Synthesis report on the effects of dredged material dumping on the marine environment (licensing period 2017-2021)

Brigitte Lauwaert[1], Bavo De Witte[2], Felien Festjens[2], Michael Fettweis[1], Laurens Hermans[3], Amélie Lesuisse[2], Hong Minh Le[1], Stephie Seghers[2], Steve Timmermans[4], David Vanavermaete[2], Gert Van Hoey[2]

Colophon

[1]RBINS-ODNature-MUMM: Royal Belgian Institute of Natural Sciences, Operational Directorate Natural Environment, Management Unit of the North Sea Mathematical Models, Vautierstraat 29, B-1000 Brussels

[2]ILVO: Flanders research institute for agriculture, fisheries and food, Animal Science - Aquatic Environment and Quality, Ankerstraat 1, B-8400 Ostend

[3]AMT: Department of Mobility and Public Works, Maritime Access Division, Thonetlaan 102, B-2050 Antwerp

[4]AMCS-CD: Department of Mobility and Public Works, Agency for Maritime and Coastal Services, Coastal Division, Vrijhavenstraat 3, B-8400 Ostend

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Email and telephone of corresponding authors
blauwaert@naturalsciences.be: +32(0)2 7732120
laurens.hermans@mow.vlaanderen.be: +32(0) 492183216
gert.vanhoey@ilvo.vlaanderen.be: +32(0)59 569847

Also participated in the research

RBINS-OD Nature: Joan Backers, Matthias Baeye, Xavier Desmit, Tjorven Ditillieu, Frederic Francken, Kyra Gesquiere, Kevin Hyndrickx, Nicky Jespers, Koen Parmentier, Tom Scholdis, Nathan Terseleer, Dries Van den Eynde, Dimitri Van der Zande, Wim Vanhaverbeke, Markus Schartau (Geomar), Rolf Riethmüller (Hereon Centre).

ILVO: Charles Lefranc, Jan Wittoeck, Hans Hillewaert, Jan Ranson, Naomi Breine, Kris Hostens, Kevin Vanhalst, Lode Jacobs, Carina Pardon, Michael Dekimpe, Patrick Calebout, Eddy Buyvoets

AMT: Greet Mendonck

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The photo on the front page is a Sentinel-2A image from 5 May 2016 (contains modified Copernicus Sentinel data, see https://odnature.naturalsciences.be/remsem/software-and-data/acolite) showing the Belgian nearshore area around the port of Zeebrugge and the dumping site ZBO, where the dark spot contains the nearfield plume of the dumped sediments. The dredging vessel is sailing back to the dredging site. The white lines are traces of the spring Phaeocystis bloom.

Content

1. Introduction

Dumping at sea of dredged material is carried out in accordance with the federal law of 20th January 1999 and a permit is given in accordance with the procedure defined in the royal decree (RD) of 12th March 2000, and revised by the RD of 18th October 2013 by which the validity period for the permits has changed from 2 years till 5 years. Corresponding to article 10 of this procedure, every 5 years a “synthesis report” has to be established for the Minister who has the North Sea under his competences. After 2.5 years a “progress report” has to be written and sent to the Minister. The synthesis report needs to include recommendations which support the development of an enforced environmental management (see chapter 2). The current synthesis report covers the period 2012-2016.

Permits for dumping of dredged material at sea were given to the Maritime Access Division who is responsible for maintaining all maritime access channels to the coastal ports as well as to the Coastal Division of the Agency for Maritime Services and Coasts who is responsible for the maintenance of the coastal marinas. In the ministerial decree (MD) of 19 December 2013 (BS 16.01.2014) the validity of the MD for the dumping of dredged material at sea for both Flemish authorities (AMT and AMCS-CD) has been prolonged until 31 December 2016, in accordance with the royal decree (RD) of 18 October 2013. The permitted and the actual dumped quantities are presented in chapter 3.

The international framework for dumping at sea of dredged material is the (regional) OSPAR Convention (1992) and the (worldwide) London Convention (1972) and Protocol (1996). These conventions and their associated guidelines consider the presence of any contaminants within the sediment and whether some alternative beneficial use is possible. In implementing these guidelines, e.g. action levels (sediment quality criteria) have to be defined, dumping sites have to be chosen and a permanent monitoring and research program has to be carried out (see chapters 4 to 7). Recently, the Marine Strategy Framework Directive (EC-MSFD 2008), has been implemented (Buhl-Mortensen et al. 2017). The Directive is based on an ecosystem approach to manage the impact of human activities on the marine environment through the establishment of targets and associated indicators.

2. Recommendation to the Minister

2.1. Present state of the recommendations

Policy recommendations

Recommendation	Intermediate status (2021)
Further to the research carried out during the period 2009-2016, the study of the practical implementation of a new dumping site west of Zeebrugge needs to be continued. The research should focus on possible alternatives, concerning the location as well as the exploitation scenarios, and the environmental impact of these possible alternatives should be investigated. The latter will serve as input for the EIA	Zeebrugge West (ZBW) has been assigned as a new dumping site.In the upcoming permit period 2022-2026, a permit for this new zone will be applied for in combination with the existing zone ZBO. This new zone is within the search area of the MRP 2020-2026 and will be applied for as a dumping site in the MRP 2026-2034.
The remaining capacity of dumping site S1 is limited. In the near future possible alternatives for the dumping site need to be investigated. A new search area has to be defined, comparable as with the alternative dumping location of Zeebrugge West. This search area can be used as input for the modification of the MSP (2020-2026).	The new search area has been defined, confirmed and approved and is incorporated in the MRP 2020-2026. Further research has not yet started
The research on dumping methods and sites for the dredging at Blankenberge and Nieuwpoort needs to be continued.	Because of technical difficulties (pumping of dredged material) the item will not be continued, see Lauwaert et al. (2019, §5.3) and appendix 2.
The monitoring and evaluation of indicators relevant for the dumping of dredged material for the MSFD - Good Environmental Status needs to be developed further.	Has been done and is included in the MSFD report of 2019, see Lauwaert et al. (2019, §5.2) and appendix 2.

Policy supported research

Recommendation	Intermediate status (2021)
With the use of the Sediment Profile Imaging (SPI) technique, near bed ecological and sedimentological processes need to be better investigated.	This SPI technique is tested multiple times at S1, as outlined in chapter 5.4. It indeed proofs its ability to detect near bed ecological and sedimentological processes. It gives complementary information compared to the classical grab sampling and allows a visual overview of the sediment characteristics at the dumping site.
Specific emphasis needs to be given within the MSFD framework to "Marine Litter". Further research to the definition of a baseline and of the origin of the litter is needed. If relevant then the research should be carried out in cooperation with other actors.	Within this report, data on macro litter on the seafloor from 2013 to 2019 was used to set a baseline of seafloor litter contamination at the Belgian Part of the North Sea, including dumping sites. Results indicated high amounts of litter at dumping site ZBO, which is likely to be caused by sedimentation processes, dumping activities and/or illegal dumping of larger litter items.
The research on anti-fouling products, their use and dispersion, needs to be continued and where necessary extended.	Concentrations of tributyltin as well as booster biocides were measured at the dumping sites. At multiple sampling locations, TBT values exceeded the proposed environmental assessment criterion while values for

	booster biocides irgarol and to lesser extent diuron exceeded the risk characterisation ratio. This indicates a potential environmental risk and a close-follow-up is indispensable in future monitoring.
A large scale sediment sampling campaign needs to be setup, inclusive the checking towards actualisation of the sampling locations.	Has been carried and the results including the new sampling sites have been reported in BOVA ENVIRO+ NV (2020).
Based on the analysis results of the large scale sampling campaign (carried out in 2018), investigations should be pursued to check if an actualisation of the SQC is needed.	It was concluded that clearer guidance is needed on method and sampling requirements for sediment analysis before dredging and that an update of the contaminant list would be advisable. This is included in a recommendation for 2022-2026.

2.2. New recommendations (2022-2026)

1. The new dumping site ZBW will be put into use and will be added to the monitoring programs.

2. An alternative for the dumping site S1 will be identified within the search area defined in the MRP 2020-2026.

3. Towards the next MRP 2026-2034, new search areas for dumping of dredged material will be defined where necessary, together with the inclusion of ZBW as dumping site.

4. The monitoring and evaluation of indicators relevant for the dumping of dredged material for the MSFD - Good Environmental Status will be continued and if needed adapted if new obligations for MSFD are defined.

5. Based on the litter baseline of 2013-2019, a more in depth study on seafloor litter hotspots is advised. Presence of large litter items (>1 m) at dumping site ZBO will be investigated. An in-depth study on litter accumulation, quantifying sources and fluxes and proposing mitigation actions is recommended, but this falls outside the scope of routine monitoring and requires additional project-based funding.

6. The collection of necessary ecological, chemical, hydrodynamical and sedimentological data for the basic research on the effect of dumping of dredged material will be continued and if necessary optimised in function of policy choices.

7. It should be examined whether the Sediment Quality Criteria (SQC) need to be updated or expanded for the next permit period. This includes defining analytical requirements such as the minimum required detection limit of the analytical method as well as recommendations for sampling design and an update of the contaminant list.

8. Sediment Profile Imaging (SPI) – monitoring was in detail performed on S1 (and S2) as a test, which will be repeated and performed for the other dumping sites (if practical possible). Such a detailed survey is needed to have a better insight in the local spatial variation in sedimentological (and ecological) characteristics in a quick way, as we observe quite some variation in those characteristics at the dumping sites. Such information cannot be obtained by grab sampling only.

3. Dredging and dumping

To conserve the maritime access channels and to maintain the depth in the Belgian coastal harbours dredging is needed in order to guarantee safe maritime transport. This type of dredging is called maintenance dredging. Most of the dredged material is being dumped at sea except when the dredged material is contaminated or when the quality is suitable for beach nourishment. The last use is called beneficial use of dredged material.

3.1 Dredging activities

Since 2008, dredging years are following calendar years and since 2006 a distinction is being made between permits for maintenance dredging (validity 2 years) and permits for capital dredging (these permits are granted for the period of working). The areas to be dredged are divided in accordance with the target depth which is defined in function of the expected vessel types and their maximum draught.

The use of certain dredging technique is dependent upon the site, the hydrodynamic and meteorological circumstances and the nature of the sediment to be dredged. Evaluation is being made on the basis of economical, ecological and technical criteria. In Belgium most commonly, trailing suction hopper dredgers are used with a hopper capacity from 5000 to 10000 m³.

In the access channels and Flemish harbours, maintenance dredging is virtually continuous throughout the year. Maintenance dredging in fishing harbours and marinas is taking place before and just after the coastal tourist period. A major port - and its connected access channels - with a diversity of customers may need to carry out a capital project every few years to accommodate changes in the patterns of trade and growth in the size of the vessels to be accommodated.

During the execution of (maintenance) dredging works, marine litter is currently considered. The dredged litter is if possible removed from the hopper and stored in a container on board for further sorting and treatment.

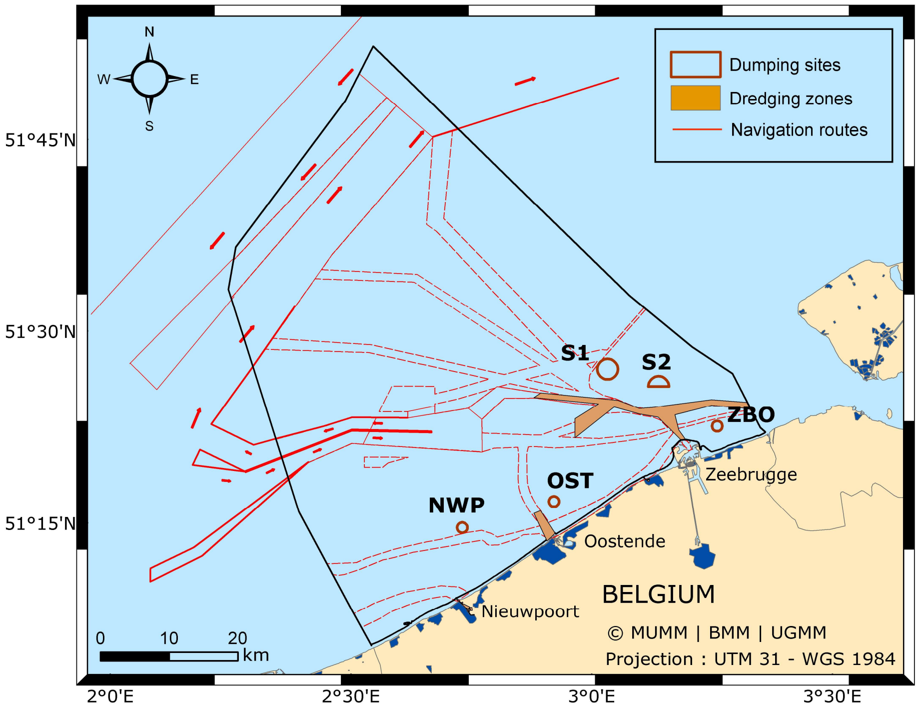

Figure 3.1: Dumping sites in the Belgian part of the North Sea.

3.2 Dumping activities

3.2.1. Quantities permitted

In the former licensing period 1 January 2017 – 31 December 2021 for maintenance dredging were granted to the Maritime Access Division as well as three permits to the Agency for Maritime and Coastal Services. The maximum and average attributed quantities which may be dumped at sea per year and per dumping area are given in Tables 3.1 and 3.2. The location of the dumping sites is shown in Figure 3.1. It should be noted that the permit holder is requested to not exceed the average quantities.

One permit (BS/2020/01) has been granted to the Maritime Access Division for capital dredging in the harbour of Ostend: valid from 13th of January 2020 till 31st of December 2020; 44.000 m³ silt and 120.000 m³ sand, dumped at dumping place Ost.

Table 3.1: Permits for the Maritime Access Division (AMT).

Permit reference	Dredging site	Type dredging	Dumping site	Yearly permitted quantities (TDM)	
				average	maximum
M.B. ref. BS/2011/01	• Scheur West • Scheur Oost • Pas van het Zand, CDNB en Voorhaven Zeebrugge	maintenance	S1	2,300,000 2,300,000 6,400,000	2,800,000 2,800,000 7,150,000
		Total :		**11,000,000**	**12,750,000**
M.B. ref. BS/2011/02	• Scheur West • Scheur Oost • Pas van het Zand, CDNB en Voorhaven Zeebrugge	maintenance	S2	500,000 375,000 2,000,000	600,000 450,000 2,400,000
		Total :		**2,875,000**	**3,450,000**
M.B. ref. BS/2011/03	• Toegangsgeulen Oostende (Stroombankkil, ingangsgeul) • Haven Oostende	maintenance	OST	600,000 500,000	900,000 700,000
		Total :		**1,100,000**	**1,600,000**
M.B. ref. BS/2011/04	• CDNB Zeebrugge • Haven en Voorhaven Zeebrugge	maintenance	ZBO	3,900,000 2,100,000	5,500,000 3,150,000
		Total :		**6,000,000**	**8,650,000**
		GRAND TOTAL		**21,045,000**	**26,550,000**

Table 3.2: Permits for the Agency for Maritime and Coastal Services.

Permit reference	Dredging site	Type dreding	Dumping site	Yearly permitted quantities (TDM except when indicated)	
				average	maximum
MB ref. BS/2016/05	* Jachthaven van Oostende – RYCO * Jachthaven van Oostende – Montgomery dok	maintenance	OST	15.000 20.000	25.000 35.000
		Total :		**35.000**	**60.000**
MB ref. BS/2016/06	* Vaargeul Blankenberge * Vlotdok Blankenberge * Spuikom te Blankenberge	maintenance	ZBO	50.000 20.000 50.000	80.000 40.000 80.000
		Total :		**120.000**	**200.000**
MB ref. BS/2016/ Nieuwpoort	* Toegangsgeul Nieuwpoort * Vaargeul en havengeul te Nieuwpoort * Oude Vlotkom te Nieuwpoort * Nieuwe jachthaven te Nieuwpoort * Novus Portus te Nieuwpoort	maintenance	NWP	60.000 50.000 25.000 30.000 30.000	100.000 80.000 50.000 55.000 55.000
		Total :		**195.000**	**340.000**
MB ref. BS/2016/ 07	* Oude Vissershaven Zeebrugge	maintenance	ZBO	50.000	100.000
		Total :		**50.000**	**100.000**
MB ref. BS/2020/01	Haven Oostende	capital	OST		
		Total			**164000 m³**
GRAND TOTAL (TDM) **GRAND TOTAL (m³)**				**400.000**	**700.000** **164000 m³**

3.2.2. Quantities dumped

Since 2007 dredging years are following calendar years. Table 3.3 gives an overview of the quantities dumped at sea since 1991 till March 2008 to keep historical data. It should also be noted that the amounts mentioned in the table are being used for the yearly OSPAR reporting of dumped dredged material, also for continuation in former reporting years. Table 3.4 gives the overview of the quantities of maintenance dredged material dumped yearly since 2007.

The maps in appendix 1 give a visual image of the maintenance dredging and dumping intensity during the period 2016 to 2020. The dredging intensities give an indication of the rate of sedimentation, while the dumping intensities show where most of the dredged material is being dumped over the surface of the dumping site. Both, dumping and dredging intensity maps are being used for validation of the mathematical models and for defining monitoring stations.

3.2.3. Beneficial use

To keep the access channel to Blankenberge harbour open, maintenance dredging on a regular basis is needed. Wind and current patterns cause a rapid influx of sand from the nearby beaches and a sand plate is being built up. As a consequence of this, the chemical and morphological qualities of this sand are very good. Contamination is virtually non-existent. Within the environmental legislation of the Flemish Region, re-use of dredged material as soil is possible, providing a specific certificate is delivered. Table 3.5 gives an overview of the quantities of dredged material from the access channel to Blankenberge used beneficially to reinforce coastal defence on the nearby beaches.

Table 3.3: Quantities of dredged material dumped since 1991.

Quantities dumped in wet tonnes(*)								
period	**S1**	**S2**	**ZBO**	**OST**	**NWP**	**R4 (**)**	**S3 (**)**	**Total**
April 1991 - March 1992	14,176,222	7,426,064	10,625,173	4,416,386				36,643,845
April 1992 - March 1993	13,590,355	5,681,086	10,901,837	3,346,165				33,519,443
April 1993 - March 1994	12,617,457	5,500,173	10,952,205	3,614,626				32,684,461
April 1994 - March 1995	15,705,346	2,724,157	8,592,891	3,286,965				30,309,359
April 1995 - March 1996	14,308,502	2,626,731	8,432,349	4,165,995				29,533,577
April 1996 - March 1997	14,496,128	1,653,382	7,609,627	2,763,054				26,522,191
Quantities dumped in tonnes dry matter (*)								
maintenance								
capital								
period	**S1**	**S2**	**ZBO**	**OST**	**NWP**	**R4**	**S3**	**Total**
April 1997 - March 1998	6,045,581	1,563,485	6,593,905	745,147				14,948,118
April 1998 - March 1999	7,455,619	482,108	2,976,919	467,107				11,381,753
April 1999 - March 2000	2,885,801	89,556	3,189,077	591,605				6,756,039
	6,187,601	41,583						6,229,184
April 2000 - March 2001	1,684,517	784,343	4,971,782	559,332		310,670	51,150	8,361,794
	3,873,444	614,657						4,488,101
April 2001 - March 2002	2,031,147	329,798	2,623,069	565,938				5,549,952
	2,527,392							2,527,392
April 2002 - March 2003	3,314,115	858,607	2,311,650	491,217	289,949			7,265,538
	2,413,760	208,885	1,369,939					3,992,584
April 2003 – March 2004	5,246306	716,427	3,126,392	646,276	142,420			9,877,821
	829,486	24,896	447,219					1,301,601
April 2004 – March 2005	1,826,561	1,826,033	3,003,397	464,307	71,928			7,192,226
April 2005 – March 2006	3,017,123	1,234,640	2,973,545	599,905				7,890,077
April 2006 – March 2007	3,791,724	505,644	2,394,828	819,665	178,269			7,690,130
	7,930,966	90,673	401,944					8,423,583
April 2007 – March 2008	5,769,680	1,266,266	2,361,012	428,839	201,581			10,027,378
	545,907	369,804		335,283				1,250,994

(*) Before April 1997, the manual "bucket" method was used to evaluate the quantity of dredged material on board a ship. Since April 1997, an automatic measurement device is used which allows directly evaluating the quantity of dry material on board ships. Comparison between both systems is not possible.
(**) Closed for dumping since end 2004

Table 3.4: Quantities of maintenance dredged material dumped at sea per calendar year (in TDM).

Year	S1	S2	ZBO	OST	NWP	Total
2007	5.592.676	127.704	2.219.780	460.167	118.100	**8.518.427**
2008	4.589.589	80.014	4.667.225	864.863	103.541	**10.305.232**
2009	6.144.522	1.591.871	3.776.038	241.544	156.456	**11.910.431**
2010	3.642.577	2.598.212	3.342.526	304.235	179.186	**10.066.736**
2011	5.290.142	2.946.850	2.062.762	562.690	64.234	**10.926.678**
2012	4.320.751	2.650.587	2.843.505	359.997	175.121	**10.349.961**
2013	5.988.596	1.969.370	3.021.397	654.488	211.722	**11.845.573**
2014	3.806.194	121.361	4.226.341	407.767	121.361	**8.683.024**
2015	5.538.995	2.913.203	3.945.216	504.944	162.128	**13.064.486**
2016	5.658.408	2.764.075	3.185.295	1.196.719	177.248	**12.981.745**
2017	5.690.034	1.983.285	2.832.670	284.015	111.235	**10.901.239**
2018	4.192.492	1.686.373	2.759.644	599.360	214.675	**9.452.544**
2019	4.890.011	1.924.513	2.164.986	416.630	230.638	**9.626.778**
2020	6.376.343	1.417.255	3.678.058	742.105	235.776	**12.449.537**

Table 3.5: Beneficial use of dredged material.

Period	Beneficially used dredged material (m^3)
November 2007 – February 2008	69.526
May 2008 – June 2008	18.661
November 2008 – December 2008	30.884
April 2009	9.588
November 2009 – January 2010	21.354
October 2010 – October 2011	22828
2012	148.757
2013	96.924
2014	155.166
2015	67.848
2016	121.671
2017	48.591
2018	79.530
2019	0
2020	94.478
Total	**144.013**

4. Physical aspects related to dredging and dumping operations

There are plenty examples of human activities in coastal environments that affect the physical dynamics or conditions of the water column, the benthic boundary layer and the seafloor, amongst which dredging and dumping. Identifying changes that are not natural requires measuring or modelling the status and trends in dynamic coastal environments. As monitoring all aspects is impossible due to the range of variables and driving processes, indicators are used that characterize ecosystems and that are cost effective, reliable, easy to monitor or to model and that predict changes that can be averted by ecosystem-based management (Dale & Beyeler 2001; Crowder & Norse 2008; Heink & Kowarik 2010; Burgass et al. 2017). For the MSFD descriptors 6 (seafloor integrity) and 7 (hydrographical conditions), there are yet no well-established monitoring programmes of physical indicators that allow assessing human-induced changes in the nature and dynamics of physical parameters of the water column and seabed. The following three indicators characterize to a part the pelagic, benthic and the interface between both: Suspended particulate matter (SPM) concentration and composition for the pelagic zone, seabed/habitat type for the benthic zone and bed shear stress as the interactive force between both zones (Fettweis et al. 2020).

Marine ecosystems comprise SPM whose concentration and heterogeneous composition varies both temporally and spatially as a result of multiple natural processes and human activities. In coastal regions, flocculation processes combine these particles into biomineral flocs that contain inorganic matter such as cohesive (e.g., clay) and non-cohesive minerals (e.g., quartz, carbonates) together with particulate organic matter (POM). The POM itself consists of living (e.g., phytoplankton, bacteria) and non-living components. In nearshore areas the concentration of SPM is often high and the flocs are dominated by minerals and the SPM may also contain sand or silt grains. Towards the offshore the SPM concentration decreases, while its POM content increases. The POM can be discriminated between a labile and a more refractory fraction. The labile or fresh part of the POM is subject to seasonal variations, as it is produced by primary production. The refractory POM is incorporated in the mineral fraction where it is particularly bound to the clay minerals (Mayer 1994), entering the water column through resuspension of sediments.

Changes in coastal ecosystems are often correlated with changes in SPM concentration and thus also with the POM content of the SPM (e.g. May et al. 2003; Capuzzo et al. 2015). During the last decades, the North Sea has been subject to changes in SPM concentration and composition due to a decrease in phytoplankton production and changes in community structures (Capuzzo et al. 2018; Nohe et al. 2020); the shift in chlorophyll a phenology (Desmit et al. 2020); the imbalance in the biogeochemical cycles of nutrients (Rousseau et al. 2006, Desmit et al. 2018); the impact of major construction works (Van Maren et al. 2015) and the dumping of dredged material (Fettweis et al. 2009; 2011; 2016; Houziaux et al. 2011).

