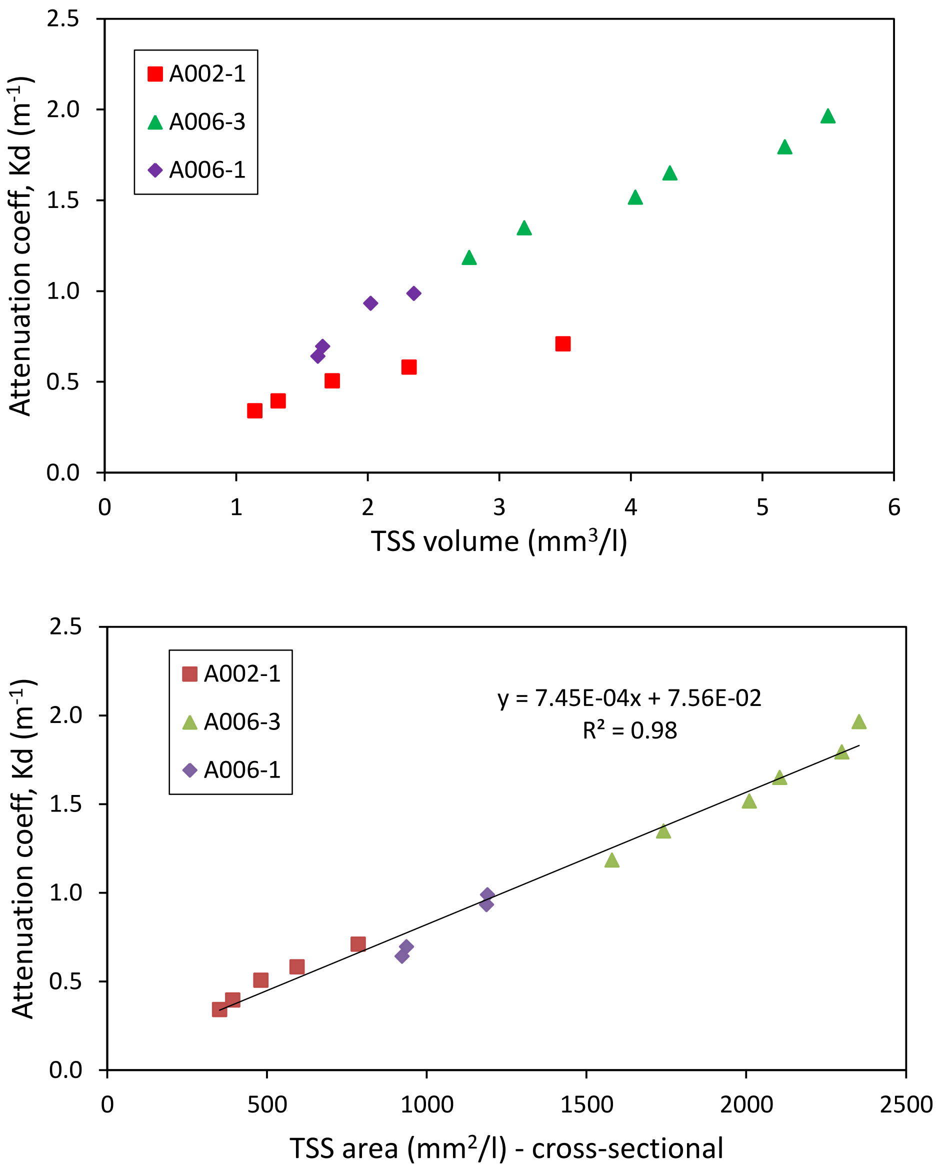


Figure D.2 Light attenuation coefficient, Kd as function of total (summed) particle volume (upper panel) and total cross-sectional particle area (lower panel). Common linear regression line and equation relating summed cross-sectional area and Kd shown for the 3 experiments conducted.

Implementation of light attenuation values from experiments with suspended solids from Fehmarnbelt in the FEMA model.