The objective of the study is to focus on both the composition and the concentration of SPM and its use as an indicator for ecosystem changes. We have developed protocols that allow to quantify the uncertainty of the sensor-based measurements of SPM concentration, a key property necessary to calculate statistically significant trends. Further, based on the POM composition of SPM derived from water samples, a mechanistic modelling approach has been applied that differentiate the POM as fresh and mineral-associated. With the data-model syntheses we aim at consolidating robust relationships that can be applied to SPM concentrations derived from other sources, like satellite or high-resolution in-situ time series. The final syntheses products yield spatio-temporal compositional changes of the SPM, with respect to POM, both on large scales and for anomalous events. This will greatly facilitate the monitoring of water quality parameters along the gradient from a domination of mineral-associated POM towards a domination of fresh POM.

This chapter is structured in four parts. First, the SPM concentration and composition measurements are described. Secondly, the uncertainties of these measurements are evaluated. Thirdly, fresh and mineral-associated POM concentrations in the particulate organic carbon (POC), nitrogen (PON) and Transparent Exopolymer Particles (TEP) fractions are differentiated through the use and refinement of the POM-SPM modelling approach of Schartau et al. (2019). Fourthly, the model is applied to estimate fresh and mineral-associated POC, PON and TEP from satellite SPM measurements to discuss the temporal and geographical variability of SPM concentration and composition on the Belgian continental shelf (BCS).

4.1. Measurements of SPM concentration, floc size and composition

The SPM occurs as flocs with highly heterogeneous composition (Droppo et al. 1997, Droppo 2001, Maggi 2013, Shen et al. 2018) and highly variable concentration. Flocs can be regarded as individual microecosystems with autonomous and interactive chemical, physical, and biological reactions and processes activating within the floc matrices. They contain three major groups of heterogeneous components, including inorganic components that contain cohesive (e.g., clay) and non-cohesive minerals (e.g., sand, quartz, carbonates), biological components that include living (e.g., phytoplankton, bacterial) and non-living components (e.g., TEP) and other organic compounds. The heterogeneous composition of flocs affects their structure, porosity, density, and size, and, as such, ultimately the SPM dynamics in water environments.

The composition of the SPM was measured from water samples and chemical analysis. We have used TEP, POC and PON to characterize the POM of the SPM.

4.1.1. Long-term SPM concentration measurements

0ptical and acoustic sensors

Long-term and high frequency data series of SPM concentrations are typically collected indirectly with autonomous sensors that measure either the optical beam attenuation as a percentage of light transmission (Moody et al. 1987; Spinrad et al. 1989; Agrawal & Pottsmith 2000), the back- or sidescatter intensity of light in volt or factory calibrated turbidity units, or the acoustic backscatter in counts or volts (Thorne & Hanes 2002; Downing 2006; Rai & Kumar 2015). In addition to these sensors, gravimetric measurements of filtered water samples are generally used as ground truth reference (e.g. Neukermans et al. 2012; Röttgers et al. 2014; Fettweis et al. 2019). The combination of indirect and reference measurements requires two main calibration steps (sensor and model parameter calibration) at different moments during the workflow to extract reliable and homogeneous SPM concentration. These calibration steps are essential for relating changes in calibration constants (both sensor and model parameter constants) to either sensor degradation or to natural variability in SPM inherent properties. We have made a detailed analysis of the uncertainty associated with measurements of SPMC in order to increase the applicability of an indicator based on SPM concentration (Chapalain et al. 2019; Fettweis et al. 2019).

Turbidity refers to the optical water cloudiness caused by suspended particles and dissolved substances, which scatter and absorb light (Downing 2005; Ziegler 2003; Gray & Gartner 2009). Turbidity does not have a SI unit, is not uniquely defined and depends strongly on the applied protocols. It is thus an arbitrary unit that is incomparable to measurements taken at other times and places or with different turbidity meters, which diminishes the comparability of turbidity data (Downing 2006). There are two international recognized methodologies: the ISO Method 7027 (ISO 1999) and the American EPA Method 180.1 (EPA 1993). Both estimate turbidity, for the ISO method it is in formazine Nephelometric Units (FNU), and for the EPA method in Nephelometric Turbidity Units (NTU), respectively, and in both methods, the optical sensor to be used is a nephelometer that must measure side-scattered light at 90°. The strengths of the ISO method include the use of a stable monochromatic near infrared light source of 860 nm with low absorbance interference with samples, which is critical in reducing the impact of particulate and coloured dissolved organic matter absorption and in having low amounts of stray light (Sadar 1999). Sensors designed according to the ISO definition of turbidity provide thus a better basis for the comparability of measurements than those designed following the EPA specification (Barter & Deas 2003; Nechad et al. 2009; Bright et al. 2018).

The relationship between the signal from an optical backscatter sensors (OBS) or sidescatter sensors (nephelometer) and the SPMC is almost linear as long as the sensor is not deployed in highly concentrated waters (Downing 2006), and the simplest model is a linear regression model. The same holds for single point acoustical sensors (ADV) or for the first bin of a profiling acoustical sensor, where the target volume is very close to the sensors. As far as SPM concentration are lower than several g/l, a direct empirical relationship can be built such as $log_{10}(SPM\ concentration) \sim S_v$, where the acoustic volume backscatter strength S_v can be related to the signal/noise ratio (Fugate & Friedrichs 2002; Voulgaris & Meyer 2004; Verney et al. 2007; Ha et al. 2009; Salehi & Strom 2011).

For profiling acoustic sensors, the sonar equation should be corrected for the signal loss along the acoustic path. Close to the transducer, the acoustic signal has to be corrected for near-field effects (Downing et al. 1994) and for ringing effects that may affect the first bins, in particular when blank distance is set too small in the configuration parameters. Corresponding data cannot be corrected and should be discarded (Muste et al. 2006). A formulation for the water absorption coefficient was proposed by e.g. Francois & Garrison (1982a, b) and later simplified by Ainslie & McColm (1998), who showed that their result did not differ from the original equation more than the accuracy error. The sonar equation yields the so-called water-corrected backscatter, which is a property of the suspension at all locations along the acoustic path. Subsequent processing depends on the SPM concentration. In case of moderately turbid environment, i.e. lower than O(100) mg/l and depending on the acoustic frequency, sound attenuation by SPM is usually neglected as it is one or two orders of magnitude lower than the water absorption coefficient (Ha et al. 2011). SPMC is then either determined by applying an appropriate calibration, similar to single point optical sensors, or by a theoretical acoustic model. In the latter case, physical properties of the transducer and of the SPM must be exactly known, which are rarely available. If SPMC exceeds several 100 mg/l, sediment absorption should be considered. However, this term is a function of the SPM concentration, which is also the unknown of the calculation. The inversion problem is solved by iterative methods (Thorne et al. 1994; Holdaway et al. 1999). This technique is efficient but requires assumption or knowledge about transducer physical properties, SPM characteristics (size, density) and is based on the choice of an acoustic model adapted to the observed SPM, and may in some specific case exponentially propagate uncertainties and fail to estimate SPM concentration (Becker et al. 2013). Theoretical acoustic models were originally built to simulate the physical interactions between particles and the acoustic signal (Sheng & Hay 1988, Medwin & Clay 1998) and were applied to sand particles in suspensions (Thorne & Hanes 2002). These models were later adapted to represent low density SPM flocs (Stanton 1989; MacDonald et al. 2013; Thorne et al. 2014) and were shown to correctly estimate SPM concentration in estuarine environments (Sahin et al., 2017). Differences between models mainly appear in the methodology to calculate the total scattering and backscattering cross section as well as the compressibility of flocs and their ability to interact with sound.

Remote sensing measurements

Surface SPM concentration have been derived from the Ocean and Land Colour Instrument (OLCI). OLCI is a multispectral radiometer carried on board Sentinel-3A (launched in 2016) and B (launched in 2018) with 21 bands on the 400-1200 nm spectral range and a spatial resolution of 300 m. The two satellites provide a daily revisit time over the southern North Sea. Sentinel-3/OLCI baseline water products (L2-WFR) were retrieved from the Copernicus

Online Data Access (CODA) service hosted by EUMETSAT (coda.eumetsat.int). The baseline products were processed with IPF-OL-2 version 06.13 (EUMETSAT 2019) with standard masking applied, i.e. excluding INVALID, LAND, CLOUD, CLOUD_AMBIGUOUS, CLOUD_MARGIN pixels. Additionally, a custom quality control was applied to remove outlier pixels with a spectrally flat signal. The SPM product was generated by an artificial neural network as a multiple non-linear regression technique to deal with the optically complex waters in the study area. The artificial neural network, originally developed by Doerffer & Schiller (2007), was updated to become the Case 2 Regional (C2RCC) processor suitable for Sentinel-3 (EUMETSAT, 2019).

SPM particle size measurements

Complementary to SPM concentration measurements, particle size measurements are essential to evaluate the floc size dynamics and the SPM settling fluxes. In coastal systems, particle size distribution measurements are often conducted from laser-based or camera-based systems. The latter is based on prototypes, while the former is the mostly used, with commercially available systems (e.g. LISST instruments). The LISST 100 instrument has become a standard measuring instrument for particle size spectra and volume concentrations. LISST measurements consist in emitting a laser beam which is scattered by particles at small forward angles and detected by ring detectors. The particle size distribution (PSD) is then back-calculated using an optical model. Two models are available. The first one is based on the Mie theory assuming that particles have a spherical shape while the second one is based on random shaped particles (Agrawal et al. 2008). The volume concentration is estimated using the particle size distribution together with an empirical volume calibration constant that is specific to spherical or random shaped particles.

4.1.2 SPM and POM concentration from water samples

At every sampling occasion, three subsamples for SPM concentration were taken and filtered on board using pre-combusted (450°C, 24 hours) and pre-weighted 47mm GF/C filters. The filters were rinsed with MilliQ water and immediately stored at -20°C, before being dried during 24 hours at 50°C and weighted to obtain the concentration.

POM was determined through POC, PON and TEP measurements. The samples for POC and PON were filtered on board using 25mm glass fibre filters, stored immediately at -20°C, before being analysed using a Thermo Finnigan Flash EA1112 elemental analyser (for details see Ehrhardt and Koeve, 1999). and analysed in the laboratory through catalytic oxidation and gas chromatography using a FLASH EA 1112 – Element analyser.

The method for TEP analysis follows the one described in Nosaka et al. (2017). This method is, as many other semi-quantitative methods, based on Aldredge et al. (1993) and Passow & Aldredge (1995). Three subsamples for TEP concentration were taken and filtered using 25mm 0.4 µm polycarbonate filters with low under-pressure. The filters were coloured immediately after filtration with Alcian blue, rinsed with MilliQ water and stored at -20°C. The stained particles are related to a weight equivalent for the anion density of TEP and standardized using xanthan gum (Passow & Alldredge, 1995; Passow, 2002). The units for TEP are expressed as mg xanthan gum equivalents per litre (mg XG eq./l).

4.2 Uncertainty of SPM measurements

4.2.1 SPM concentration from sensors

The overall error of the SPM concentration data set consists of random errors that lead to uncertainties of individual SPM concentrations but approximate the accurate value with increasing amount of data, and of systematic errors (biases) that lead to an average over- or underestimation of all data. Some errors can be detected, and to some extent corrected, whereas, others are inherently associated with the applied technologies and its interference with the environment and remain spurious and difficult to quantify or to control. The first types of errors are related to the sensors, sampling and lab protocols or the modelling techniques, while the latter are mainly related to systematic, often gradually changing natural variability in SPM inherent properties.

Long-term observations of SPM concentration are the result of a complex ladder of operations that involve field, laboratory and modelling methods. Each step contributes its own random and systematic errors to the overall uncertainties of the sensor SPM concentration. Systematic errors related to the functioning of the sensors, the environment, the collection and processing of calibration samples and faulty human operations are detectable and sometimes correctable. As long as protocols for sample analysis and sensor calibration are carefully followed, uncertainties can be confined within ±5%, otherwise they may reach up to ±20%. Biofouling may add a further bias of 100% (positive for optical, negative for acoustical sensors), and their detection generally leads to a loss of data. A good understanding of the processes that are causing changes in SPMC and particle inherent properties (size, shape, density and composition) is required in order to estimate their importance and to possibly rescale the sensor data to some reference particle properties. We will discuss the composition of the SPM in chapter 4.3. Variations in these properties may result in over- or underestimation of the SPM concentration by up to a factor 2 or more. Based on the uncertainties, listed in table 3 of Fettweis et al. (2019), one can achieve random errors below 25% and biases below 40% only with substantial efforts in technologies that indicate the changes in inherent particle properties.

Acoustical and optical sensors require both the conversion of the sensor output (after sensor calibration) to a mass concentration. This is done by relating the sensor output to a reference SPM concentration, which is preferably the sample SPM concentration. The choice of the regression method, the dependent and independent variable, and the error associated with the reference SPM concentration determines the coefficient of determination Using the R^2 and the normalized turbidity/dB the uncertainty of the sensor derived SPM concentration in the calibration range and outside of it can be quantified (Fettweis et al. 2019). The model shows that the Robust fit (iteratively reweighted least squares regression) and the Eigenvalue regression have less prediction bias than the Theil-Sen estimator and the ordinary least square regression. This bias is not an issue for $R^2>0.9$ and remains below 10%, but it becomes significant for lower R^2 and can amount to 30%. Short-term variabilities in the model-regressions generally show up as random noise limiting the R^2 of the calibration data set, but the extrapolation of the regression parameters to longer periods or larger areas may introduce biases of more than 50%.

4.2.2 SPM particle size

The different methods that are used to measure in situ particle size distributions (PSD) may not give the same results. A PSD measured by a LISST will differ from the one measured by a digital camera (e.g., Mikkelsen et al. 2005). The uncertainties associated with a measuring technique are related to the characteristics of the particles occurring in nature (Mikkelsen et al. 2006; Andrews et al. 2010; Davies et al. 2012; Graham et al. 2012), and to the measuring principle itself (Mikkelsen et al. 2005; Goossens 2008). Generally, camera systems cannot resolve the fine particles smaller than 10 µm, while LISST has a limited size range for the fine and the very large particles.

Uncertainties using LISST 100C detectors may arise to non-spherical flocs (such as complex aggregates), to floc sizes exceeding the instrument range, to a too high or too low SPM concentration or to stratification of the water column (Chapalain et al. 2019). The effect of the floc shape on LISST measurements is complex to estimate, and can only be evaluated through the choice of the inversion model, i.e. spherical or random shape, in the LISST postprocessing. The main consequence of the model choice for a given distribution is a shift towards smaller class sizes, without changing significantly the spectrum shape. The LISST provides reliable measurements along an operational concentration or turbidity range. In low SPMC environments (i.e. transmission above 90% or SPMC below 5 mg/l), LISST measurements are strongly dependent on the background quality, and instabilities in raw signal measurements can produce artefacts and bad detection particles, mainly in the largest size classes. In high concentration ranges, multiple scattering occurs and can generate additional unrealistic signal in the extreme size classes. This upper limit corresponds to SPMC values of several 100 mg/l, i.e. far lower than the saturation level. The last source of uncertainty regarding LISST measurement is certainly the most critical in coastal waters, as related to density stratification. This effect known as the Schlieren Effect (Styles 2006) is caused by the deviation of the laser beam due to salinity gradients and related changes in refraction indices and increases the signal recorded by the inner detectors and artificially increases the volume concentration in the largest size classes.

Out of range particles are influencing the size distribution of the LISST. For example, particles smaller than the size range of the LISST affect the entire PSD (Andrews et al. 2010; Graham et al. 2012). A rising tail in the lowest size classes of the LISST is frequently observed in our data and is interpreted as an indication of the presence of very fine particles rather than providing a correct number. Particles exceeding the LISST size range of 500 µm also contaminate the PSD. Davies et al. (2012) reported that large out of range particles increase the volume concentration of particles in multiple size classes in the range between 250 and 400 µm and in the smaller size classes and recommended to interpret the PSD with care in case particles outside the size range may potentially occur. The importance of these spurious results depends on the number of large particles in the distribution (Davies et al. 2012). Nowadays, there is still no good way for correcting PSDs for these spurious data, but we should be aware that the very large (macroflocs) and the very small particles (primary particles) maybe under-represented or over-represented in the in situ LISST derived PSDs. Despite the uncertainties and limitations of the LISST-100C, it is well suited to collect long-time series of PSD autonomously.

Even if the size distributions of flocs are well resolved, there are still uncertainties involved in the estimation of the density and the settling velocity. To investigate settling dynamics, estimates of floc size and floc density are required. In literature, the fractal theory is commonly used for relating floc size and floc excess density (Kranenburg 1994; Chen & Eisma 1995; Dyer & Manning 1999). Small changes in fractal dimension may induce large changes in the settling velocity. A sensitivity analysis of the fractal approach to model floc density has been described in Chapalain et al. (2019). In the fractal model, primary particles are characterized by a unique size and density and it is generally assumed that a floc only includes mineral particles whereas particulate organic matter (OM) is not considered (Khelifa & Hill 2006; Maggi 2013). However, these assumptions must be questioned as the primary particle size may vary spatially and temporally within the same area (Fettweis 2008; Maggi 2013), as biological or biomineral aggregates are ubiquitous in marine environments (e.g. Maggi 2009; Fettweis & Lee 2017; Shen et al. 2018), and as the density of primary particles may change with changes in the composition of the SPM (Markussen & Andersen 2013). Our analysis also confirms that the application of the fractal approach, i.e. flocs are built from a unique type of primary particles characterized by constant size and density, has limitations. Also, depending on the history of flocs (eroded from beds, dynamically formed in the water column), flocs of similar sizes might be characterized by different densities (Smith & Friedrichs 2011). Fall et al. (2018) for example have demonstrated that there is not necessary a unique relation between floc size and excess density but that the fractal approach could be valid for the large floc sub-population (macroflocs).

4.2.3 SPM, POC, PON and TEP concentration from water samples

The uncertainty of SPM concentration from the filters is expressed as the RMSE of the triplicates divided by the mean value. The uncertainty decreases with increasing concentration from 8.5% (SPM concentration < 5 mg/l) to 6.7% (<10 mg/), 3.5% (10–50 mg/l) and 2.1% (>100 mg/l) and represent the random error related to the lack of precision during filtrations. Especially in clearer water, systematic errors due to the offset by salt or other errors become much larger than the random errors (Neukermans et al., 2012; Fettweis et al., 2019). These are not included, and have been estimated based on Stavn et al. (2009) and Röttgers et al. (2014) as 1 mg/l. The analytical uncertainty for POC and PON are 12% and 18% respectively. The uncertainty for TEP is assumed to be equal to the one of POC.

4.3. Organic and inorganic composition of SPM

The inorganic and organic components of the SPM have different origins. The inorganic particles may have a detrital or biogenic origin. The detrital mineral fraction typically incorporates clays, quartz and other minerals, while biogenic inorganic particles consists of minerals such as carbonates and amorphous silicates. In the further considerations, we will only consider particulate inorganic material (PIM) as a whole. The POM is a mixture of compounds derived from marine photosynthesis or terrestrial sources. It is a combination of diverse detrital organic substances as well as of living organisms such as bacteria, phyto- and zooplankton. The POM (POM stands here for POC, PON and TEP) can be refractory or fresh. The first one has a low susceptibility and the last one a high susceptibility towards microbial degradation (Arndt et al., 2013). The fresh part of the POM (POM_f) is subject to seasonal variations, as it is produced by primary production. The refractory POM (POM_m) is incorporated in the mineral fraction where it is particularly bound to the clay minerals (Mayer, 1994; Blattmann et al., 2019), entering the water column through resuspension of sediments.

4.3.1. Modelling approach

The fresh and mineral-associated POC, PON and TEP fractions have been separated using a mechanistic modelling approach, based on Schartau et al. (2019), who considered Loss-on-Ignition (LoI) measurements for describing the POM:SPM ratio as a function of SPM concentration. The conceptual basis of the POM-SPM model is that the POM concentration can be written as the sum of the POM_f and POM_m concentrations and that POM_m is assumed to be linearly correlated with the PIM concentration by a constant proportionality factor m_{POM}:

$$POM = POM_f + POM_m = POM_f + m_{POM} PIM \tag{1}$$

The second assumption is that the seasonal built up of POM_f can be described as a saturation function of the SPM concentration with a parameter K_{POM} that is varying seasonally:

$$POM_f = \frac{K_{POM}}{\frac{K_{POM}}{SPM}+1} \tag{2}$$

The parameter K_{POM} has the same units as SPM concentration and is the second parameter of the POM-SPM model. The function reaches a saturation at high SPM concentrations where POM_f equals K_{POM}, while at low SPM concentration POM_f concentration tends to zero. This approach assumes that the production of POM_f is eventually limited by nutrients, temperature and light availability. Optimised values of K_{POM} were shown to be subject to seasonal variations, whereas values estimated for m_{POM} turned out to be fairly constant and independent of seasonal conditions (Schartau et al., 2019). According to the POM-SPM model, the POM content of SPM can be approximated by:

$$\frac{POM}{SPM} = \frac{POM_f}{SPM}\frac{1}{m_{POM}+1} + \frac{POM_m}{SPM} \tag{3}$$

A non-linear dependency between POM content and SPM concentration is obtained by including Eq. (2) in Eq. (3). The combined equation has some meaningful and desired convergence characteristics. For SPM concentration approaching zero, POM content converges to 1 (and POM_f fraction dominates POM) and for high SPM it approaches $\frac{m_{POM}}{m_{POM}+1}$ (and POM_m fraction dominates POM). Instead of using LoI data for POM content, we considered three different types of organic matter data, namely the concentrations of POC, PON, and TEP. The POM-SPM model was refined by introducing two parameters (f_1 and f_2) for every observational type Xi (POC, PON, and TEP) to the POM-SPM model (Eq. 4):

$$\frac{Xi}{SPM} = f_{1,X_i}\frac{POM_f}{SPM}\frac{1}{m_{POM}+1} + f_{2,X_i}\frac{POM_m}{SPM}\;;\;\; X_i\ [POC, PON, TEP]; \quad (4)$$

These additional parameters $f_{1,Xi}$ and $f_{2,Xi}$ represent relative proportions of X_i to POM, e.g. $f_{1,X2}$ and $f_{2,X2}$ express the ratios of fresh- and of mineral-associated PON to POM (or $f_{1,X1}$ and $f_{2,X1}$ for respective ratios of POC to POM) in units of molecular weight (g g-1) or TEP to POM in units of (g XG eq.)/(g POM), respectively. In this manner, consistent and meaningful estimates of $f_{1,Xi}$ and $f_{2,Xi}$ could be derived. Values assigned to or estimated for K_{POM} and m_{POM} should be largely independent of the observational type, no matter whether POC, PON, or TEP concentrations are considered. Overall, the refined model requires values to be assigned to four parameters (m_{POM}, K_{POM}, f_1 and f_2). Consequently, and somewhat different from the POM:SPM model, the respective fraction of X_i converges to $f_{1,Xi}$ at SPM concentration approaching zero. For high SPM concentrations the portion X_i of SPM approaches $f_{2,X_i}\frac{m_{POM}}{m_{POM}+1}$.

For model descriptions of POC:SPM and PON:SPM we considered seasonal variations and distinguished in parameter estimates accordingly, combining data from three months: winter (December, January, and February), spring (March, April, and May), summer (June, July, and August), and autumn (September, October, and November). We refer to Fettweis et al. (2021) for a detailed description of the parameter optimisation.

4.3.2. SPM, POC, PON and TEP concentration from water samples

The water sample data used here were taken in the Belgium part of the North Sea between October 2004 and until August 2020. The data set consists of hourly (or 1.5 hourly) water samples and particle size distributions collected during 125 tidal cycles (sometimes half tidal cycles) in 12 stations. The three main stations (MOW1, W05 and W08, Figure 4.1) are located along a cross-shore section that ranges from the nearshore coastal turbidity maximum (MOW1) to the offshore under complete Channel water influence (W08). W05 is located in between at the outer margin of the coastal turbidity maximum.

From 2004 until November 2018 the measured parameters were SPM, POC, PON concentrations. From March 2018 onward near surface samples were also collected. From December 2018 onward, the range of parameters was extended with TEP. The water samples were filtered on board and analysed in the laboratory to obtain the concentration of SPM, POC, PON, TEP. The Particulate Organic Matter (POM) content was determined by loss-on ignition until November 2019. The total amount of samples collected with at least SPM - POC, SPM - PON and SPM - TEP data pairs is equal to 1900, 1719 and 598, respectively. All the TEP data and about 80% of the POC and PON data are from the three main stations.

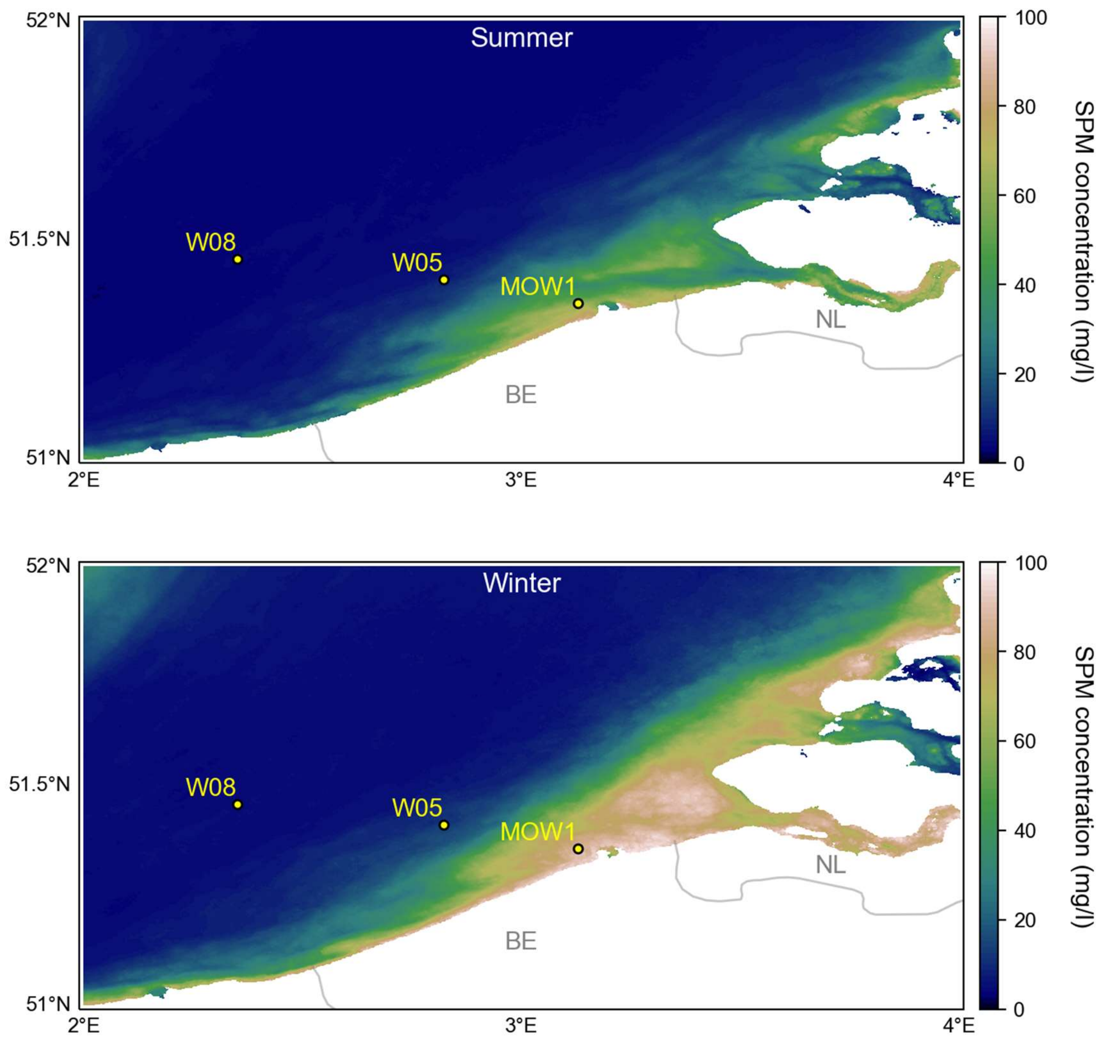

Figure 4.1: Map of sampling stations MOW1, W05 and W08 (BE: Belgium, NL: The Netherlands). The background displays the averaged near surface SPM concentrations in the Belgian coastal zone (southern North Sea) computed from satellite images taken by Sentinel-3/OLCI from April 2019 to September 2019 (above) and from November 2019 to March 2020 (below).

POC, PON and TEP content of the SPM

The POC and PON fractions of SPM as a function of SPM concentration are shown in Figure 4.2. The graphs indicate an increase of the POM content with decreasing SPM concentration, which has been documented as a characteristic feature (e.g. Eisma & Kalf 1987; Jago et al. 1994; Fettweis et al. 2006; Schartau et al. 2019). The fraction of POC incorporated in SPM varies between ~2.5% and 30% (POC), while the PON fraction is clearly lower, ranging between ~0.35% and 4%. Thus, the POC and PON content of SPM are about 4 and 28 times smaller than the POM content respectively. From about a 100 mg/l SPM concentration onward the POC and PON content reach an asymptotic value of about 2.5% and 0.35% respectively. The 10 to 15 time increase of the POC and PON content occurs over two orders of magnitude in SPM concentration and shows that SPM in the nearshore contains proportionally significantly less OM than in the offshore.

TEPs incorporate mainly organic carbon but also include fractions of organic nitrogen. Thus, measured TEP concentrations are not independent of the POC and PON measurements, which is reflected in the significant ($p<0.05$) correlation between POC and TEP (PON and TEP) in the data with an $R^2=0.59$ ($R^2=0.62$). Accordingly, the dependency between TEP and SPM concentration (Figure 4.2c) is similar to those found for POC and PON (Figures 4.2a and 4.2b).

Instead of a percentage fraction of SPM, the TEP:SPM ratio is given here in mass units (g XG eq.)/g (Figure 4.2c), because the concentration of the Alcian blue stained microgels cannot be easily related to a mass concentration e.g. of organic carbon, in the presence of resuspended mineral particles.

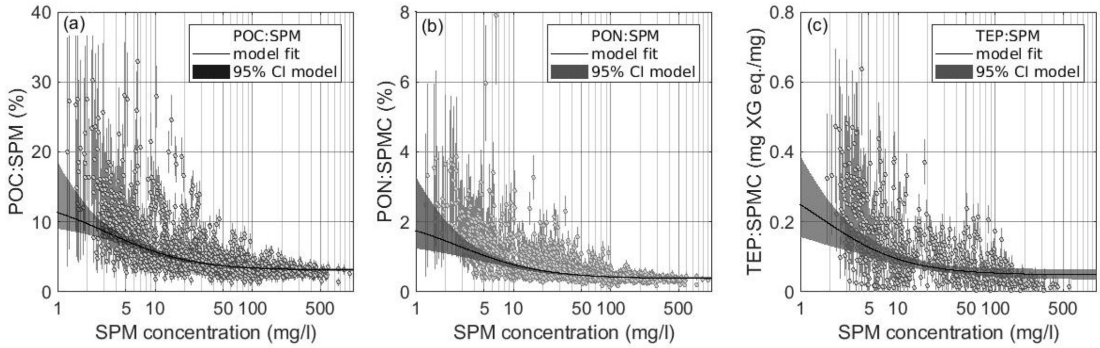

Figure 4.2: Model fit and 95% confidence intervals through all data for the POC content (left), PON content (middle) and the TEP content (right) as a function of the SPM concentration. The error bars represent the uncertainties of the measurements (see Fettweis et al., 2021). The shaded area is the 95% confidence interval of an ensemble of individual model fits, based on 100 optimisations with different, randomly sampled, data subsets.

Fresh and mineral associated POC, PON and TEP content of the SPM

The annual composite data of POC:SPM, PON:SPM and TEP:SPM, as depicted in Figure 4.2 exhibit extensive variability. For the most part, this variability can be attributed to seasonal changes. The fits of the models to annual composite data resolve and explain only differences between the different observational types. Seasonal variations have been further resolved by fitting the models to seasonally sorted data. In general, the non-linear dependency of the POC and PON content of SPM (Figure 4.3) varies in a similar way as the TEP:SPM ratio (Figure 4.4), with clearly altered seasonal signals.

At high SPM concentrations, greater than 100 mg/l, the variability remains small and temporal differences between the model solutions are indistinguishable for POC:SPM and PON:SPM. A large spread in the seasonally resolved model solutions were obtained for TEP:SPM ratio at SPM > 100 mg/l (Figure 4.4). The only noticeable difference is the lower estimate of f_2 obtained for modelling the TEP:SPM ratio at high SPM concentrations in winter. Whether this estimate is actually associated with a clear difference in the mineral-associated fraction of TEP in winter is unclear. Apart from this, the overall spread does not follow any seasonal pattern and must be attributed to larger uncertainties in the model fits of the TEP:SPM ratio. Overall, seasonal variations in the mineral-associated fractions of POC, PON, and TEP in SPM could not be identified and appear to be negligible. The small changes may rather be associated with variations in sediment types that contain variable constituents and fractions of minerals.

For SPM concentrations below ~100 mg/l, we identified clear and distinctive seasonal patterns. Our results show a correlation with season (and thus with primary production), which is most pronounced in the low-turbid data. In all cases, the seasonal changes could be well resolved (Figures 4.3 and 4.4). During the winter season the variations of the SPM's content of POC, PON, and TEP remain small for a large range of SPM concentrations, with only a small increase of respective fractions at low SPM concentrations. The general picture changes drastically for the spring period when phytoplankton blooms induce a substantial increase in the POC and PON content of the SPM, and also the TEP:SPM ratios follow this signal. At SPM concentrations of 1 mg/l, the lower end of the sample values, the POC and PON fractions of SPM are ~17% and ~2.5%, and for TEP ~0.4 (g XG eq.)/g. During spring the measured POC and PON contents of the SPM feature some high values at SPM concentration between 10 to 50 mg/l, which are not captured by the model solution and are likely caused by the high spatio-temporal variability of patches with elevated phytoplankton biomass concentrations. Still, the model's optimized solutions for spring yield highest in the production of fresh POC, PON, and TEP in this range of SPM concentrations. According to the optimized model solutions, the elevated spring values gradually decrease during summer and autumn, a trend that can hardly be recognized on the basis of the highly scattered sample data alone. The differences between the summer and autumn signals are somewhat less distinctive than their differences to spring conditions. This is because transitional months like September may include a prolonged bloom signal from summer or involve secondary bloom events due to the recurrence of elevated nutrient concentrations. The transitions from autumn to winter conditions are again well pronounced.

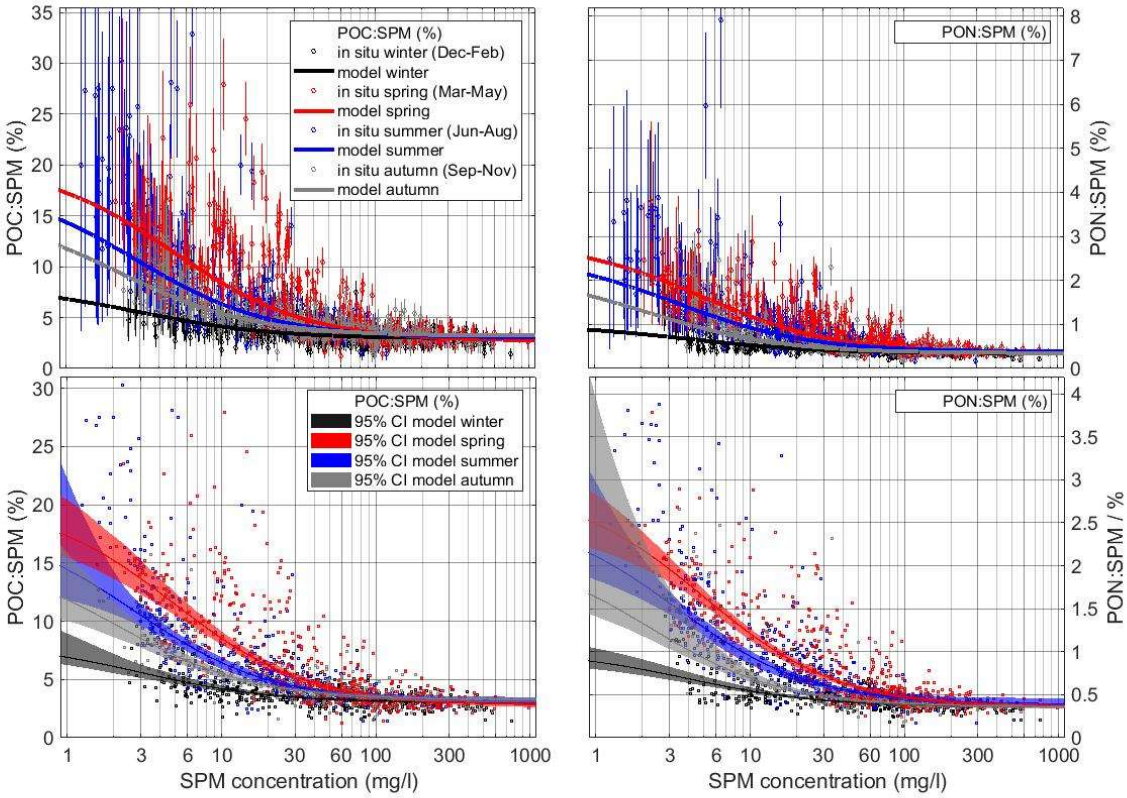

Figure 4.3: POC and PON content (in %) as a function of SPM concentration (data from 2004-2020). Top panels show the uncertainties of the data as described in Fettweis et al. (2021) and model estimates for the different seasons. Bottom panels show the 95% confidence interval of an ensemble of individual model fits, based on 100 optimizations with different, randomly sampled, data subsets.

4.3.3. Spatial and temporal variation of fresh and mineral associated POM

We used the Sentinel-3/OLCI satellite images of SPM concentration to extract the mineral-associated and fresh components of POC, PON and TEP at the water surface. Taking the model parameters for f_1, f_2, K_{POM} and m_{POM} we applied the model pixelwise to the remote sensing data. In this way we generated eight further secondary satellite products purely based on SPM surface concentration. The SPM, mineral-associated and fresh POC concentration together with the ratio between both is shown in Figure 4.5 for all the seasons along the transect that connects the three measuring stations from the coast (MOW1) with SPM concentration ≅ 80 mg/l via W05 to the offshore (W08) with ≅ 2 mg/l.

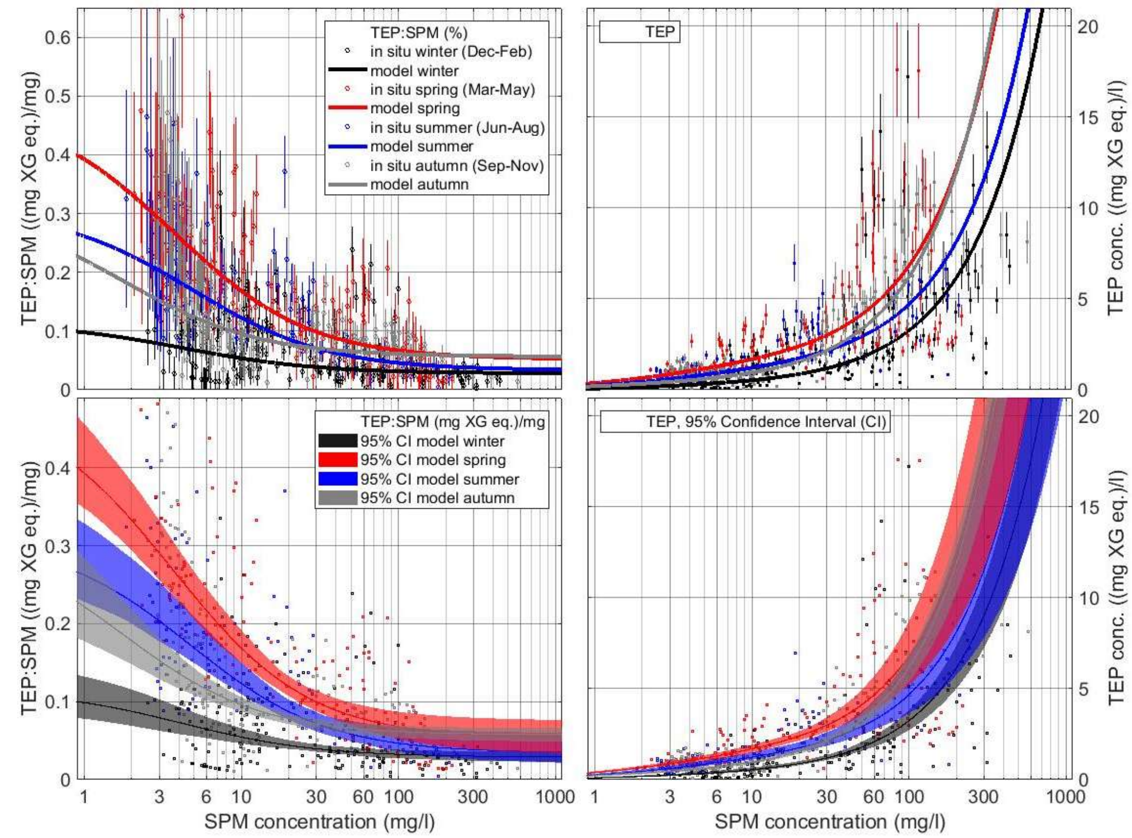

Figure 4.4: TEP content (left) and TEP concentration (right) as a function of SPM concentration. The lines are the result of the TEP-SPM model for the different seasons. The error bars represent the uncertainties of the TEP measurements, see Fettweis et al. (2021). Bottom panels show the 95% confidence interval of an ensemble of individual model fits, based on 100 optimizations with different, randomly sampled, data subsets.

The mineral-associated POC, PON and TEP follows to a large part the SPM concentration and has the same strong decrease with increasing distance from the coast. The fresh POC, PON and TEP concentrations in contrast, nearly keep their level along the entire area or transect. The major seasonal formation of fresh POC, PON and TEP thus occurs not only within the shallow coastal regions but extends along the whole area. These patterns are similar for all components of the POM and for all seasons. For the mineral-associated parts, winter has the highest and summer the lowest values except for TEP where the highest values are during the periods of spring and autumn blooms. The fresh parts are always highest in spring and lowest in winter. The ratios of fresh to mineral-associated POC, PON and TEP show that there is a narrow zone, where both fractions are about equal (ratio = 1). This may be identified as a transition zone where, seen from the land, the near coast conditions with particle dominance from the sea bed turn into open sea conditions with particles who are of pelagic origin. This transition zone is located in between MOW1 and W05 in spring and summer at a water depth around 10 to 15 m and corresponds with a surface SPM concentration of about 40 mg/l in spring and about 20 mg/l in summer. It is located more offshore in autumn where it occurs around W05 (water depth around 20 m) with about 8 mg/l SPM concentration and close to W08 in winter.

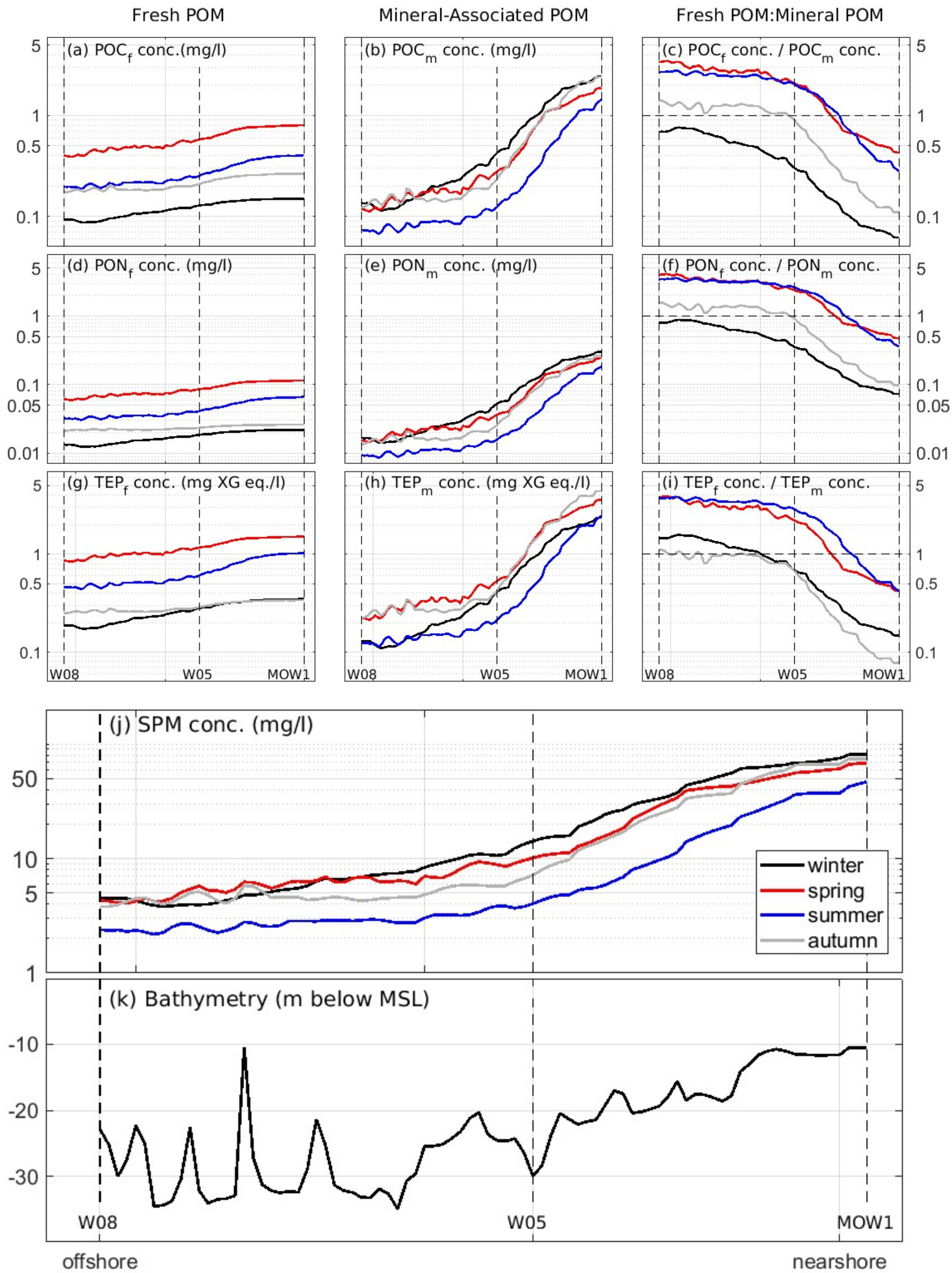

Figure 4.5: Model estimates of the nearshore (MOW1) to offshore (W08) fresh (left column) and mineral-associated (middle column) POC (a-b), PON (d-e) and TEP (g-h) concentrations calculated from the Sentinel-3/OLC derived surface SPM concentrations (j) using the model parameters. The right column shows the ratios of fresh to mineral POC (c), PON (f) and TEP (i) concentration. Values above 1 have a fresh POC, PON or TEP concentration that exceed the mineral-associated one. The lower panel (k) shows the bathymetry (m below mean sea level) along the transect.

4.3.4. Location of dumping sites within coastal to offshore gradient

Changes in coastal ecosystems are often correlated with changes in water clarity or SPM concentration and thus with the POM content of the SPM (e.g. May et al., 2003; Capuzzo et al., 2015). The area where the concentration of fresh and mineral-associated POM is about equal is of particular interest. Though still imperfect, the application of the refined model to satellite SPM concentration products or high-resolution in-situ time series of calibrated optical or acoustical instruments yields spatio-temporal compositional changes of the SPM, with respect to POC, PON, and TEP.

Figure 4.6 shows the location of the dumping sites within the fresh to mineral-associated POC distribution. During winter the POC is dominated by the mineral-associated fraction, while in spring and summer the fresh POC dominates in the offshore region and becomes more important in the nearshore area. However, the turbid nearshore area is always dominated by the mineral-associated fraction, because of the high SPM concentration (see Figure 4.1). Three dumping sites (i.e. S2, ZBO and OST) are located in the mineral-dominated POC area and we expect that the dumping of dredged material has only minor effects on the pelagic habitat. The other two dumping sites (i.e. S1 and NWP) are located in the transition zone, which seems to correspond with the *Abra alba* benthic habitat zone (Van Hoey et al. 2007). We hypothesize that the dumping of dredged material has shifted the transition zone further offshore at these locations and has affected the pelagic habitat.