In the FEMA model light attenuation is described by the Kirk formula:

$K_d = [(a_w+a_{al}+a_{ss}+a_{doc}+a_{dc})^2 + c*(a_w+a_{al}+a_{ss}+a_{doc}+a_{dc})*(b_{al}+b_{ss}+b_{dc})]^{0.5}$,

where a_w, a_{al}, a_{ss}, a_{doc}, a_{dc} represent the absorption due to water itself, algae, dissolved organic matter and detritus, respectively, and b_{al}, b_{ss}, b_{dc} represent scatter caused by algae, suspended solids and detritus. The constant c was fixed at 0.256. The Kirk formula must be regarded as state-of-the-art and is used in a large number of scientific studies in preference to common practice where weight-specific attenuation coefficients are calculated (or obtained from literature) and summed to a common attenuation coefficient.

The attenuation experiments using suspended sediments from the Fehmarnbelt showed that light attenuation scaled linearly to cross-sectional surface area of particles which indicate that attenuation primarily is due to light scattering rather than absorption (see Figure D.3). The linear equation between attenuation (Kd) and summed cross-sectional area, A:

$$K_d = 7.45*10^{-4}*A + 0.0756$$

may suggest that the y-axis intercept (0.0756 m^{-1}) represents an average absorption of experimental (particle-free) water, but the intercept was not different from 0 (t-test).

Dissolved organic matter (or colloid material passing 1 µm filter) did contribute to light attenuation in the experiments. Based on the limited amount of data (3 experiments) the initial concentration of TSS scaled linearly to light attenuation (corrected for attenuation of artificial seawater) measured in filtered water after the experiments with a specific attenuation coefficient of 0.028 g/m^2 (Figure D.3). It should be stressed that the initial TSS concentrations in experiments only represent 10-20% of the total sediment weight initially suspended. The major part of total consists of fine sand, that settled prior to the supernatant containing fines was added to the experimental container.

The contribution of absorption, a and scatter, b to the measured attenuation in the lab experiments was estimated under two different assumptions:

1. Absorption from dissolved matter will be part of the additional light attenuation when sediment is spilled from dredging and, dissolved matter is the only absorbing agent in spilled sediment

2. Absorption from dissolved matter will not contribute to the additional light attenuation when sediment is spilled from dredging. Instead, spilled sediment absorp light according to their mass with a constant weight-specific attenuation coefficient.

Scatter coefficients b was estimated by regression:

$$Kd = (a^2 + 0.256 * a * b)^{0.5}$$

where

- Kd is the measured light attenuation in experiments

- a is the measured absorption coefficient due to dissolved matter, i.e 0.028 m^2/g (see Figure D.3) or

- a is calculated using a weight-specific absorption coefficient (representing suspended solids with a low organic content) of 0.00915 (Bowers et al. 2009)

- b is the scatter coefficient expressed by: c*A, where c is determined by regression.