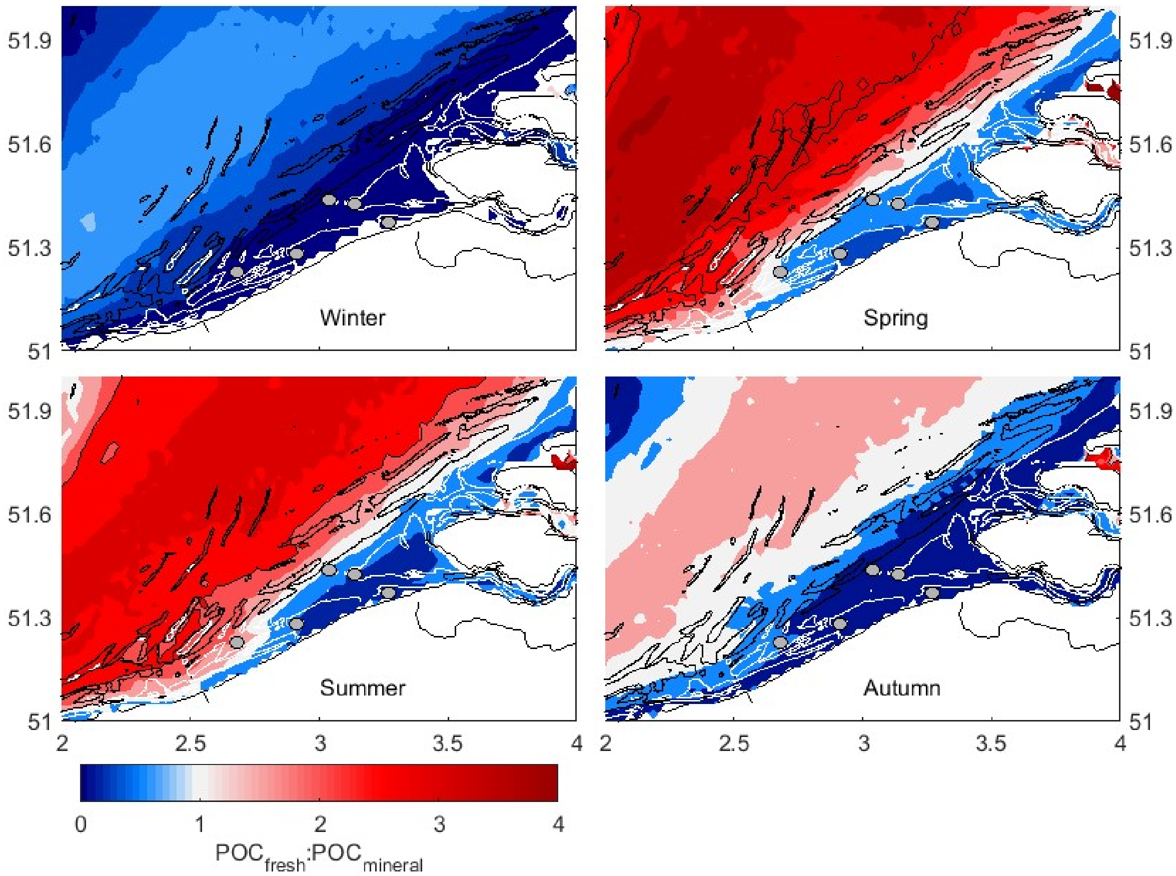

Figure 4.6: Model estimates of the ratio of fresh to mineral associated POC calculated from the surface SPM concentration from 2019-2020 from the Sentinel-3/OLCI using the model parameters. Values of the ratio above 1 have a fresh POC concentration that exceed the mineral-associated one. The white and black lines correspond with the 10m and 20m depth isolines. The grey dots are the dumping sites.

4.4 Relevance of SPM concentration and composition for ecosystem monitoring

SPM concentration

SPM concentration or turbidity are among the listed parameters to be monitored to quantify hydrographic conditions (descriptor 7), however no indicators or thresholds are yet designed. Any change in coastal management (dredged sediment disposal sites, sand mining, port developments, bottom trawling…) is expected to produce a change in the turbidity and SPM concentration, at the scale of the pressure and the surroundings. This research demonstrated that the monitoring based on optical and acoustical sensors are adapted for tracking these changes statistically, as far as they are larger than the uncertainty range, i.e. 25%, and certainly lower if analysing trends.

Our study confirms that the relation between turbidity and sample SPMC is depending on protocols, technology and the manufacturer, and even may differ between sensors of the same type (e.g. Downing 2006; Rai & Kumar 2015; Rymszewicz et al. 2017). The relation between the output of an acoustical sensor and SPMC is even more variable. In spite of these uncertainties, turbidity is often used as a proxy for water clarity or SPMC as is the dB of acoustical sensors. We advise to not use turbidity (or dB) for scientific purposes as it diminishes the comparability of the data. Instead, the sensor output should be transformed into a mass concentration, a unit that is comparable in time and between regions. If this is not possible, then the turbidity data should always be referred to the instrument used and the protocol applied. The problem aggravates when turbidity data that have been collected using different technologies and protocols over long periods of time and regional scales are stored in international data bases (e.g. turbidity in EMODnet, see http://www.emodnet.eu), and used to derive conclusive trends of the environmental status of marine and estuarine areas (Fettweis et al. 2019).

Monitoring in situ high frequency turbidity and SPMC is no longer an issue, considering that common guidance and protocols are applied to restrict their measurement uncertainties. However, in situ measurements from coastal observatories are still confined to local measurements and must not be considered alone but within a multi-source monitoring program. Hence local high frequency observations must be interpreted together with remote sensing ocean colour data, which provide daily synoptic surface turbidity/SPMC measurements, and numerical sediment transport model results, to assess the spatial extend of the pressure. The main challenge is now to evaluate model results uncertainty and improve the formulation of natural processes, together with the effects of pressures in the models.

SPM composition

The area where the concentration of fresh and mineral-associated POC, PON and TEP is about equal is of particular interest. For the German Bight, this transition zone is typically found at water depths of approximately 15 to 20 m along cross-shore transects and is characterized by a maximum of the settling velocities (Maerz et al. 2016). The same holds for the BCS. Human activities, e.g. dredging and dumping operations in the turbid nearshore, wind farms in the low turbid offshore areas or the effect of global warming (Fettweis et al. 2012; Jaiser et al. 2012; Baeye et al. 2015; Høyer & Karagali 2016) might influence the localization of this transition zone and could make it a key element in monitoring programs such as the EU MSFD. With our data-model syntheses we have consolidated a relationship and have applied it to the BCS using SPM concentrations derived from remote sensing. The application of the model to satellite or sensor derived SPM concentration products yields spatio-temporal compositional changes of the SPM, with respect to POC, PON, and TEP. This may greatly facilitate the monitoring of water quality parameters along the gradient from a domination of mineral-associated OM towards a domination of fresh OM, as documented in Figures 4.5 and 4.6.

Many oceanographic quantities are often inaccessible to direct observation, due to the high cost of in situ sampling, the limitation of the standard water quality parameters and the low spatial and time resolution. Proxies, based on automated, highly resolving instruments are valuable as they help to extend under-sampled or unobserved parameters. In this regard, SPM concentration as a proxy for POM, POC, PON, and TEP concentrations seems to be a key ingredient for the assessment and calibration of numerical models of coastal waters, ranging from SPM transport and deposition to key ecosystem processes.

5. Biological and chemical aspects related to dredging and dumping operations

The disposal of dredge material will, by its very physical nature, change the seafloor ecosystem. An overview of the potential effects of dredge disposal on the different components of the seafloor ecosystem are summarized in Table 5.1 (Hitchcock et al. 2002; Newell et al. 2004). The follow-up of several of those effects (habitat loss, increased suspended sediment concentration, smothering, contaminants) are part of the ILVO-dredge monitoring program.

Table 5.1: Overview of potential effects on the seafloor ecosystem (habitats and species) resulting from sediment disposal.

Direct effect	Indirect effect	Impact on sea floor ecosystem
1. Habitat loss		Disappearance of benthic community
2. Increased suspended sediment concentration	Increased light attenuation	Reduced primary production, reduced benthic biomass
	Decreased filtering efficiency (filter feeders)	Reduced biomass of filter feeders
	Decreased predation efficiency for visual predators	Reduced biomass of predators
3. Smothering (sedimentation)		Burial resulting in reduced biomass, density and death of individuals
	Changed sediment composition	Changes in biodiversity and biomass
4. Release of organic matter and nutrients		Hypoxia, dominance of eutrophication related species
5. Introduction of contaminants		Bioaccumulation of contaminants in marine food chain
	Changes in water quality	Changes in biodiversity and dominance of species tolerant to pollution
6. Changes in hydromorphological regime	Hydrodynamic changes, altered grain size	Changes in biodiversity and biomass. Loss of habitats and species
	Changed light attenuation	Reduced primary production and carbon flow to the seafloor, reduced benthic biomass

The impact on the seafloor and its biological components depends on the intensity of disturbance which can be related to the type, the intensity, the frequency, the duration and the spatial extent of the disposal activity (Bolam et al. 2006). A key aspect that determines the level of impact caused by dredge disposal is if the disposed material is similar to the sediments at the disposal site and what the natural morphodynamical characteristics of the disposal site are. Besides, the amount of material per area and over which time period (frequency) is crucial for this impact determination. Disposal can be on a long term/permanent or temporary basis and it can affect only the disposal location or also its wider surroundings.

The aim of this chapter is to evaluate the ecological and chemical status of the seafloor ecosystem at the five dumping sites. This is mainly funded on data collected over the last license period (2015-2020), which is for certain aspects compared to the results obtained in previous license periods to get an overview over the long-term. This chapter contain several topics, reflecting the regular monitoring outcomes or specific research actions in support of the dredge disposal research program. A more in-depth analysis and outline of the obtained results is available in Van Hoey et al. (2021).

In the regular ecological monitoring program, a number of biological population parameters of the macrobenthos, epibenthos and demersal fish fauna were monitored according to a control/impact design and with some periodicity at the 5 dumping sites. The periodicity of monitoring depends on the dumping intensity in the areas, the historical time series and the biological value of the zones. The macrobenthos (infauna) is a very good indicator due to its sessile way of life and sensitive to the pressures related to dredging discharges (paragraph 5.2). The epibenthos and demersal fish fauna are indicators of the higher trophic level, but are more tolerant due to their mobility (paragraph 5.1). In this way, an important part of the marine ecosystem is monitored (biologically). The aim of this basic research is to evaluate how the status of the marine benthic ecosystem evolves under the pressure of the dredging policy. This evaluation is done as much as possible on the basis of indicators. In Belgium, the Benthic ecosystem quality index (www.beqi.eu) is used for the ecological status assessments. Therefore, the evaluation of the macro-, epi- and fish fauna is summarized by means of the BEQI indicator (official benthic indicator in our environmental legislation), which quantifies and judges on how the fauna within the disposal area deviates from the surrounding not impacted fauna (see paragraph 5.1 and 5.2). Also, other indicator approaches are tested based on our dredge disposal monitoring data, as for example the general-purpose biotic index (Labrune et al. 2021). Until know, the evaluation of the impact of dredge disposal looked at changes in densities, biomass and diversity of the benthic communities (structural characteristics), which is the main requirement in EU environmental legislation. Nevertheless, it is also necessary to understand if possible changes in the structural characteristics lead to changes in the functioning of the benthic communities. This aspect is explored in paragraph 5.3, where we evaluate the use of functional diversity indices and a multivariate visual tool in assessing changes in benthic functioning. These analyses are based on biological trait analysis, where the benthic communities are quantified based on morphological, behavioural and other life history characteristics (Beauchard et al. 2017; Bremner et al. 2003, 2006) and hence enables us to detect possible changes in ecosystem functioning. In relation to optimize the monitoring of the sea-bottom for the EIA of dredge disposal, we applied the Sediment Profile Imaging (SPI) technique (paragraph 5.4). SPI provides an in-situ perspective of the sediment-water interface and subsurface sediments, providing both quantitative and qualitative data on the biological, chemical and physical characteristics of the sediments (Germano et al. 2011). Specific SPI monitoring was executed over the period 2014-2018. Several biological, physical and chemical parameters and derived SPI indices (BHQ, OSI) were assessed, through image analysis, and the performance evaluated for detecting environmental disturbance related to dredge disposal (paragraph 5.4).

The chemical evaluation of the disposal of dredged material focused on priority chemicals such as metals, PAHs and PCBs (paragraph 5.5). In addition to these priority components, an assessment was made of the presence of the antifouling agent TBT and booster biocides applied in antifouling paints (paragraph 5.6). The occurrence and distribution of marine litter on and near dumping sites is also monitored (paragraph 5.7). The study on antifouling agents is part of a broader investigation on contaminants of emerging concern. This research already started in the period 2012-2016 with the execution of a general chemical screening as well as the determination of pesticides and is continued in 2017-2021, considering internationally available knowledge on new pollutants, e.g. through the ICES Marine Chemistry Working Group. For the upcoming monitoring cycle, this work, based on targeted analysis of contaminants, will be broadened with analysis of unknown contaminants through untargeted screening approaches.

For the evaluation of the chemical status, it will be checked whether the chemical quality of the sediment on the dredge disposal sites does not deviate from the rest of the Belgian part of the North Sea (BPNS) and whether the investigated organisms, coming from the dredge disposal sites, do not accumulate increased concentrations of certain contaminants. Trends in chemical contamination will be discussed and the measured contaminant concentrations will be assessed against environmental limit values such as the OSPAR environmental assessment criteria (EACs) and the effect range low values (ERLs).

The collected biological and chemical data and evaluation within this project has also served as input for the implementation and reporting requirements for European directives (e.g. the Marine Strategy Framework Directive (MSFD)) (Belgische Staat 2018).

5.1 Epibenthos and fish fauna at the dumping sites based on 15 years of data

Impacts of dredge disposal on the benthic community are elucidated in this and the following chapters of this report. This chapter focuses on epibenthos and demersal fish. In contrast to the macrobenthic community, these animals live on or just beneath the sediment surface and are mobile, which should make them less vulnerable to disturbances such as dredge disposal. However, indirect impacts on epibenthos and fish are still possible because of habitat changes.

Epibenthos and demersal fish were monitored using an 8-meter beam trawl (22mm mesh size) in two seasons (March and September/ October) over the period 2005-2019 at the five dredge disposal sites. The dredge disposal sites were sampled according to a control-impact design. Therefore, 1 or 2 fish tracks were executed in the impact (I) and nearby control (nC) site and 1 track was carried out within a reference area (1-2 far control areas (fC) for each disposal site), allowing three assessment categories (I versus fC+nC, I versus fC, I versus nC). All samples were sorted and the animals were identified to the lowest taxonomic level possible, counted and weighed. The exact sampling time and coordinates were recorded in order to convert the data towards surface units (1000 m²). In the analyses, only data on bentho-pelagic fish and epibenthos were included. Pelagic species and macrobenthos were excluded, since these groups cannot be sampled adequately using a beam trawl. After standardizing both datasets (fish and epibenthos), the number of taxa, the Shannon-Wiener diversity, density and biomass (g Wet Weight, except for fish) were determined per 1000 m². Afterwards, the mean species richness and density (+ standard errors) were calculated per year and season for the I, nC and fC samples. To assess the deviations between the impact and control areas, a BEQI assessment (Benthic Ecosystem Quality Index; http://www.beqi.eu/) was executed for each assessment category and for three time periods (period 1: 2005-2009; period 2: 2010-2015; period 3: 2016-2019). This indicator is based on four biological parameters, i.e. species richness, Bray-Curtis similarity, density and biomass, and can have values between 0 and 1, where scores below 0.6 indicate a significant deviation from the reference areas.

Table 5.2: Overview of the average BEQI scores between the impact and nearby and far control sites for each dredge disposal site and for the three periods. The colours indicate the boundaries of the different classes (blue: high (0.8-1); green: good (0.6-0.8); yellow: moderate (0.4-0.6); orange: poor (0.2-0.4); red: bad (0-0.2) comparability). Values in bold have a good or moderate confidence; those in italic have a low or very poor confidence.

		Epibenthos					Fish				
		S1	NWP	OST	ZBO	S2	S1	NWP	OST	ZBO	S2
Period 1 (2005-2009)	Sep/ Oct	*0.78*	**0.77**	*0.76*	*0.86*	*0.84*	**0.87**	**0.89**	*0.89*	**0.62**	**0.88**
Period 2 (2010-2015)	Sep/ Oct	**0.45**	**0.80**	*0.84*	*0.68*	**0.78**	**0.64**	**0.84**	**0.83**	**0.73**	**0.87**
Period 3 (2016-2019)	Sep/ Oct	**0.56**	**0.78**	**0.72**	**0.81**	**0.83**	**0.56**	**0.73**	**0.81**	*0.77*	**0.48**
Period 1 (2005-2009)	March	*0.82*	**0.71**	*0.75*	**0.85**	*0.81*	*0.87*	**0.75**	**0.80**	**0.68**	**0.88**
Period 2 (2010-2015)	March	**0.38**	**0.83**	**0.40**	**0.85**	*0.71*	**0.77**	**0.83**	**0.68**	**0.79**	**0.81**
Period 3 (2016-2019)	March	*0.26*	*0.88*	*0.65*	*0.40*	*0.97*	**0.66**	**0.66**	**0.85**	**0.49**	**0.64**

At the dumping site OST, the epibenthic diversity and density was very similar in the I, nC and fC samples, despite some discrepancies due to a patchy distribution of specific species (e.g. brittle stars *Ophiura ophiura*, starfish *Asterias rubens*, common shrimps *Crangon crangon*). This high similarity was reflected in the high average BEQI scores (> 0.6), except in period 2 March (Table 5.2). Generally, a good comparability was also observed between the I, nC and fC samples for epibenthos at ZBO (BEQI scores > 0.6), despite some variability in period 3 due to a high density of brittle stars (*O. ophiura)* at the control site (Table 5.2). For fish, there was a good similarity between the I, nC and fC samples, both at the sites near Zeebrugge and Ostend (BEQI scores > 0.6), although the BEQI scores were slightly lower in some cases due to higher or lower densities of gobies (*Pomatoschistus*) (Table 5.2). These results suggested that there was no significant impact from dredge disposal at both areas, both situated in the muddy *Limecola balthica* habitat. This is explained by the similarity of the disposed sediments to the habitat and also the relative high resilience of the habitat (Bolam & Rees 2003; Bolam et al. 2006).

At the dumping site NWP, average BEQI scores for epibenthos were higher than 0.6, although there were some lower values for density and biomass in the winter period, especially because of a high abundance of brittle stars (*O. ophiura*) and starfish (*A. rubens*) at the I site, compared to the nC site (Table 5.2). There also was a good comparability for fish, despite higher densities of certain fish species at the I site in period 3 (e.g. pout *Trisopterus luscus*, dab *Limanda limanda*). At the dumping site S1, situated in the same fine muddy *Abra alba* habitat as NWP, a lower number of taxa and lower densities were found for epibenthos at the impact site, especially during period 2 and 3. For fish, only densities were lower at the impact site; the species richness was similar between categories. These findings were reflected by low BEQI scores for S1, especially in case of density and biomass of epibenthos and fish, but for epibenthos also in case of species similarity and richness (*Table 5.2*). Especially densities and biomass of brittle stars and starfish were lower at the impact site. The contrasting findings between S1 and NWP were related to the higher amount of disposed material at S1, compared to NWP. The results indicated that the impact at LNP was minimal and that especially the epibenthic community at S1 was clearly affected by dredge disposal.

The structural characteristics (i.e. species richness and density) of epibenthos and fish were very similar at the I, nC and fC sites of the duping site S2 (located in the sandy *Nephtys cirrosa* habitat). This was also reflected by good to high BEQI scores (respectively higher than 0.6 or 0.8) (*Table 5.2*). Some deviations were found for density due to differences in densities of brittle stars (*O. ophiura*) and gobies (*Pomatoschistus*), especially in period 3.

In general, the epibenthos and fish communities were not much affected by the dredge disposal activities, mainly related to their mobility capacity. Most differences were related to natural variability and to the patchy distribution of certain species (e.g. *O. ophiura*, *A. rubens, C. crangon*, *Pomatoschistus*). Due to this , the confidence of the BEQI assessments was low in some cases (e.g. density, biomass) and the possibility to observe deviations between impact and reference areas was therefore difficult. The dumping site S1 was the only area at which a significant impact, especially for epibenthos, was found. Because of the high disposal intensities every year, recovery of the epibenthic community is less probable in this area. Furthermore, the input of fine sediments also causes changes in habitat, which affects the benthic community (Bolam et al. 2006). Since densities of epibenthos and macrobenthos were lower at S1 and thus also the food availability for fish species was reduced, fish densities were also lower, although fish were impacted to a lower extent. Probably, the fish can use the S1 area less as feeding ground.

5.2 Patterns in the structural characteristics of the benthic fauna at the dumping sites and summarized by benthic indicators

The macrobenthic community is potentially more affected by dredge disposal, compared to epibenthos and demersal fish, since these animals have a more sessile life mode. The impact on macrobenthos can be related to the dredge disposal intensity and also to the environment (Bolam & Rees 2003; Bolam et al. 2006).

In the BPNS, the macrobenthic community was monitored yearly according to a control-impact design over the period 2006-2019 (autumn sampling) at five areas designated for dredge disposal. Because of lower dredge disposal intensities at the sites near Nieuwpoort and Ostend, these areas were only sampled once in three years. The samples were sorted in the lab and the species were identified, counted and weighed, following specific guidelines which are under accreditation. For each dredge disposal site approximately 7 impact samples (I), a number of nearby control samples (nC) (varying from 4 in the beginning period to about 12 in the later years) and 6-15 far control samples (fC) (3 replicates per control area) were defined. Three assessment categories were considered: the impact samples were compared to a combination of the nC and fC samples and to the nC samples and the fC samples separately. After standardizing the dataset, several structural characteristics, i.e. species richness, density and biomass, were determined for all samples. The mean (+ standard error) of these biological parameters was calculated per year for the impact, nearby control and control area of each dredge disposal site. Furthermore, a BEQI assessment (Benthic Ecosystem Quality Index) was executed, based on species richness, Bray-Curtis similarity, density and biomass, to assess the ecological status of the different dredge disposal areas. BEQI scores can be between 0 and 1, where values below 0.6 imply a deviation between the assessment categories. Another indicator, the GPBI (General-Purpose Biotic Index), also with assessment values ranging between 0 and 1, was calculated for areas with no pressure (reference) and with high and intermediate dumping pressure (Labrune et al. 2021).

Table 5.3: Overview of the average BEQI scores between the impact and nearby and far control sites for each dredge disposal site. The colour code represents the boundary values, where blue shows high (0.8-1), green good (0.6-0.8), yellow moderate (0.4-0.6), orange poor (0.2-0.4) and red bad (0-0.2) comparability. Values in bold have a good or moderate confidence; those in italic have a low or very poor confidence. Blank fields represent no data.

	S1	NWP	OST	ZBO	S2
2006	**0.26**	*0.87*	*0.86*	*0.54*	*0.67*
2007	**0.27**	**0.66**	**0.89**	**0.56**	*0.69*
2008	**0.18**	**0.62**	**0.71**	*0.65*	*0.90*
2009	No data				
2010	**0.24**	*0.87*	*0.92*	*0.81*	**0.61**
2011	**0.46**	*0.78*	*0.88*	*0.85*	*0.53*
2012	**0.39**			*0.79*	*0.85*
2013	**0.41**		**0.88**	*0.68*	**0.82**
2014	**0.44**	**0.75**		*0.67*	**0.88**
2015	No data				
2016	**0.34**		*0.80*	*0.82*	**0.76**
2017	**0.44**	**0.84**	*0.84*	*0.64*	**0.84**
2018	**0.30**		*0.74*	*0.64*	
2019	**0.37**			**0.66**	*0.85*

For the dumping sites OST and ZBO (situated in the muddy *Limecola balthica* habitat), the biological parameters of the I, nC and fC samples showed a similar trend over time, related to natural variability, although the species richness was relatively low at the control sites. This was explained by the low diversity at the control stations ZEB and ZVL. Despite the low diversity at these stations, the number of species tended to increase over time, especially at site ZBO. The BEQI assessment confirmed the similarity between the three categories, with a high comparability for OST (average scores > 0.8) and a good comparability for ZBO (average scores > 0.6) (Table 5.3). The slightly lower BEQI scores for ZBO can be related to the higher amount of disposed sediments per year. However, the impact on the *L. balthica* community was still negligible, which can be related to the similarity between the sediments and the disposed material and the high resilience of the *L. balthica* community.

Despite some discrepancies in the BEQI patterns at dumping site NWP, there was a good too high comparability between the I, nC and fC areas (BEQI scores > 0.6) (Table 5.3). Here, an increase in species richness over the study period was also observed in all categories, similar to the *L. balthica* habitat. In contrast, a consistently moderate/ poor/ bad similarity (BEQI scores < 0.6) was observed over time at the dumping site S1, especially for density and biomass (Table 5.3). This was confirmed by the structural characteristics, for which lower species richness, densities and biomass were found at the impact site (Figure 5.1). The contrasting findings for LS1 and LNP, both located in the fine muddy sand *Abra alba* habitat, were related to the yearly difference in the amount of disposed sediments. With high disposal intensities every year, as at S1, the macrobenthic animals are highly affected by smothering and recovery is less probable (Bolam & Rees 2003; Bolam 2011). Furthermore, the repeated introduction of a high amount of fine sediments in a fine muddy sand habitat causes a change in sedimentology, which has an influence on the species composition. Also, the nearby control site tended to be affected by dredge disposal, since the biological parameters in the impact and nearby control site were more similar during the last period. Therefore, the ecological status tends to decrease with increasing disposal intensities in the *A. alba* habitat.

At the dumping site S2, the similarity between the I and nC areas was good throughout time (BEQI scores > 0.6), although the BEQI values were slightly lower for the assessment between I compared to the fC (scores between 0.4 and 0.6). This was probably caused by the higher species richness and density at the impact and nearby control site. An explanation for this 'positive' trend is the creation of a new habitat by the introduction of fine sediments in a sandy environment (*Nephtys cirrosa* habitat) and therefore the attraction of species associated with muddy sediments (De Backer et al. 2014).

Figure 5.1: Average species richness and density (+ standard errors) for macrobenthos at the impact, nearby control and control sites of the dredge disposal site S1. Data for 2009 and 2015 were lacking for the three categories.

The new tested indicator GPBI on this dredged disposal data, gave similar results. Low GPBI values were found for areas with high and intermediate dumping pressure within the *A. alba* habitat, compared to reference areas with no pressure. For the *L. balthica* and the *N. cirrosa* habitat, the significant differences between GPBI values for the three levels of pressure were not consistent over time. Nevertheless, GPBI was still defined as a good indicator to assess the impact of any kind of disturbance on the benthic community (Labrune et al., 2021).

The present study showed that the time series graphs based on the structural characteristics of the assessment categories and the indicator analyses (BEQI and GPBI) were appropriate methods to assess the ecological status and its evolution of the dredge disposal sites in the BPNS. Based on these results, the *A. alba* habitat was defined as the most sensitive habitat to dredge disposal, while the *L. balthica* and *N. cirrosa* habitats were more resilient. Nevertheless, the impact at ZBO may be underestimated because of the possibility that the reference areas ZEB and ZVL are not containing a healthy *L. balthica* community. Therefore, it is recommended that the far control areas for the *L. balthica* habitat should be reconsidered in the future. Future monitoring is necessary to look whether the observed patterns are consistent and to investigate the similarity between the impact and nearby control sites at some of the dredge disposal sites (possible disposal effect in the surroundings). Also, the changes in species composition and the increasing trend of species richness over the study period should get attention in future monitoring and analyses.

5.3 Patterns in functional diversity and functional trait composition of the benthic fauna at the dumping sites

Biological trait analyses are used to evaluated the impact of long-term dredge disposal dumping on the functional diversity of soft-bottom macrobenthic communities. In biological trait analysis, differences in benthic communities are not quantified based on taxonomic differences but based on differences in morphology, behaviour and other life history characteristics (Beauchard et al. 2017; Bremner et al. 2006). Trait based analyses enable us to detect possible changes in ecosystem functioning, with a decrease in trait characteristics (modalities) that are sensitive to dredging pressure, when this pressure persists and/or increases. The utility of functional diversity indices (=univariate quantitative parameters) and fuzzy correspondence analyses (=multivariate visual tool) in assessing changes in benthic functioning for environmental impact assessment for dredge disposal is evaluated in this paragraph.

For this analysis, a long-term time series (2007-2016, 635 stations sampled in total) of macrobenthic data was used. An impact-control sampling strategy is followed, where the control locations (CTRL) are located in the vicinity of the impact sites (IMP) in a similar physical environment. A total of ten relevant traits were selected and subdivided in 44 modalities as done by Breine et al. (2018). Each taxon was assigned to the trait categories using a 'fuzzy coding' approach (Chevenet et al. 1994). A species-by-trait matrix was combined with the species abundance-by-station matrix to create the final station-by-trait matrix on which all subsequent trait analyses were based (Beauchard et al. 2017). The functional diversity (FD) indices consist of a range of multidimensional indices, based on principal co-ordinates analysis (PCoA). Calculated indices were functional richness (FRic), functional evenness (FEve) and functional divergence (FDiv) (Villéger et al. 2008), functional dispersion (FDis; Laliberté & Legendre, 2010) and Rao's quadratic entropy (RaoQ; Botta-Dukkat 2005). Each of these indices is an independent measure of functional trait space, and the way species are dispersed within this trait space. To test for an effect of the different human activities on the functional diversity indices, linear mixed-effect models (lmer, from the 'lme4' package in R, Bates et al. 2015) were used. To identify shifts in trait composition due to dredge disposal, Fuzzy Correspondence analysis (FCA) was performed (Dray & Dufour, 2007). FCA Ordination biplots were made whereby points in closer proximity are indicative of stations with a functional similarity. The dredging pressure categories ('none', 'low', 'medium' and 'high') were then superimposed on the reduced two-dimensional ordination output and the pairwise distances between the centroids were calculated and used as a proxy for the relative similarity between those groups (see Bolam et al. 2016). In case the impact regimes are separated on the first or second axis without too much overlap, the arrangement of the traits on that FCA axis is extrapolated and put on a gradient ranging from impact to control. In this way traits associated with the impact stations could be identified. All statistical analysis and data visualization were performed using the R software (R Development Core Team 2019).

*Table 5.4: Significance levels of the linear mixed models for the dredge disposal case per habitat. (A= A. alba, L= L. balthica, N= N. cirrosa). The significance levels are presented as: ns = non-significant, * = p<0.05, ** = p<0.001.*

		FRic			FEve			FDiv			FDis			RaoQ		
		A	*L*	*N*	*A*	*L*	*N*	*A*	*L*	*N*	*A*	*L*	*N*	*A*	*L*	*N*
none	low	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
	medium	**	ns	ns	**	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
	high	**	ns	/	ns	**	/	ns	ns	/	**	**	/	**	**	/
low	medium	**	**	ns	*	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
	high	**	**	/	ns	*	/	ns	ns	/	**	**	/	**	**	/
medium	high	ns	ns	/	ns	*	/	ns	ns	/	*	*	/	**	**	/

Functional diversity indices were useful in detecting changes in the benthic ecosystem due to dredge disposal (statistics summarized in Table 5.4). Nevertheless, in our case, functional divergence (FDiv) does not seem sensitive enough (no significant changes in any habitat or disposal site), whereas functional richness (FRic) and functional evenness (Fev) are very relevant. FDis and RaoQ followed the same trends (using one of the two is fine), as they are mathematically related indices (FDis is based on the amount of trait similarity between species in a community, whereas RaoQ between individuals in a community).

Functional correspondence analysis (FCA) is giving similar and some complementary information to the patterns revealed by the functional diversity indices. For the *Abra alba*-habitat (Figure 5.2), there is a visual separation on the first ordination axis as 'none' and 'low' appear on the left-hand side and 'medium' and 'high' are nearly overlapping on the right-hand side, as supported by the pair-wise distance values. This pattern can be allocated to a shift in trait composition at S1 (not at dumping site NWP). The trait modalities associated most with this higher dumping pressure are a short lifespan (l<1) and small body size (sr10), asexual (edAsex) or benthic egg development (edBen) and a free-living lifestyle (lhFree). Trait modalities that occur at the far-left side of the graph are mainly downwards conveyers (btDown) with a large body size (sr201-500), which are lost as a result of dumping. For the *Limecola balthica*-habitat, the pairwise distances between the pressure categories centroids suggest some separation between the dumping categories, but in the ordination, there is rather on overlap. Therefore, there is a limited effect on the functional trait composition of dumping pressure in the *Limecola balthica* habitat. For the *Nephtys cirrosa*-habitat, which corresponds with dredge disposal site S2, none of the stations experienced a 'high' dumping pressure and the remaining categories are separated on the second ordination axis with the 'none' centroid at the top and 'low' and 'medium' overlapping at the bottom. At the impacted stations, there is a higher occurrence of bioturbators (btUp and btDown) with an epiphytic or crevice-dwelling living habit (lhEpi and lhCrevice) and large body size (sr201-500).

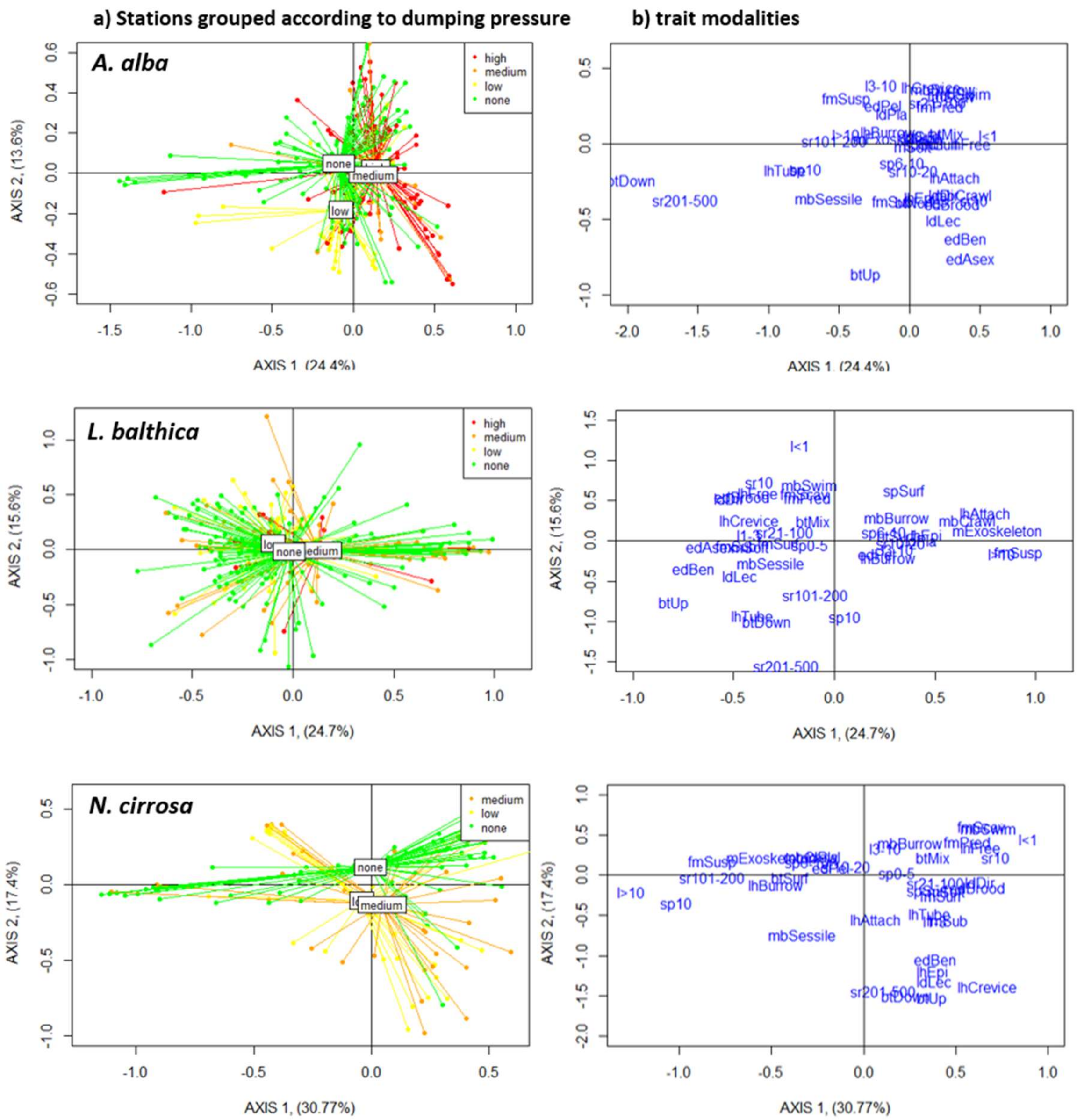

Figure 5.2: FCA ordination plots for the Abra alba habitat, Limecola balthica and Nephtys cirrosa based on a) the stations according to their pressure categories and b) the trait modalities.

As such, this FCA technique is a very useful tool to visualize shifts in traits composition between control and different impact categories. It delivers descriptive insights in which functional traits are responding to dredge disposal within different habitats and their communities. However, a quantitative parameter suited for indicator-based assessments could not yet be derived apart from the centroid distances between the factors (e.g. impact categories), but the values cannot be compared between analyses and also no thresholds values can be determined. This can possibly evolve when further specific FCA statistics are being developed.

The context-dependent responses in the functional diversity indices and FCA stress the necessity to evaluate per habitat and impact degree in environmental impact assessments. This especially in shallow dynamic areas, as in this study, where the natural environmental characteristics are crucial. Therefore, it is not really possible to generalize much of the observed patterns to other areas. This means that continuous monitoring and evaluation stays necessary.

5.4 Monitoring optimization: use of sediment profile imaging

The current sea bottom monitoring program for the environmental impact assessment (EIA) of dredge disposal is based on grab sampling (see paragraph 5.2). This is the standard technique, but the lab handling of the samples can be rather time consuming. Such type of data is absolutely necessary to execute an in depth EI assessment. This does not mean that more time and cost-efficient tools can be deployed to complement this EIA. Therefore, in the search for optimizing the monitoring practices for dredge disposal monitoring, we have looked for techniques which can deliver relevant information in a time and cost-efficient way. Sediment profile imagery (SPI) is such rapid technique that has been used since the 1970's, but has seen a resurgence of its use within the last decade for monitoring and EIA studies.

SPI provides an in-situ perspective of the sediment-water interface and subsurface sediments, providing both quantitative and qualitative data on the biological, chemical and physical characteristics of the sediments (Germano et al. 2011; Van Hoey et al. 2014). This chapter describes the test we did on the performance and applicability of the SPI as a quick screening tool for EIA for dredge disposal. Specific SPI monitoring was executed over the period 2014-2018, mainly at dumping site S1 and once at S2. Several biological, physical and chemical parameters and derived SPI indices (BHQ, OSI) were assessed, through image analysis, and the performance evaluated for detecting environmental disturbance related to dredge disposal.

The SPI sampling design at S1 consists of 42 different stations along two transects and extra locations within quadrant 1 (western part of S1) and 2 (eastern part of S1), which were sampled in autumn 2014, summer 2015, and winter, spring and autumn 2018. Not at every sampling moment, all those stations were sampled. A deviation between quadrant 1 and 2 is made, because of the fact that dumping occurs exclusively in quadrant 2 from 2014 onwards. For S2, all 16 stations of the regular sampling program were sampled in autumn 2018.

In our study, recent disposal, as well as historic disposal signs were recorded, however, the data suggests that the sediment reworking within the area is very high. The integration of the disposed sediment was clearly observed and created heterogeneous sediment profiles. The % of anoxic surface was higher in the quadrant 2 part of the dumping site, which was used much more in the period before sampling, compared to quadrant 1. This is related to the shallowness and hydrodynamics within the area, alongside continuous disposal over time.

Table 5.5: Overview of the SPI characteristic values for dumping sites S1 and S2.

		% anoxic area	Avg. penetration (cm)	Sediment classess (%)						OSI
				mud	very fine sand	fine Sand	medium sand	coarse sand	very Coarse Sand	
Br&WS1	Q1_Impact	12.08	73.58	0.0	4.3	43.6	45.3	6.8	0.0	10.24
	Q1_Control	15.43	71.30	0.0	12.6	65.5	21.8	0.0	0.0	10.01
	Q2_Impact	26.51	80.05	0.0	17.4	76.5	6.1	0.0	0.0	9.16
	Q2_Control	23.17	64.93	0.0	16.1	82.3	1.6	0.0	0.0	9.53
Br&WS2	Impact	11.59	61.27	0.0	0.0	100.0	0.0	0.0	0.0	10.62
	Control	5.54	68.35	0.0	0.0	100.0	0.0	0.0	0.0	10.07
		Mud clasts classes (%)				Surface relief (%)		Bedforms (%)		BHQ
		none	few	some	a lot	even	uneven	present	absent	adapted
Br&WS1	Q1_Impact	58.5	28.0	7.6	5.9	32.5	63.2	21.4	78.6	4.68
	Q1_Control	72.4	19.5	5.7	2.3	35.7	64.3	35.7	62.6	4.61
	Q2_Impact	62.6	21.7	4.3	11.3	28.7	71.3	56.3	43.7	4.16
	Q2_Control	80.6	9.7	4.8	4.8	25.8	74.2	48.4	51.6	4.78
Br&WS2	Impact	95.2	0.0	4.8	0.0	14.8	85.2	51.9	48.1	5.28
	Control	70.4	29.6	0.0	0.0	14.3	85.7	57.1	42.9	5.07

The relevance for using the SPI and the SPI derived parameters (Table 5.5) for dredge dumping assessment in our study area is evaluated as :

- **Sediment class and mud clasts:** Detection of significant changes in sediment type class due to dredge disposal is hard, due to the lower resolution of SPI analyses compared to quantitative sediment grain size analyses. In our study, within-class changes in sediment due to the disposal were clearly visible in comparison to control areas (e.g. S1). Mud clasts were clearly observed in many SPI pictures, but no real distinction could be made between impact and control or between the two quadrants. This is probably linked to the dumping historic in the area and the sediment reworking process due to the hydrodynamical conditions.
- **Surface relief and bedforms:** It is not easy to attribute changes in surface relief and bedforms to natural and/or anthropogenic disturbances. In dynamic, shallow areas, wave action, bottom currents or fauna presence (e.g. tube worms) are mainly shaping the sediment surface, which make it difficult to assess the impact of the disposal activity on these SPI derived parameters.
- **Prism penetration**: The results show that recent disposal activities influence the compactness of the sediment (compactness is slightly lower in Q2 of S1 compared to control and Q1) and is measurable through the prism penetration.
- **aRPD & % anoxic sediment surface**: In our study area, the a-RPD could not be determined in a proper way, due to the heterogeneity (no neatly layered structure), low penetration depth (<5cm) for some samples, or dominance of well-oxygenized sandy sediments (no a-RPD measurable). Therefore, we proposed an alternative way for those situations by using the % of anoxic sediment (dark grey or black) as a proxy of the biogeochemical redox status of the sediment. Based on this parameter, we could make a better judgement regarding the impacted areas. Quantitatively, the % anoxic area was higher for the Q2 samples, in line with the area where most recent dumping activity takes place. Nevertheless, there was almost no difference between impact and control samples.
- **Surface and sub-surface faunal characteristics**: At the dredge disposal sites, faunal characteristics were detected, but mostly in low quantities. For example, at S2, we see a difference in epifaunal organisms (i.e. brittle stars) between the impact and control. Most burrowing species in our study area do not make permanent burrows, which also lowers their SPI detectability. Two other larger burrowers, the sea urchin *Echinocardium cordatum* and mud anemone *Sagartia* spp. were regularly detected, but not in amounts sufficient for EIA assessments. The same for the tube-building polychaetes such as *Lanice conchilega* (*Owenia fusiformis* and the smaller *Spiophanes bombyx* were also observed), which were easily detectable with the SPI, but were not present in high quantities at the disposal sites to allow an EIA assessment.
- **SPI-derived indicators**: Based on the combination of the above described parameters, a SPI-derived indicator can be derived with OSI and BHQ as the most frequently used indicators (Diaz & Schaffner 1988; Nilsson & Rosenberg 1997, 2000). The OSI index is not sensitive enough to distinguish anthropogenic impact from natural disturbance processes in dynamic environments, due to the fact that the infaunal successional stages are not discriminable in our type of samples. The BHQ index is usable, but some adaptations are recommended to increase its applicability for dynamic, coastal areas. In our area, the BHQ index was not sufficiently discriminating in a situation where the aRPD was low to moderate, but fauna is present and samples with a very good aRPD but no visible infauna (only epifauna). In addition, another proxy for aRPD is defined, by using % anoxic surface area, as this seems more appropriate for heterogeneous sediments. Nevertheless, we did not find differences in BHQ between control and impact samples in any case despite the high influence of fauna to anthropogenic activity (Br&WS1). Nevertheless, the variance in BHQ values was larger in the impact samples compared to the control. Earlier studies have

suggested that increased variability could be a powerful approach to detect stress (Caswell & Cohen 1991; Warwick & Clarke 1993; Thrush et al. 1998). This indicator is also linked to ecological status classifications, with a scaling ranging between 0 (severely disturbed with no macrofauna) and 14 (in our study) - 15 ('undisturbed' mature benthic community) and a threshold of 5 to distinguish between good or poor (Nilsson & Rosenberg 1997). Generally, the habitat quality in our case studies was not optimal, with values between 4 and 6. This seems appropriate, as the benthic habitats visited at S1 are not in good status (paragraph 5.2).

Of course, this evaluation of the applicability of the SPI derived parameters is context depending and maybe slightly different when the activity is executed in another habitat type or another area. To conclude, such visual monitoring (through the SPI) of the dumping site (Figure 5.3 as example of the SPI pictures taken in November 2018 at S1) has clear advantages for the evaluation process. First, we can detect patterns very rapidly due to the image overview you get from the investigated area. Second, we get a clear view on the sediment composition and how it is structured, which give us information on the smothering and sediment reworking processes going on in the area (and outside at the borders). Third, sedimentological and biological characteristics in a (semi-) quantitative way can be derived from it. Therefore, in the next monitoring cycle a SPI assessment of each disposal site will be conducted.

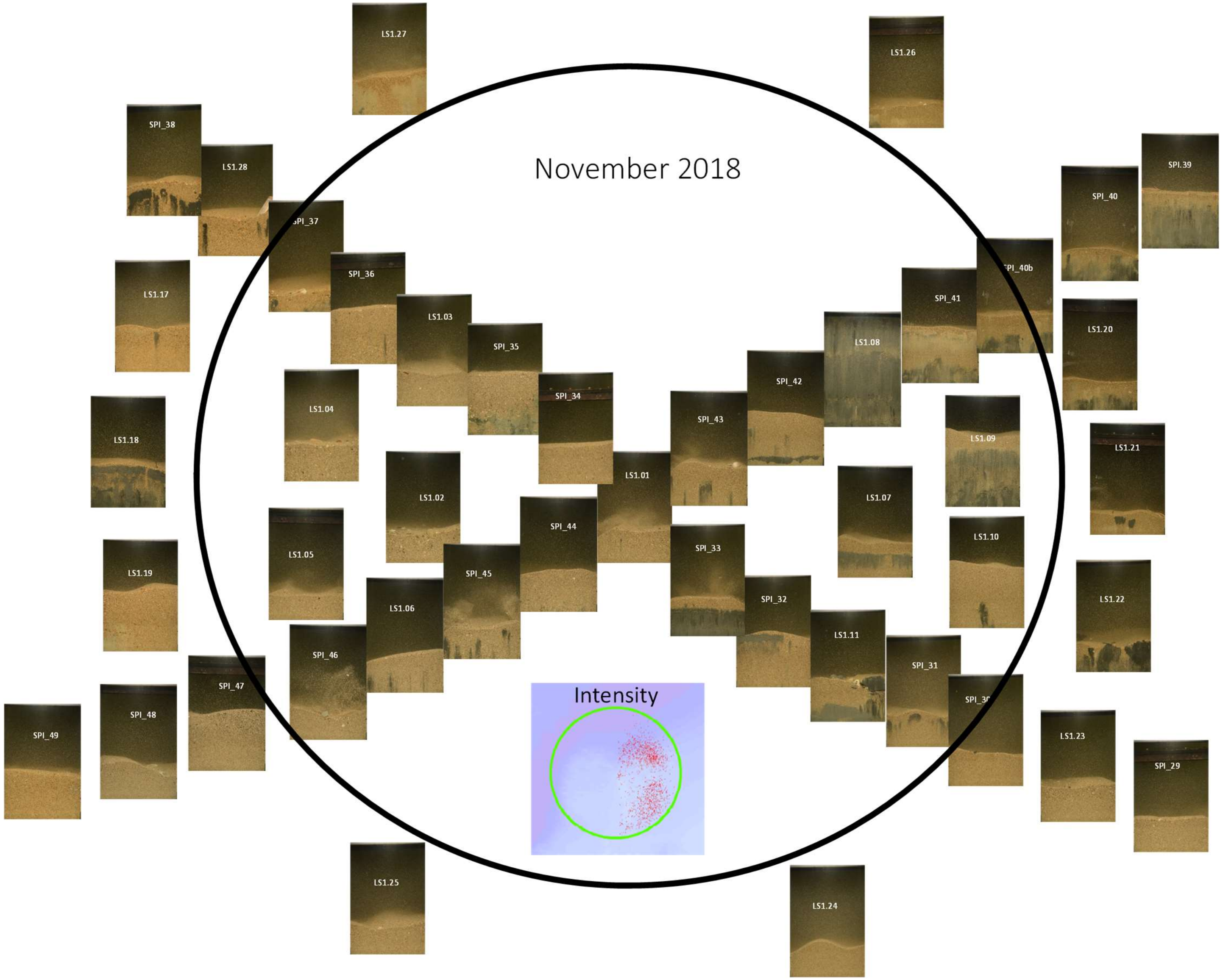

Figure 5.3: Overview of the SPI pictures taken over the dredge disposal site S1 in November 2018

5.5 Trends in chemical contaminants at Belgian dredge disposal sites

The chemical status of dumping sites is assessed by evaluating concentrations of polycyclic aromatic hydrocarbons (PAHs), polychlorobiphenyls (PCBs) and metals in sediment and biota. Sediment samples were taken from 2007 to 2019 at (1) the actual dumping site, (2) the directly impacted zone, i.e. outside but less than 0.3 nautical mile away from the actual dumping site, and (3) reference locations at longer distance from the dumping sites. Two groups of reference samples were defined, based on the Folk sediment classification. Reference area 1 contained sandy reference samples, reference area 2 contained reference samples categorized as mud to muddy sand (Van Lancker, 2009). Biota samples were taken from 2005 to 2018. Shrimp (*Crangon crangon*), starfish (*Asterias rubens*) and swimming crab (*Liocarcinus holsatus*, *Liocarcinus marmareus*) were sampled at or around the dumping sites or at reference areas.