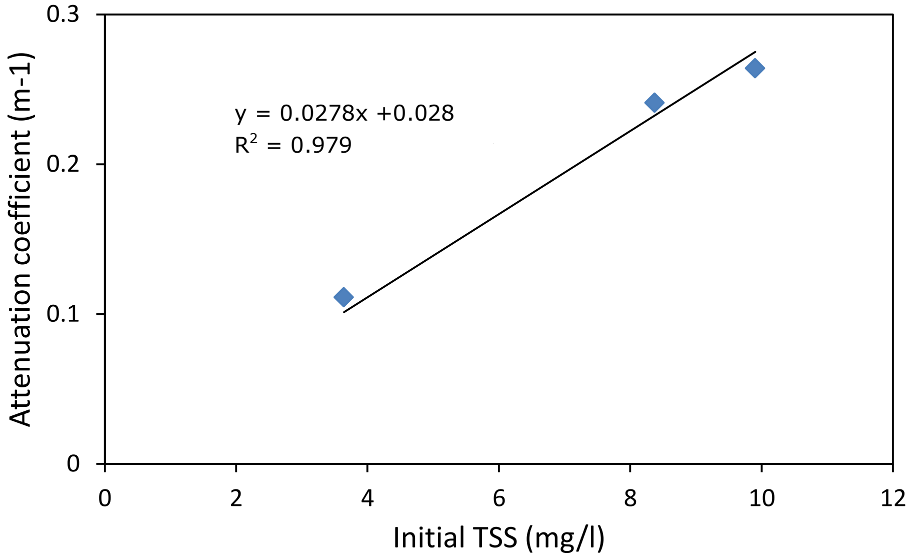


Figure D.3 Light attenuation in filtered experimental water as function of initial TSS concentration in experiments

Comparison of observed and modelled (by regression) light attenuation using a value of 12 for c in scaling the magnitude of scatter is shown in Figure D.4.

With this information we can calculate the contribution of scatter to total attenuation by multiplying the cross sectional area of the 4 size classes of particles modelled in sediment spill scenarios under the assumption of dissolved material participate in the light absorption in sediment spill and further is linearly related to sediment weight irrespective of the particle size (see Table D.3).

In the FEMA modelling we have used the absorption and scatter coefficients where dissolved organic matter is included.

If dissolved (organic) matter is excluded in the attenuation, scatter coefficients will increase slightly (Table D.2) but the mass specific attenuation coefficients will be reduced by ca. 40%.

Table D.2 Optical properties of different sized particles. Mass-specific absorption (a - m^2/g), scatter (b – m^2/g) coefficients and mass-specific attenuation coefficients (Kd – m^2/g)

Sediment fraction	**Reference (incl. diss. attenuation)**		**Specific Kd (m^2/g)**	**Sensitivity test (dissolved matter not included in attenuation)**		**Specific Kd (m^2/g)**
Diameter (mm)	**a**	**b**		**a**	**b**	
0.064	0.0278	0.354375	0.057	0.00915	0.39375	0.032
0.028	0.0278	0.756	0.078	0.00915	0.84	0.045
0.010	0.0278	1.8144	0.117	0.00915	2.016	0.069
0.0065	0.0278	2.713846	0.142	0.00915	3.015385	0.085

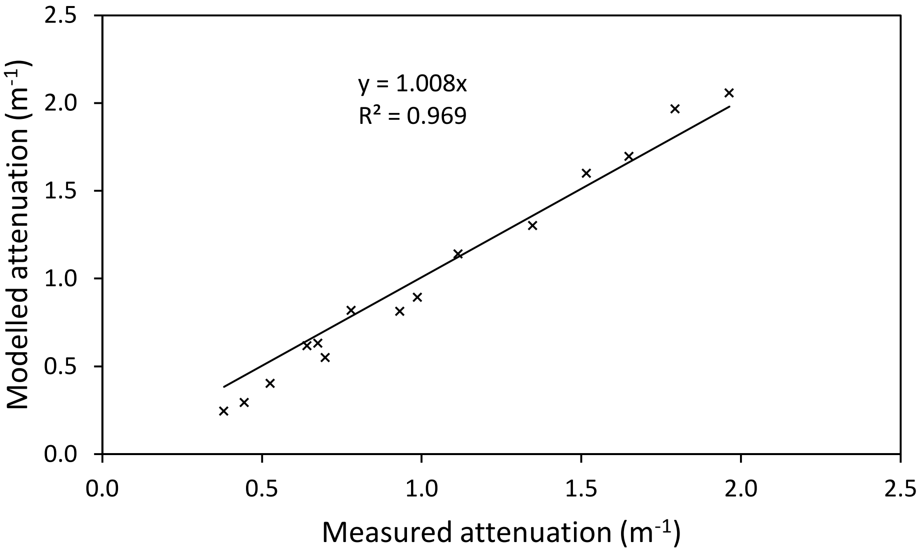


Figure D.4. Comparison of measured and modelled light attenuation in experiments using a scaling factor c for scatter at 12.

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