For data assessment, sediment contaminant concentrations were expressed in dry weight and normalised according to OSPAR procedures: data on metals was normalised to 5.8% aluminium (Al) considering a total digestion procedure, data on organic contaminants was normalised to 2.5% total organic carbon (TOC) (OSPAR 2008). Sediment data also implied a granulometric normalisation as all analyses were done on the <63 µm fraction (OSPAR 2008). Biota data was expressed on wet weight for metals and PAHs or lipid weight for PCBs. Linear mixed effect models were applied in R (version 4.0.3), using a parametric trend modelling procedure. Prior to model fitting, influential outliers were eliminated based on Cook's distance, which calculates the impact of each data point on the regression analysis. A cut-off value of 0.2 was applied to discard outliers from the modelled datasets. The initial model contained parameters year, year^2, season (defined as quadrimesters), area, analysis code and the interaction term area:year. Log transformation of the contaminant concentrations was needed to ensure that model residuals followed a normal distribution and to comply with normality conditions necessary to make the fitted model valid. Stations (locations) were included as random effect to capture the spatial correlations between contaminant levels in sediments sampled at the same location. Given that sediment contaminant concentrations at one location over several years are expected to be more similar compared to concentrations in the sediment at other locations, each location is assigned a different random intercept value that is estimated by the mixed-effects models. Sampling season, area and analytical method were included as fixed effect terms in the full optimal mixed effect model, to evaluate how pollution concentrations are evolving over time at the different dumping sites. An interaction term between time and areas was added to allow different trends/slopes per dumping site. The analytical method was included as explanatory variable to allow to correct for concentration shifts due to changes in analytical method. The factor analytical code was applied for the biota analysis and for the metal and PAH sediment analysis. For PCB in sediment, this factor was not included as 3 method changes took place between 2009-2019, resulting in too short time periods to include the factor "analytical method".

Model fitting was deduced from restricted maximum likelihood (REML) estimations using R function nmle:lme. (Pinheiro et al. 2018). Starting from the full optimal mixed effect models, fixed variables were selected by means of the maximum likelihood estimation (ML) via the R lme4:drop1() function (Bates et al., 2014), which allows for single term deletions for nested models and likelihood ratio tests (LRT) based on a Chi^2 statistic. All remaining terms were significantly different from 0 at the 5 % level. Model validation as described by Zuur et al. (2009) was performed on normalized residuals to verify homogeneity of variance and independence.

Sediment data was compared to background assessment concentrations (BAC) and environmental assessment criteria (EAC). BAC values indicate whether contamination levels are "near background” (for naturally occurring substances) or “close to zero” (for man-made substances). EAC values represent the contaminant concentration in the environment below which no chronic effects are expected to occur in marine species, including the most sensitive species (OSPAR 2009). Effect Range Low (ERL) values, developed by US-EPA, are given, not normalised to TOC or Al. ERL values also indicate the concentration below which effects are not likely (OSPAR 2009). The ERL value is defined as the lower tenth percentile of the data set of concentrations in sediments which were associated with biological effects. Adverse effects on organisms are rarely observed when concentrations fall below the ERL value, and the ERL therefore has some parallels with the philosophy underlying the OSPAR EACs and WFD EQSs.

5.5.1. Inorganic contaminants

The routine analysis of inorganic contaminants at the dumping sites involves arsenic (As), cadmium (Cd), chrome (Cr), cupper (Cu), mercury (Hg), nickel (Ni), lead (Pb) and zinc (Zn) in marine sediments and/or biota. Al is measured as normalizer for sediment analysis. Analytical methods for sediment analysis are described in De Witte et al. (2016). For heavy metal analysis on biota, microwave extraction with HNO_3 is performed, followed by ICP-MS or ICP-OES quantification. Hg is determined by dry combustion with oxygen and Au-adsorption by an AMA254 Hg analyser. Table 5.6 compares model outcomes with BAC, EAC and ERL data. Tables 5.7 to 5.9 summarize time series results.

Concentrations of Cd, Cr, Cu, Hg, Ni, Pb, and Zn decrease in the sediment of each dumping site and at both reference areas. In contrast, the model revealed a strong increase of 78% for arsenic (As) from 2007 until 2018. Metal concentrations do not reveal a clear effect of dumping of dredged material, as concentrations at the impacted area are not clearly elevated from reference areas (Table 5.7). Highest concentrations of Cu, Ni and Zn were recorded at the nearby zone of S2. Trends of inorganic compounds in biota give a mixed view, with decreased or steady state metal concentrations in swimming crab, except for Zn, decreased or steady state metal concentrations in starfish, except for Cr, and steady state or increased concentrations in brown shrimp. Metal concentrations are much lower in brown shrimp compared to swimming crab and starfish. Differences in trends in metal concentration in brown shrimp compared to other species might be linked to a levelling off of metal concentrations over the last decade combined with a larger impact of measurement uncertainty on low metal concentrations.

A decrease of Pb, Cd, Cu, Ni and Hg concentrations is as expected and can be related to pollution control measures in industry and transport (Le et al. 2021). Moreover, trends are in line with an assessment on longer time scale, evaluating metal trends at the BPNS from 1971 until 2015 (Le et al. 2021). Although concentrations are decreasing, strict follow up is needed, as Pb and Hg concentrations are not only above BAC at the Southern Part of the North sea, but also above the ERL, indicating a potential environmental risk (Table 5.6). For Cr, decreasing trends in sediment samples are confirmed by measurements in swimming crab but not by measurements in starfish and shrimp, indicating the need for a close follow up of Cr contamination in the next 5-years’ time frame.

Zn concentrations were found decreasing in marine sediments at or around dumping sites from 2007 to 2018. On a longer timeframe, however, Zn concentrations were found to be increasing at the BPNS from 1971 to 2015 (Le et al., 2021). Results are consistent, as Le et al. (2021) modelled not only an increase in Zn concentrations since 1971, but also a levelling off and even decrease of Zn concentrations between 2010 and 2015. Similarly, Le et al. (2021) detected strong decreases in As concentration from 1997-2015, with an increase over a shorter time period (2010-2015). Again, increasing 2007-2018-data on As are in line with the work of Le et al. (2021). For both Zn and As, fluctuations in trend over time make it necessary to continue a close monitoring of metals in order to evaluate the chemical status of the marine environment at or around dumping sites.

5.5.1. Organic contaminants

The routine analysis of organic contaminants at the dumping sites involves PCBs and PAHs in marine sediments and biota. Analytical methods are described in De Witte et al. (2014, 2016) and Le et al (2021). For PCBs in sediment, an additional method shift occurred from 2018 samples onwards, due to an instrumentation change. Samples from 2018 and later were extracted with the procedure as applied for PAHs in sediment (De Witte et al., 2016) but analysed by gas chromatography (GC)-triple quadrupole mass spectrometry (MS) with separation of PCBs on a TR-PCB 8MS column (Thermo, 50m, 0.25mm, 0.25µm). Tables 5.8 and 5.9 provide trend model results for PAHs and PCBs, respectively. *Table 5.6* compares model results with BAC, EAC and ERL data.

*Table 5.6: Comparison of predicted environmental concentration (2018) with EAC (PCB) and ERL (PAH and metals). Cd and Pb values are reported in mg.kg^{-1} d.w. All other values are reported in µg.kg^{-1} d.w. All data are normalised to 2.5% TOC (PAH, PCB) or 5.8% Al (Hg, Cd, Pb). BAC values are normalised to 2.5% TOC or 5.8% Al, EAC values are normalised to 2.5% TOC, ERL values are not normalised (De Witte et al. 2016). *Exceedance of EAC*

	S1	S2	ZBE	OST	NWP	Ref1	Ref2	BAC	**EAC/** ERL
Naphthalene	24.9 (19.4-31.9)	29.7 (22.4-39.3)	38.4 (29.8-49.3)	37.1 (28.8-47.7)	30.0 (20.5-43.9)	39.3 (31.8-48.6)	43.0 (31.8-58.3)	8	160
Phenanthrene	30.7 (25.5-37.0)	49.6 (37.8-65.2)	55.4 (44.4-69.1)	56.8 (45.1-71.6)	37.8 (28.7-49.9)	42.1 (35.4-50.1)	43.4 (35.2-53.6)	32	240
Anthracene	10.3 (8.6-12.3)	16.2 (12.5-20.9)	19.9 (16.1-24.6)	18.9 (15.1-23.5)	13.1 (10.1-17.1)	13.9 (11.8-16.4)	14.8 (12.1-18.1)	5	85
Fluoranthene	48.3 (40.1-58.2)	77.2 (58.7-102)	94.0 (75.3-118)	90.9 (72.1-115)	61.2 (46.3-80.9)	66.3 (55.6-78.9)	68.0 (55.0-84.1)	39	600
Pyrene	36.8 (30.8-43.9)	55.9 (43.3-72.2)	66.1 (53.7-81.5)	67.6 (54.3-84.0)	44.1 (34.0-57.2)	50.0 (42.4-58.9)	51.0 (41.8-62.3)	24	665
Benzo(a)anthracene	20.1 (16.5-24.5)	33.7 (25.3-44.9)	40.8 (32.3-51.5)	40.3 (31.6-51.4)	27.8 (20.8-37.2)	29.3 (24.4-35.2)	29.7 (23.8-37.1)	16	261
Chrysene	25.7 (21.5-30.8)	42.8 (33.1-55.4)	49.3 (39.9-60.9)	48.2 (38.7-60.1)	34.8 (26.8-45.2)	38.1 (32.2-45.1)	37.1 (30.3-45.4)	20	384
Benzo(a)pyrene	25.5 (21.1-30.7)	41.6 (31.7-54.4)	47.6 (38.2-59.4)	48.2 (38.2-60.6)	35.4 (26.9-46.6)	38.1 (32.0-45.3)	36.8 (29.8-45.5)	30	430
Benzo(ghi)perylene	23.0 (18.8-28.1)	37.7 (29.2-48.8)	43.7 (34.9-54.7)	43.8 (34.8-55.2)	37.5 (28.5-49.3)	38.5 (32.2-46.0)	36.7 (29.6-45.7)	80	85
Indeno(123cd)pyrene	32.9 (27.1-39.9)	53.6 (40.5-71.0)	65.3 (51.9-82.1)	61.4 (48.4-78.1)	47.2 (35.5-62.9)	48.9 (40.8-58.5)	47.2 (37.9-58.7)	103	240
CB28	0.36 (0.29-0.44)	0.51 (0.39-0.68)	0.63 (0.49-0.79)	0.50 (0.39-0.64)	0.40 (0.30-0.54)	0.43 (0.35-0.52)	0.41 (0.32-0.52)	0.22	**1.7**
CB52	0.51 (0.34-0.76)	0.43 (0.28-0.64)	0.46 (0.30-0.69)	0.40 (0.26-0.60)	0.41 (0.27-0.63)	0.41 (0.30-0.55)	0.64 (0.42-0.96)	0.12	**2.7**
CB101	0.64 (0.53-0.77)	0.70 (0.56-0.88)	0.72 (0.59-0.88)	0.67 (0.54-0.82)	0.55 (0.43-0.69)	0.65 (0.55-0.76)	0.58 (0.48-0.71)	0.14	**3**
CB118	0.75 (0.61-0.92)	0.70 (0.56-0.89)	0.77 (0.61-0.95)	0.76 (0.61-0.94)	0.74 (0.58-0.95)	0.89 (0.75-1.06)	0.84 (0.66-1.07)	0.17	**0.6**
CB138	0.81 (0.71-0.93)	0.88 (0.75-1.04)	0.86 (0.74-1.00)	0.84 (0.72-0.98)	0.76 (0.64-0.90)	0.88 (0.78-0.99)	0.76 (0.65-0.88)	0.15	**7.9**
CB153	1.38 (1.11-1.73)	1.22 (0.95-1.58)	1.50 (1.19-1.89)	1.38 (1.09-1.73)	1.08 (0.83-1.41)	1.45 (1.21-1.75)	1.59 (1.23-2.05)	0.19	**40**
CB180	0.45 (0.35-0.57)	0.40 (0.31-0.53)	0.56 (0.43-0.72)	0.48 (0.38-0.62)	0.28 (0.21-0.37)	0.42 (0.34-0.51)	0.50 (0.38-0.66)	0.10	**12**
Hg	223 (190-263)	223 (190-263)	223 (190-263)	223 (190-263)	223 (190-263)	223 (190-263)	223 (190-263)	070	150
Cd	0.51 (0.39-0.66)	0.51 (0.39-0.66)	0.51 (0.39-0.66)	0.51 (0.39-0.66)	0.51 (0.39-0.66)	0.51 (0.39-0.66)	0.51 (0.39-0.66)	0.31	1.2
Pb	39.0 (34.4-44.3)	55.3 (47.5-64.2)	58.7 (50.9-67.6)	60.0 (51.6-69.8)	63.7 (53.4-75.9)	55.5 (49.7-62.0)	52.4 (45.5-60.4)	0.38	47

For all PAHs except chrysene, slight decreasing trends (2009-2018) can be found in sediment, which is also confirmed in swimming crab, whereas data in starfish and brown shrimp again give a more mixed view (Table 5.8). For indeno(123cd)pyrene, low values are noted, even below the expected BAC. For all other PAHs, values are above BAC, but clearly below the defined ERL (Table 5.6). Highest concentrations of individual PAHs are recorded at dumping sites OST and ZBO. These dumping sites are located near shore, have a high TOC content, and receive a relatively high amount of dumping from harbour Oostende and port Zeebrugge (De Witte et al. 2016). Higher concentrations of PAHs at these areas, can linked to the higher degree of PAH contamination at industrial ports and harbours (De Witte et al. 2016), higher shipping activity and larger impact of land-based atmospheric depositions in coastal, urbanised areas (Bester & Theobald 2000).

PCBs CB28, CB52 and CB118 reveal decreasing trends at the BPNS. The opposite is true for the heavier PCBs CB138, CB153, CB180 which have increasing concentrations up and around the dumping sites (Table 5.9). These results are consisted with the work of Le et al. (2021), who found a decreasing trend for all PCBs in the nineties at the BPNS, followed by a levelling off or increase from 2000 to 2015. Increased in the last decade could be linked to inputs from the port of Zeebrugge and the harbour of Oostende as well as the Scheldt estuary (Le et al. 2021). PCB concentrations are still clearly elevated compared to BAC. Moreover, proposed EAC values are exceeded for CB118 at all dumping sites as well as corresponding reference areas (Table 5.9).

Table 5.7: Summary of the monitoring data on inorganic contaminants on dumping sites for sediment and biota.

Compound	Time trend	Total modelled change (2007-2018)	Spatial effect	Remark	*L. holsatus/marmareus* (2007-2018)	*A. Rubens* (2007-2018)	*C. crangon* (2011-2018)
As	Increase	78%	No	-	ND	ND	ND
Cd	Decrease	-30%	No	-	<5% change	Area dependent (-76% up to +37%)	Increase (25%)
Cr	Decrease	-35%	No	-	Decrease (-36%)	Increase (111%)	Increase (54%)
Cu	Decrease	-18%	Yes	Lower concentration around ZBO, highest concentration around S2 and at NWP	Area dependent (-47% up to +25%)	Decrease (-10%)	Increase (16%)
Hg	Decrease	-32%	No	-	Decrease (-22%)	<5% change	Increase (7%)
Ni	Decrease	Area dependent (-18% up to -57%)	Yes	Higher concentration S2	ND	ND	ND
Pb	Decrease	-34%	Yes	Lower concentration at S1; highest concentrations at OST	Decrease (-46%)	Decrease (-5%)	<5% change
Zn	Decrease	-26%	Yes	Higher concentration around S2, lower concentration at S1	Increase (51%)	Area dependent (-18% up to +76%)	<5% change

Table 5.8: Summary of the monitoring data on PAHs on dumping sites for sediment and biota.

Compound	Time trend	Total modelled change (2009-2018)	Spatial effect	Remark	*L. holsatus/marmareus* (2013-2018)	*A. Rubens* (2013-2019)	*C. crangon* (2013-2019)
Naphthalene	Area dependent	Area dependent (-48% up to +27%)	Yes	-	ND	ND	ND
Phenanthrene	Decrease	-7%	Yes	Lower concentration at S1, higher concentration at OST and ZBO	Decrease (-52%)	Increase (142%)	Increase (20%)
Anthracene	Decrease	-13%	Yes	Lower concentration at S1, higher concentration at OST and ZBO	Decrease (-22%)	<5% change	Increase (26%)
Fluoranthene	Decrease	-23%	Yes	Lower concentration at S1, higher concentration at OST and ZBO	Decrease (-18%)	Increase (6%)	Decrease (-18%)
Pyrene	Decrease	-25%	Yes	Lower concentration at S1, higher concentration at OST and ZBO	Decrease (-41%)	Decrease (-26%)	Decrease (-37%)
Benzo[a]anthracene	Decrease	-22%	Yes	Lower concentration at S1, higher concentration at OST and ZBO	Decrease (-59%)	Decrease (-35%)	Decrease (-7%)
Chrysene	Increase	17%	Yes	Lower concentration at S1, higher concentration at OST and ZBO	Decrease (-52%)	Increase (23%)	Decrease (-88%)

Benzo[a]pyrene	Decrease	-4%	Yes	Lower concentration at S1, higher concentration at OST and ZBO	Decrease (-52%)	Increase (59%)	<5% change
Benzo[ghi]perylene	Decrease	-35%	Yes	Lower concentration at S1, higher concentration at OST and ZBO	Decrease (-83%)	Decrease (-47%)	Decrease (-40%)
Indeno[123cd]pyrene	Decrease	-32%	Yes	Lower concentration at S1, higher concentration at OST and ZBO	Area dependent (-84% up to +45%)	Increase (17%)	Increase (70%)

Table 5.9: Summary of the monitoring data on PCBs on dumping sites for sediment and biota.

Compound	Time trend	Total modelled change (2009-2018)	Spatial effect	Remark	*L. holsatus/marmareus* (2005-2018)	*A. Rubens* (2005-2018)	*C. crangon* (2007-2018)
CB28	Decrease	-54%	Yes	Lower concentration at S1, higher concentration at ZBO	Decrease (-80%)	Decrease (-40%)	<5% change
CB52	Area dependent	Area dependent (-66% up to +2%)	Yes	Trend towards more equally distri-bution over the different areas	ND	ND	ND
CB101	Area dependent	Area dependent (-58% up to +48%)	Yes	Trend towards more equally distri-bution over the different areas	Area dependent (-68% up to 36%)	Decrease (-56%)	Decrease (-70%)
CB118	Decrease	-27%	Yes	Highest concentration at S2	Decrease (-30%)	Decrease (-57%)	Decrease (-59%)
CB138	Increase	+140%	No	No significant spatial differences	Decrease (-22%)	Decrease (-50%)	Decrease (-83%)
CB153	Area dependent	Area dependent (-15% up to +122%)	Yes	Lowest concentrations at NWP	Decrease (-26%)	Decrease (-33%)	Decrease (-78%)
CB180	Area dependent	Area dependent (-51% up to +229%)	Yes	Higher increase at S2 compared to other areas	Decrease (-30%)	Decrease (-85%)	Decrease (-57%)

5.6 Evaluation of booster biocides at the dumping sites

Antifouling agents play a major role in the prevention of biofouling, a process in which organisms form biofilms on submerged vessels like ships. These biofilms often decrease the efficiency of shipping, resulting in a higher fuel consumption. In the 60', tributyltin (TBT) was introduced as an efficient antifouling agent, but due to its detrimental effect on the aquatic environment, the use of TBT was forbidden. Today's antifouling agents often use chemicals containing cupper. Because certain algae are resistant to cupper, anti-fouling agents are often used in combination with booster biocides. The effects of these booster biocides on the aquatic environment are still unknown. Therefore, monitoring these compounds is important to avoid irrevocable damage to current ecosystems (Cocquyt et al. 2019).

The concentration of six commonly used booster biocides (dichlofluanid, diuron, irgarol, medetomidine, Sea-Nine, and tolylfluanid) was measured at the five dumping sites of the BPNS and corresponding reference areas (REF) as well as at the harbours and ports of Oostende (HOO), Nieuwpoort (HNP), and Zeebrugge (HZB) (see Figure 5.4). Analysis was done by extraction sediment samples at 100°C by pressurized liquid extraction applying hexane:acetone 2:1 as solvent. For each sample, a non-spiked and spiked subsample were analysed and booster biocides were quantified applying standard addition to correct for matrix effects. All compounds were analysed by liquid chromatography-tandem mass spectrometry in multiple reaction monitoring mode. For dichlofluanid and tolylfluanid determination, a kinetix EVO C18 column was applied with acetonitrile and water (0.1% formic acid) as mobile phases and with atmospheric pressure ionisation as MS ionisation source. For diuron, irgarol, medetomidine and Sea-Nine 211, a kinetix C18 column was used with methanol and 10 mM ammonium acetate in water as mobile phases and with electrospray ionisation as MS ionisation source. In Tables 5.10 and 5.11 the minimum and maximum observed concentration of each booster biocide is reported for the dumping sites and harbours and ports, respectively. Values below the quantification limit are marked with LOQ. Following OSPAR (OSPAR 2011), concentrations are expressed as values normalized to 2.5% TOC. When maximum booster biocide concentrations were determined in samples with TOC below the quantification limit, half of the TOC quantification limit (i.e. 0.105%) was considered for calculating the normalised concentration. As this impacts the certainty of the measurement, these values are marked in red and noted with a [1].

Out of 6 analysed booster biocides, only diuron, irgarol, and medetomidine were detected at the different dumping sites. Diuron was detected at each site except at NWP. The observed maximum concentrations were between 1.9 and 2.8 times higher than the maximum concentration observed at the reference areas (Table 5.10). Irgarol was detected at dumping sites OST and ZBO. In contrast to diuron, a high concentration was also observed at 120, a reference location close to the port of Nieuwpoort. Medetomidine was detected at S1, S2, and ZBO. However, due to a low value of the measured TOC, the normalized values are only indicative.

Disposed sediments originate from the harbours or from shipping lanes. Therefore, we could expect that if a high concentration of a booster biocide is observed at a disposal site, a similar (or higher) concentration could be present in the corresponding harbour or port. High concentrations of diuron, up to 26 $ng.g^{-1}$ dry weight (DW), were observed at the harbour of Oostende and the port of Zeebrugge, which might be linked with relatively high hotspot concentrations at dumping sites OST, ZBO, S1 and S2. Remarkably lower concentrations were detected at harbour Nieuwpoort and NWP. For irgarol, the highest concentration was observed at the harbour of Oostende (12.8 $ng.g^{-1}$ DW). Detectable concentrations of irgarol were also found at harbour Nieuwpoort and the port of Zeebrugge. Medetomidine was detected at S1, S2 and ZBO, suggestion a link with the port of Zeebrugge, however, medetomidine was not detected at that port.

Although both dichlofluanid and Sea-Nine were not detected in dumping sites, detectable concentrations were measured in the harbours and ports of Oostende, Nieuwpoort, and Zeebrugge. A maximum concentration of Sea-Nine of 21.9 $ng.g^{-1}$ DW was observed at the port of Zeebrugge. Because the concentration of these 6 booster biocides strongly varies within one disposal site, monitoring these compounds at different locations is important to assess the impact of booster biocides on the marine environment.

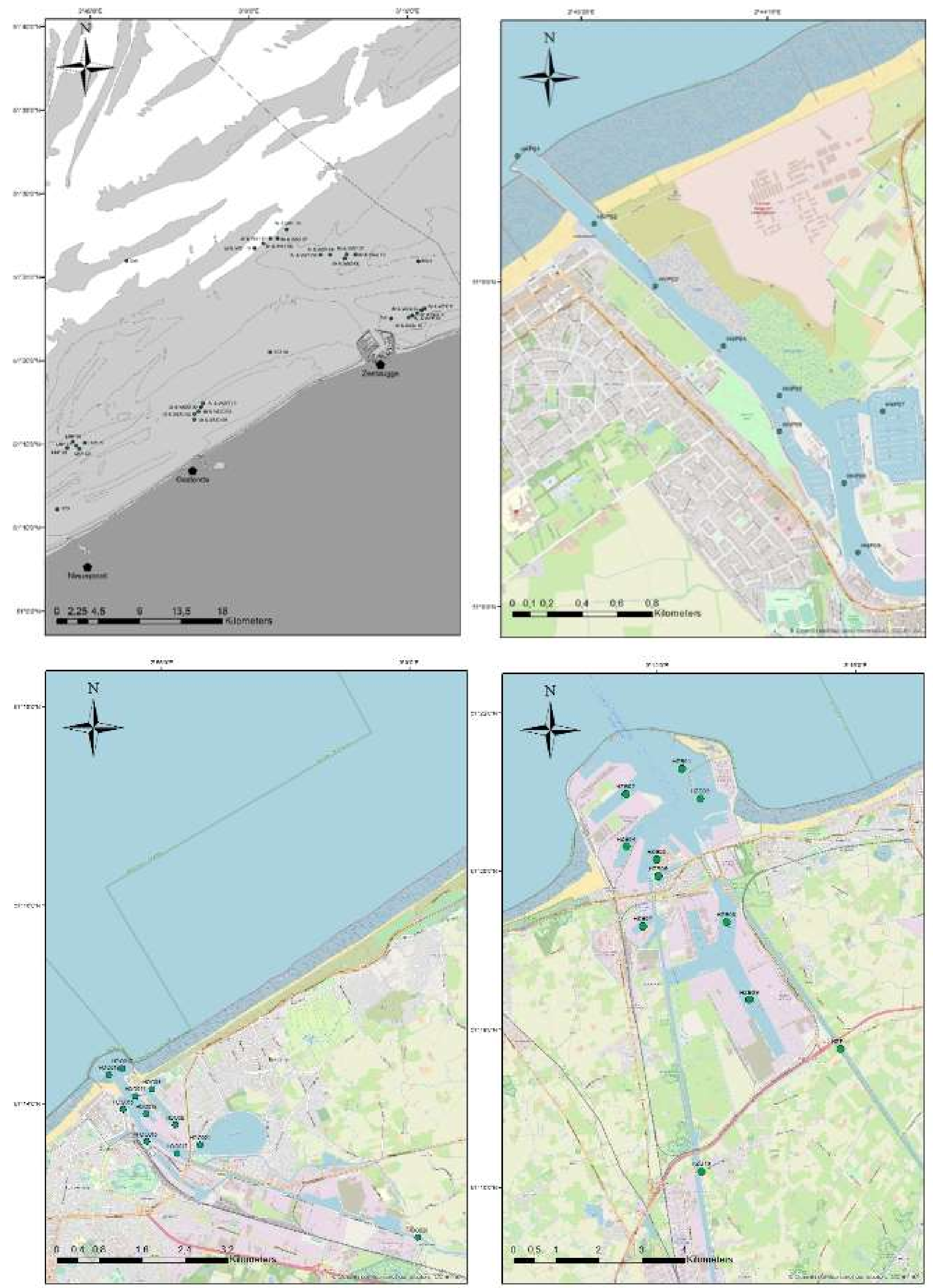

Figure 5.4: Top left panel: Overview of sampling locations at the five dumping sites. Right top panel: Overview of the sampling locations at the harbour of Nieuwpoort. Left bottom panel: Overview of the sampling locations at the harbour of Oostende. Right bottom panel: overview sampling locations at the port of Zeebrugge.

Table 5.10: The minimum and maximum observed normalised concentration (ng/g DW) of different booster biocides at 5 dumping sites and different reference locations. Concentrations below the detection limit are marked with LOQ and concentrations that are above the detection limit but with a TOC value below the detection limit are marked in red and noted with [1].

	Dichlofluanid	Diuron	Irgarol	Medetomidine	Sea-Nine	Tolylfluanid
NWP	[<LOQ]	[<LOQ]	[<LOQ]	[<LOQ]	[<LOQ]	[<LOQ]
OST	[<LOQ]	[LOQ-0.90]	[LOQ-0.32]	[<LOQ]	[<LOQ]	[<LOQ]
S1	[<LOQ]	[LOQ-0.91]	[<LOQ]	[LOQ-4.6]	[<LOQ]	[<LOQ]
S2	[<LOQ]	[LOQ-0.87]	[<LOQ]	[LOQ-28.3][1]	[<LOQ]	[<LOQ]
ZBO	[<LOQ]	[LOQ-0.61]	[LOQ-0.13]	[LOQ-31.6][1]	[<LOQ]	[<LOQ]
REF	[<LOQ]	[LOQ-0.32]	[LOQ-0.65]	[<LOQ]	[<LOQ]	[<LOQ]

Table 5.11: The minimum and maximum observed normalised concentration (ng/g DW) of different booster biocides at the ports of Zeebrugge (HZB), Oostende (HOO), and Nieuwpoort (HNP). Concentrations below the detection limit are marker with LOQ.

	Dichlofluanid	Diuron	Irgarol	Medetomidine	Sea-Nine	Tolylfluanid
HNP	[LOQ-5.6]	[LOQ-0.84]	[LOQ-9.7]	[<LOQ]	[<LOQ]	[<LOQ]
HOO	[LOQ-3.2]	[0.28-26.6]	[0.12-12.7]	[<LOQ]	[LOQ-2.5]	[<LOQ]
HZB	[<LOQ]	[LOQ-9.7]	[LOQ-3.5]	[<LOQ]	[LOQ-21.9]	[<LOQ]

To evaluate the environmental risk associated with the presence of booster biocides at dredge material dumping sites, a risk assessment was performed based on the concentrations measured in the samples from 2018. For OST, ZBO, and NWP, the median measured concentration was considered and the corresponding booster biocide concentration in the water phase was calculated considering the organic carbon partition coefficient K_{OC} and the TOC content (Cocquyt et al. 2019). The predicted no effect concentration (PNEC), i.e. the concentration below which no negative environmental effects can be expected, was also calculated based on available toxicity data and the construction of species sensitivity distributions (Cocquyt et al. 2019). The risk characterisation ratio (RCR) is the ratio between the measured concentration (MEC) and the PNEC. A value above 1 indicates that the concentration is higher than the environmentally save concentration.

Results are presented at Table 5.12. No risk is observed for dichlofluanid and Sea-Nine, as the median concentration measured was below the quantification limit and even when concentrations would reach the quantification limit, the RCR would be below 1. For diuron and irgarol, however, RCR values above 1 were noted, even up to 188 for irgarol at OST, indicating a potential environmental risk. Close follow-up of diuron and irgarol concentrations in the marine environment is therefore indispensable in future monitoring.

Table 5.12:The risk characterisation ratio for different booster biocides at different dumping sites.

	Dichlofluanid	Diuron	Irgarol	Sea-Nine
NWP	< 0.27	<0.30	<5.5	<0.008
OST	< 0.78	1.40	188	<0.005
ZBO	< 0.14	0.37	14.1	<0.0004

Although tributyltin has been banned since 2008, high concentrations are still observed at the harbours and ports of Oostende, Nieuwpoort, and Zeebrugge. The results are reported in *Table 5.13*. In Oostende and Zeebrugge, high concentrations of TBT, up to 580 $ng.g^{-1}$, were observed in two specific areas: Visserijdok (Oostende) and Prins Filipsdok (Zeebrugge). At visserijdok, the sediments were not dredged, allowing the detection of historical TBT accumulation. Concentrations of degradation products dibutyltin (DBT) and monobutyltin (MBT) are also provided in Table 5.13.

At the dumping sites, normalised concentrations of TBT vary between 0.60 and 4.24 $ng.g^{-1}$ DW. The Swedish EPA recently proposed an EAC for TBT of 1.6 $ng.g^{-1}$ DW normalised to 5% TOC. When normalised to 2.5% TOC, this results in an EAC value of 3.2 $ng.g^{-1}$ DW (Sahlin & Ågerstrand 2018). Measurements of TBT at dumping sites and reference areas at the BPNS are around the proposed EAC, indicating a potential environmental risk of TBT contamination. Due to the hazardous properties of TBT, further monitoring of this substance is required.

Booster biocides diuron and irgarol can be detected in several dumping sites and in the harbours and ports of Oostende, Nieuwpoort and Zeebrugge at concentrations higher than the risk characterisation ratio. Also, TBT can be ubiquitously found in the marine environment, with values around the proposed environmental assessment concentration. This indicates the importance to continue monitoring TBT and anti-fouling booster biocides.

Table 5.13: The minimum and maximum observed normalised concentration (ng/g DW) of tributyltin (TBT), dibutyltin (DBT), and monobutyltin (MBT) at the ports of Oostende (HOO), Nieuwpoort (HNP), Zeebrugge (HZB) and at different dumping sites and reference areas.

	TBT	**DBT**	**MBT**
HOO	[1.50-581]	[1.20-258]	[0.10-1.52]
HNP	[0.59-22.9]	[0.76-2.31]	[LOQ-0.32]
HZB	[1.48-184]	[1.42-74.9]	[LOQ-0.87]
OST	[1.36-2.06]	-	[0.23-0.41]
NWP	[0.79-1.95]	-	[0.14-0.25]
S1	[1.34-4.24]	-	[0.11-0.29]
S2	[1.21-2.53]	[1.21-2.29]	[LOQ-0.55]
ZBO	[2.27-3.27]	[1.86-2.92]	[LOQ-0.38]
REF	[0.6-3.40]	[0.50-2.14]	[LOQ-0.42]

5.7 Macrolitter distribution on the seafloor at and around the dumping sites

Vast quantities of plastic litter enter the ocean that may harm marine ecosystems (Jambeck et al. 2015; Galgani et al. 2021). Litter may scour or smother the seafloor, which may impact fragile benthic habitats, reduce photosynthesis and prevent the movement of animals, gases, and nutrients. Marine litter may also act as a vector for invasive species, transporting non-indigenous organisms into new areas where they can outcompete or prey upon native organisms (OSPAR 2017b). For these reasons, marine litter has been recognized as a global environmental concern. Within the Marine Strategy Framework Directive (MSFD), a primary aim is that composition, amount, and spatial distribution of litter on the coastline, in the surface layer of the water column, and on the seafloor are at levels that do not cause harm to the coastal and marine environment (2008/56/EG; 2017/848/EU). Therefore, monitoring of seabed litter is essential and is incorporated in the monitoring of dumping sites.

For the assessment of seafloor litter, litter is collected within the net, using an 8 m beam trawl with a cod-end mesh size of 20 mm (stretched) and a length of 1 nautical mile. Data is collected according to OSPAR (2017a) and MSFD (JRC, 2013) guidelines. Seafloor litter data was collected between March 2013 and March 2019. The collected litter was classified according to OSPAR (2017a) into six different categories: plastic, metals, rubber, glass and ceramics, natural (manmade) products, and miscellaneous objects.

Considering all fish tracks taking within the ILVO environmental monitoring campaigns at the BPNS from 2013 to 2019, 88% of all litter items caught in the net were plastic. The largest number of litter items can be found at the coastal zone, with on average 12.7 ± 1.7 litter items per ha, compared to 2.8 ± 0.2 litter items per ha outside the 12 nm zone at the BPNS. This indicates an impact from land-based sources of marine litter or from marine activities within the 12 nm. However, current patterns and sedimentation may also play a role in the accumulation of litter. At the eastern part of the BPNS, the coastal area is a known sedimentation area (Fettweis et al. 2009). Sedimentation will increase when water velocities are low, resulting in fine sediments close to the coast. This will also affect litter settling in a coastal environment, with higher amounts of litter in mud to muddy sand regions.

Table 5.14: Average total litter contamination at different dumping sites of the BPNS, pooled data, 2013-2019 (N=191).

Location	Type	Number of items	Number of tracks	Average number of items/ha	Minimum number of items/ha	Maximum number of items/ha
	Impact	90	9	6.8 ± 3.3	1.5	13.0
NWP	Nearby	79	10	5.4 ± 3.1	2.0	10.2
	Reference	252	14	12.4 ± 15.1	0	59.9
	Impact	330	10	22.2 ± 24.6	5.2	76.1
OST	Nearby	177	7	16.5 ± 15.8	4.1	50.2
	Reference	250	17	10.0 ± 16.2	0.6	70.7
	Impact	253	17	9.8 ± 13.5	1.4	52.3
S1	Nearby	111	9	8.4 ± 4.3	2.7	15.7
	Reference	252	14	12.4 ± 15.1	0	59.9
	Impact	316	11	19.0 ± 21.2	1.3	68.2
S2	Nearby	270	12	15.2 ± 20.3	1.6	68.8
	Reference	379	23	11.5 ± 14.1	0.6	58.3
	Impact	625	9	61.4 ± 79.2	3.9	252.9
ZBO	Nearby	131	6	15.0 ± 14.6	4.2	43.6
	Reference	379	23	11.5 ± 14.1	0.6	58.3

The effect of dredging on total litter contamination in the BPNS was assessed, comparing impacted areas with reference areas and nearby areas.

The dumping sites are all located within the 12 nm zone. As a result, a higher amount of average litter items is detected in the net (21.6 ± 38.6 items/ha) compared to offshore areas. Between the 5 dumping sites, large differences can be seen with the highest average of 61.4 ± 79.2 litter items/ha at the impact zone of dumping site ZBO and a lowest average value of 6.8 ± 3.3 litter items/ha at NWP (Table 5.14). Again, it is impossible to link differences univocally to 1 factor, as not only the dumping intensity differs between the sites but also the sedimentation rate is different. At NWP, dumping intensity was 0.08 ton dry matter (DM).m^{-2}.$year^{-1}$ from 2007 until 2017 while a much higher value of 1.84 ton DM.m^{-2}.$year^{-1}$ was recorded at ZBO (Lauwaert et al. 2019). However, differences between dumping sites are not a prove that dumping impacts the marine litter input as NWP is located on a sandy area at the western part of the BPNS, while dumping site ZBO is located at a sedimentation area at the eastern part of the BPNS, classified as “mud to muddy sand”.

At dumping sites NWP, OST, S1, and S2, no clear difference can be found between impact, nearby and reference areas (Table 5.14). At ZBO, however, 61.4 ± 79.2 litter items/ha were caught in the net at the impact area, compared to 15.0 ± 14.6 and 11.5 ± 14.1 litter items/ha at nearby and reference areas, respectively. As the nearby zone is also a sedimentation zone, this data suggests a significant input of litter from the dredging activities at ZBO. ZBO is a dumping site, mainly receiving dredge disposal from the port of Zeebrugge. It is also the site with the highest dumping intensity (Lauwaert et al. 2019). A more detailed source investigation at this local litter hotspot is therefore recommended.

5.8 Recommendations and new actions

5.8.1. Recommendations

Based on our extensive monitoring of the ecological and chemical status of the 5 dredge disposal sites, we can make following recommendations in relation to managing the sites:

- In relation to the ecological status, only the fauna in S1 site is really deteriorated due to the high dumping intensity. This impact is mainly expressed for the macrofauna (animals living in the sediment), but also for epi- and fish some impact is observed (lower densities). Those patterns are also confirmed by the sediment profile imaging (SPI) analyses, which reveal clear visual signs of degraded sediments (mud clasts, no fauna). Therefore, S1 site is classified under the MSFD assessment as not in good status (even bad-poor in relation to densities and biomass of species). Remedial actions are maybe not necessary (e.g., size of site impacted), but such high dumping intensities in *Abra alba* habitat at other sites should be avoided in the future.

- In relation to managing the dumping strategy in Belgian waters, we need to ensure that the intensities in dumped material are in line with what the environment can scope and that the dumped material is ideally in line with the natural occurring sediments. For example, at site S2 it is not ideal, as the dumping of muddy sediments in a sandy environment led even to mixed patterns (increase in species diversity, but effect on density, biomass). On the other hand, the intensity and type of dumped material at OST and ZBO is in line with what the environment can cope.
- Based on the comparison of the impact, near-by control and control samples, we observe at most sites (S1, S2, OST and ZBO) some side effects. Therefore, it is uncertain how far the dumping activity influence and change the surrounding environment at each site. A more in depth investigation is needed to evaluate this aspect, to determine that the possible impact of dredge disposal is within the delignated areas. This is important in relation to the MSFD assessment, descriptor 6 sea-floor integrity, where the habitat extent in physical loss and disturbance need to be quantified.
- The reference areas ZEB and ZVL are probably not containing a healthy *L. balthica* community, as the diversity is very low, compared to other muddy areas (140bis area). Therefore, the use of this control areas needs to be evaluated and eventually new areas explored. Also, due to the fact that the species richness seems to increase in last years in the *L.balthica* community, but less or not at ZEB and ZVL.
- Seafloor litter concentrations are much higher at ZBO compared to other dumping sites or reference areas. The main driving force for the high degree of litter contamination at this location remains unclear. A detailed study to elucidate the relative effect of dumping versus the role of hydrodynamic processes such as sedimentation is recommended.
- Within regular monitoring of contaminants at dumping sites, PAHs and most PCBs and metals revealed decreasing trends in marine sediments. However, contaminant concentrations are still significantly higher than background assessment concentrations. Moreover, concentrations of Hg, Pb and CB118 even exceed the environmental assessment criteria, indicating a potential environmental risk and the need for a close follow-up of these contaminants. Increasing concentrations of As and heavier PCBs (CB138, CB 153, CB180) in marine sediments and Cr in marine biota also stress the need for continuation of chemical monitoring at dumping sites of the BPNS.
- Antifouling paints can be an important marine-based source of inorganic and organic contaminants to the marine environment due to the use of biocides and Cu- and Zn-based paints. Our work suggests potential environmental issues through the historic use of biocides. Concentrations of the banned antifouling agent TBT at dumping sites of the BPNS are around the proposed environmental assessment criterion. At defined locations in the marine environment, concentrations of antifouling booster biocides irgarol and diuron are higher than the risk characterization ratio. A sustainable use of marine anti-fouling agents needs to be integrated in marine management.
- We recommend an update on the requirements for chemical analysis on sediments before dredging. This includes defining analytical requirements such as the minimum required detection limit of the analytical method, as well as recommendations for sampling design.
- A close follow-up on revisions of contaminant guidelines, ongoing at European level, is needed. Different national guidelines are in use for the management of dredged material in the EU countries. Guidelines differ (1) in the elements and organic compounds which should be necessarily analysed, (2) in the applied threshold values, (3) in the applied action levels, (4) in the grain size fractions where metals and organic compounds are analysed (ICES 2021). Initiatives are taken by the European Sediment Network (Sednet) to better align national regulations. A close follow up of ongoing initiatives is advised.

5.8.2. New actions for ILVO research program 2022-2027

Some of the recommendation mentioned above need further investigation or follow-up. This will be taken forward in the ILVO-research program on dredge disposal activities for the period 2022-2027. In this period, the basic monitoring will be continued and even complemented with other observation techniques (e.g. sediment profile imaging). Following actions will be considered:

Monitoring of the marine ecosystem at the disposal sites:

- The monitoring of the macrobenthos, epibenthos and fish fauna at the five dredge disposal sites will be continued based on a cyclic programme. Dumping site S1 and ZBO will be followed-up yearly, whereas the three other sites (NWP, S2 and OST) ones every three years.
- SPI – monitoring was in detailed performed on S1 (and S2) as test, which will be repeated for those and performed for the other dumping sites (if practical possible, see LZO) as well. Therefore, two transects in combination with the traditional sampling points will be monitored with SPI. This more detailed survey is need as we observe quite some variation in sedimentological (and ecological) characteristics, so it should allow us to have a better insight in the local spatial variation. Such information cannot be obtained by grab sampling only.
- We will explore the use of genomic based monitoring in the evaluation of the impact of dredge disposal on the marine environment. Currently, research on this is going on in the Interreg project GEANS (2019-2023; https://northsearegion.eu/geans/), where they develop a genetic tool for ecosystem health assessment in the North Sea region. A test case for the impact evaluation at one disposal site will be developed.
- The monitoring of chemical contaminants PAHs, PCBs, metals and TBT will be continued at the 5 dredge disposal sites. Sediment samples will be taken yearly. The sampling scheme for biota (swimming crab, brown shrimp, starfish) will be based on the available ship time, but yearly collecting 2 samples of each species for each dumping site is intended.
- The distribution of booster biocides in marine sediments will be measured on a frequent basis and an updated risk assessment will be performed.
- The spatial distribution of seafloor litter at dumping sites will monitored. As seafloor litter is assessed by counting litter in the net, this will only be done at dumping sites where fishing activities are practically feasible.

Specific research questions:

- Clear spatial differences could be found in seafloor litter concentrations, with highest amounts at ZBO. In the next monitoring cycle, it will be investigated if seafloor litter contamination directly links with microlitter contamination.
- In previous monitoring cycle (2016-2021), antifouling contaminants were identified by a targeted analysis approach. However, also other, currently unknown, contaminants might occur at higher concentrations at dumping sites. An untargeted screening approach will be envisaged to identify contaminants of emerging concern and to evaluate the chemical status of the different dumping sites.
- The general changes in structure and function of the benthic communities at the disposal sites are well-described and are depending on the natural environment, dumping intensity, type of dumped material and sensitivity of the benthic species. Next step to unravel is which key species (and their functional traits) are really influenced (positive, neutral or negative) to the sedimentological changes caused by dredge disposal. Therefore, more in depth analyses (e.g. modelling of species-specific responses) will be performed on the monitoring data to quantify this.

Actions in supporting the management:

- Better and inclusive integration of dredge disposal data in other portals on national (BMDC) and international level (OSPAR, Emodnet, ICES datras). Therefore, a data management plan can be set-up to ensure and guide this process.
- Belgium is required to execute a MSFD assessment towards 2024, where the dredge disposal monitoring contribute to. Therefore, we will further support the research (monitoring, evaluation) and report needs in function of the Marine Strategy Framework Directive.
- European initiatives on revisions of contaminant guidelines will be followed up and a proposal to update the requirements for chemical analysis on sediments before dredging will be done.

6. Implemented projects and studies

A new dredge disposal site Zeebrugge West (ZBW) was designated at the western part of the harbour of Zeebrugge (Figure 6.1). Two studies have been executed (see below).

6.1 Longterm test on technical feasibility of ZBW dumping site

The results can be found in Lauwaert et al. (2019, §6.4) and in Arcadis (2017, 2018).

6.2 Ecological characterization of new dumping site ZBW

In order to describe the ecology and sedimentology of this site, a T0 monitoring campaign was performed in the autumn of 2018 for macrobenthos and during winter and autumn in 2018, 2019 and 2020 for epibenthos and demersal fish. For macrobenthos, 7 (LWO.01-LWO.07) and 9 (LWO.11-LWO.19) stations were predefined as impact and control area respectively (*Figure 6.1*B). For epibenthos and fish, 1 fish track was executed inside the predefined impact site and 1 track in the control area. The biota in the samples was analyzed and the taxa richness, Shannon Wiener index, density and biomass was calculated. Also, the median grain size and mud content was determined for each sample. Afterwards, a cluster analysis based on 25 % similarity between the samples, was performed. After defining the clusters, the average median grain size, mud content, taxa richness, Shannon Wiener index, density and biomass was calculated for each cluster and also for the predefined impact and control samples.

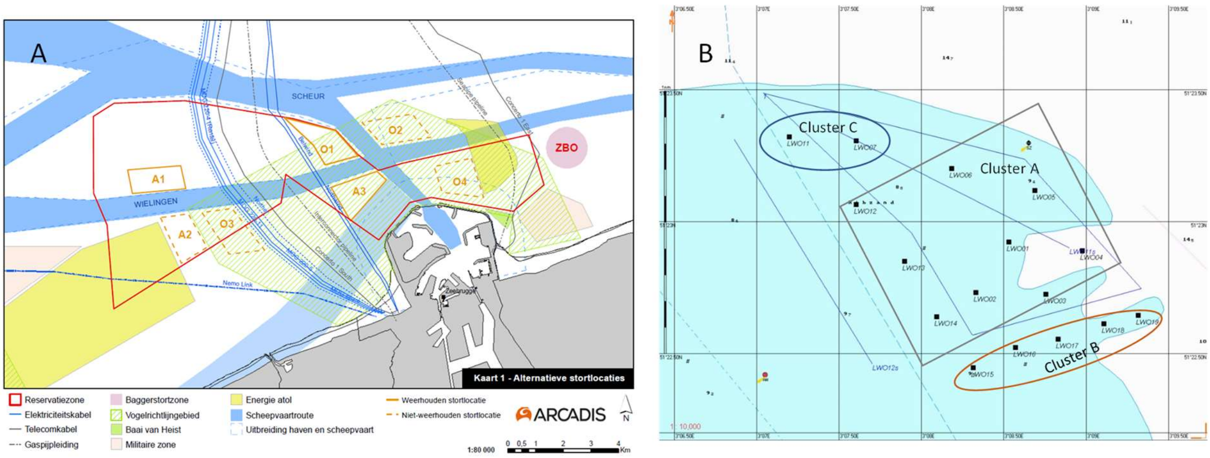

Figure 6.1. A) Overview of the possible new dumping locations for ZBO. O1 is eventually the dumping site ZBW (Arcadis, 2018). B) Overview of the sampling points and epi-fish tracks, with indication of the 3 clusters, defined on the basis of specific ecological and sedimentological characteristics.

Based on the cluster analysis and non-metric multidimensional scaling, the predefined impact and control macrobenthos samples were not distinctive. Instead, three clusters (A, B and C) of three sub-areas were distinguished, forming a gradient in characteristics from the north to the south of the area (*Figure 6.1*B). The top of the area is characterized by coarse sand with no mud and low species richness and density-biomass. The second sub-area (cluster A), is the south part of the dumping site and the west control sites, which both shows similar characteristics. The mud content of those samples is around 15.2 %, but with high average median grain size (392 µm). This indicate that this area is characterized by coarse sediment and mobile mud, but a proper seafloor characterization by sonar (multibeam, side scan) is required to better map this variation. In relation to the ecology, this area is poor in relation to biodiversity and density-biomass (*Table 6.1*) and indeed characterized by coarse sediment species (cf *Micropthalmus* spp.) and some typical muddy ones (Oligochaeta spp.). The third sub-area (cluster B) is defined by the south control stations, which were characterized by a high mud content (mostly cohesive muds), as indicated by the high densities of the American piddock (*Petricolaria pholadiformis;* alien species), which lives buried in “hard” substrates. The species richness is also clearly higher than within the two other sub-areas, with some extreme samples (species richness of 26 at LWO.17 and 27 at LWO.19), characterized by the tube-building polychaete *Lanice conchilega*.

The epibenthic and demersal fish community did not differ between the predefined impact and control sites. However, two clusters containing consisting of the winter and autumn samples were distinguished, due to the lower species richness and densities of both epibenthos and fish during winter, compared to autumn samples.

Based on these findings, monitoring of the benthic community at the new dredge disposal site Zeebrugge-West should be adapted in the future. The choice of impact and control stations should be reconsidered, taking the different sub-areas into account. Especially the sub-area in the southern part of the disposal site was ecologically and sedimentological different.

Table 6.1: Average median grain size, mud content, species richness, Shannon Wiener index H', density and biomass (+ standard errors) at the tentative impact and control stations and cluster groups of stations (A, B, C), as defined by a cluster analysis (based on 25 % similarity of macrobenthic species) at the site Zeebrugge-West (ZBW).

Group	**Stations**	**D50 grain size (µm)**	**Mud content (%)**	**Species richness**	**Shannon Wiener index H'**	**Density (ind./m²)**	**Biomass (g/m²)**
Impact	LWO.01-LWO.07	355 ± 65.7	15.6 ± 6.4	9 ± 0.8	2.5 ± 0.2	330 ± 126.2	13.8 ± 5.3
Control	LWO.11-LWO.19	265 ± 60	25.9 ± 7.5	13.7 ± 2.5	2.3 ± 0.3	1298.9 ± 472.9	399.8 ± 204
Cluster A	LWO.01-LWO.06; LWO.12-LWO.14	392.4 ± 53.4	15.2 ± 5	9 ± 0.7	2.4 ± 0.2	363.3 ± 99.4	12.3 ± 5.2
Cluster B	LWO.15-LWO.19	122.6 ± 34.5	40.4 ± 8.6	17.8 ± 3.6	2.4 ± 0.5	2088 ± 669.7	702.9 ± 313.2
Cluster C	LWO.07; LWO.11	362.8 ± 15.2	1.5 ± 0.02	8 ± 2	2.8 ± 0.4	145 ± 55	34.9 ± 7.1

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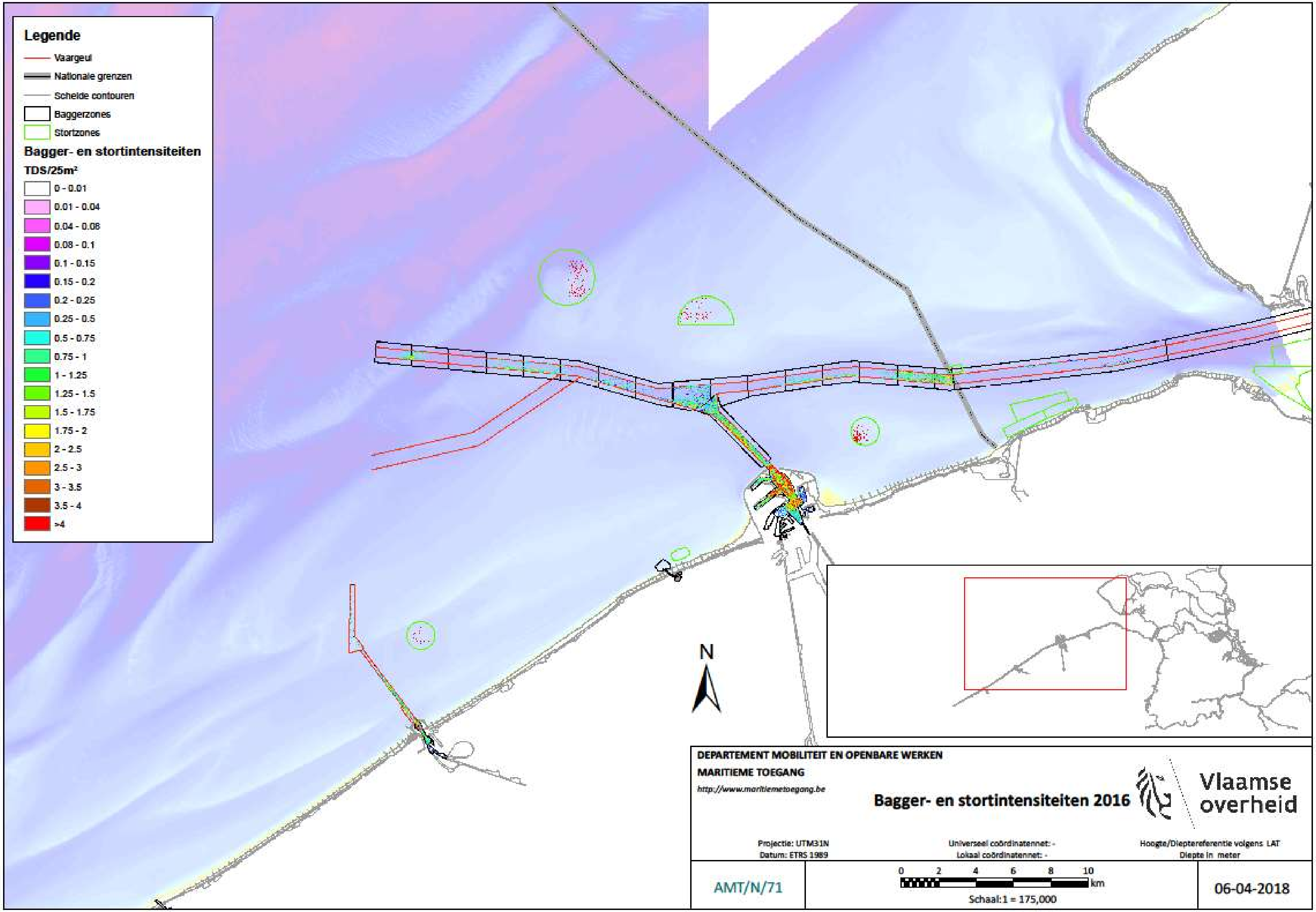
Abbreviations and definitions

ADV	Acoustic Doppler Velocimeter, measured current velocity in a vertical profile.
BAC	Background Assessment Concentrations
BCS	Belgian Continental Shelf
BEQI	Benthic Ecosystem Quality Index, www.beqi.eu
BPNS	Belgian Part of the North Sea
DMP	actual dumping site
d.w.	dry weight
EAC	Environmental Assessment Criteria
EIA	Environmental Impact Assessment
ERL	Effect Range Low values as developed by US-EPA
EQR	Ecological Quality Ratio
EQS	Environmental Quality Standard
FDI	Fish Disease Index
GES	Good Environmental Status
HCB	hexachlorobenzene
HCBD	hexachlorobutadiene
IMZ	directly impacted zone outside but less than 0.3 nautical mile away from the DMP
Ind	individuals
LISST	Laser In-Situ Scattering and Transmissometer, measured particle size distribution and volume concentration
mab	meter above bed
MRP	Marine Spatial Plan
MSFD	Marine Strategy Framework Directive
OBS	Optical Backscatter Sensor, measures turbidity
PAH	polycyclic aromatic hydrocarbons
PCB	polychlorobiphenyls
POC	Particulate Organic Carbon
PON	Particulate Organic Nitrogen
PSD	Particles Size Distribution
REF	reference samples taken on longer distance from the dumping site than IMZ
SPM	Suspended Particulate Matter
TDM	Ton Dry Matter
TEP	Transparent Exopolymer Particles
TOC	Total Organic Carbon
VITO	Vlaams Instituut voor Technologisch Onderzoek
WFD	Water Framework Directive
ww	wet weight

Appendix 1: Dredging and dumping intensity maps



Legende
Vaargeul
Nationale grenzen
Schelde contouren
Baggerzones
Stortzones
Bagger- en stortintensiteiten
TDS/25m²
0 - 0.01
0.01 - 0.04
0.04 - 0.08
0.08 - 0.1
0.1 - 0.15
0.15 - 0.2
0.2 - 0.25
0.25 - 0.5
0.5 - 0.75
0.75 - 1
1 - 1.25
1.25 - 1.5
1.5 - 1.75
1.75 - 2
2 - 2.5
2.5 - 3
3 - 3.5
3.5 - 4
>4
N
DEPARTEMENT MOBILITEIT EN OPENBARE WERKEN
MARITIEME TOEGANG
http://www.maritiemetoegang.be
Bagger- en stortintensiteiten 2016
Vlaamse overheid
Projectie: UTM31N
Datum: ETRS 1989
Universeel coördinatennet: -
Lokaal coördinatennet: -
Hoogte/Dieptereferentie volgens LAT
Diepte in meter
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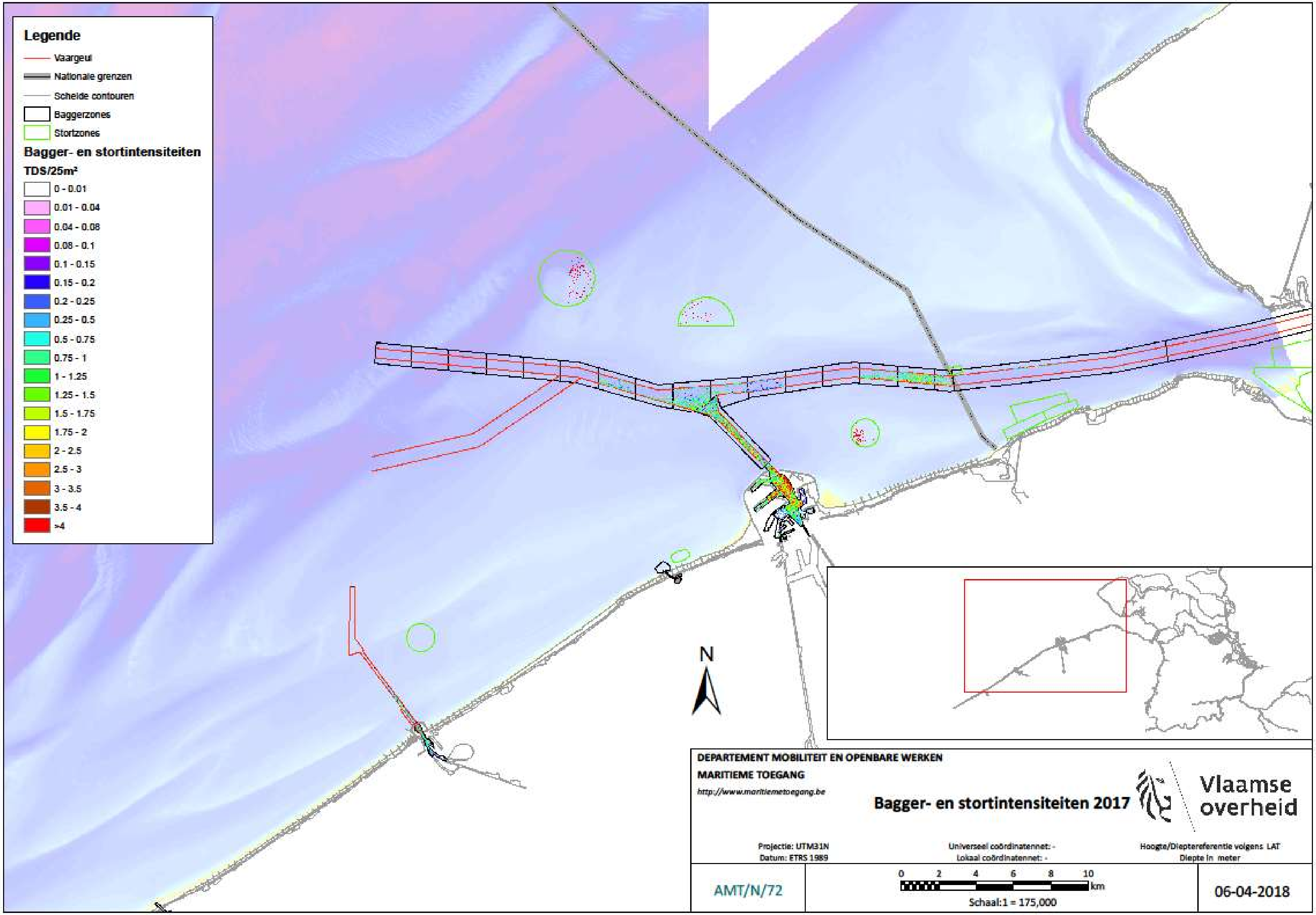
km
Schaal:1 = 175,000
06-04-2018



Legende
Vaargeul
Nationale grenzen
Schelde contouren
Baggerzones
Stortzones
Bagger- en stortintensiteiten
TDS/25m²
0 - 0.01
0.01 - 0.04
0.04 - 0.08
0.08 - 0.1
0.1 - 0.15
0.15 - 0.2
0.2 - 0.25
0.25 - 0.5
0.5 - 0.75
0.75 - 1
1 - 1.25
1.25 - 1.5
1.5 - 1.75
1.75 - 2
2 - 2.5
2.5 - 3
3 - 3.5
3.5 - 4
>4
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DEPARTEMENT MOBILITEIT EN OPENBARE WERKEN
MARITIEME TOEGANG
http://www.maritiemetoegang.be
Bagger- en stortintensiteiten 2017
Vlaamse overheid
Projectie: UTM31N
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Lokaal coördinatennet: -
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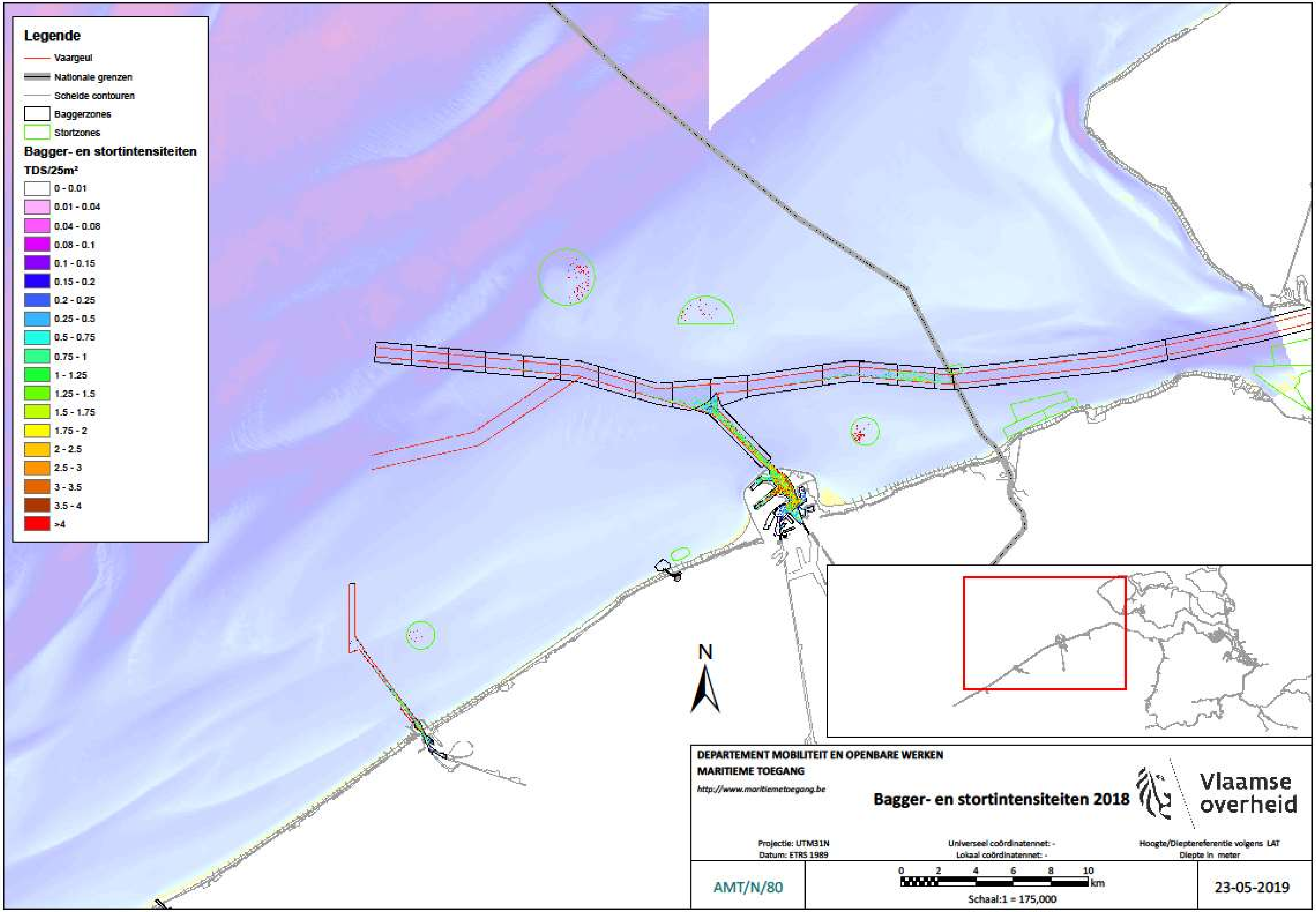
06-04-2018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Vaargeul
Nationale grenzen
Schelde contouren
Baggerzones
Stortzones
Bagger- en stortintensiteiten
TDS/25m²
0 - 0.01
0.01 - 0.04
0.04 - 0.08
0.08 - 0.1
0.1 - 0.15
0.15 - 0.2
0.2 - 0.25
0.25 - 0.5
0.5 - 0.75
0.75 - 1
1 - 1.25
1.25 - 1.5
1.5 - 1.75
1.75 - 2
2 - 2.5
2.5 - 3
3 - 3.5
3.5 - 4
>4
N
DEPARTEMENT MOBILITEIT EN OPENBARE WERKEN
MARITIEME TOEGANG
http://www.maritiemetoegang.be
Bagger- en stortintensiteiten 2018
Vlaamse overheid
Projectie: UTM31N
Datum: ETRS 1989
Universeel coördinatennet: -
Lokaal coördinatennet: -
Hoogte/Dieptereferentie volgens LAT
Diepte in meter
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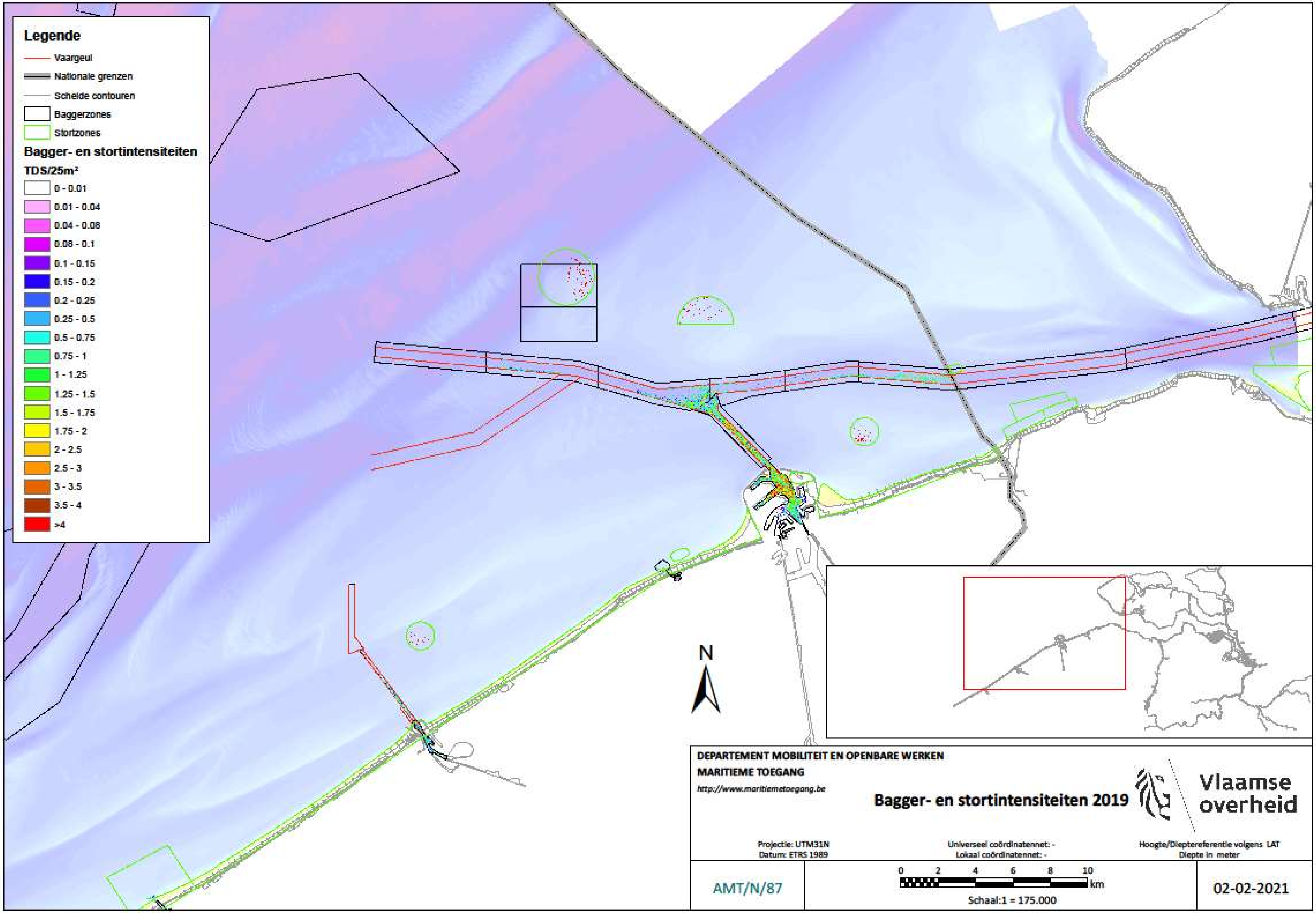
23-05-2019



DEPARTEMENT MOBILITEIT EN OPENBARE WERKEN
MARITIEME TOEGANG
http://www.maritiemetoegang.be

Bagger- en stortintensiteiten 2019

Vlaamse overheid

Projectie: UTM31N
Datum: ETRS 1989

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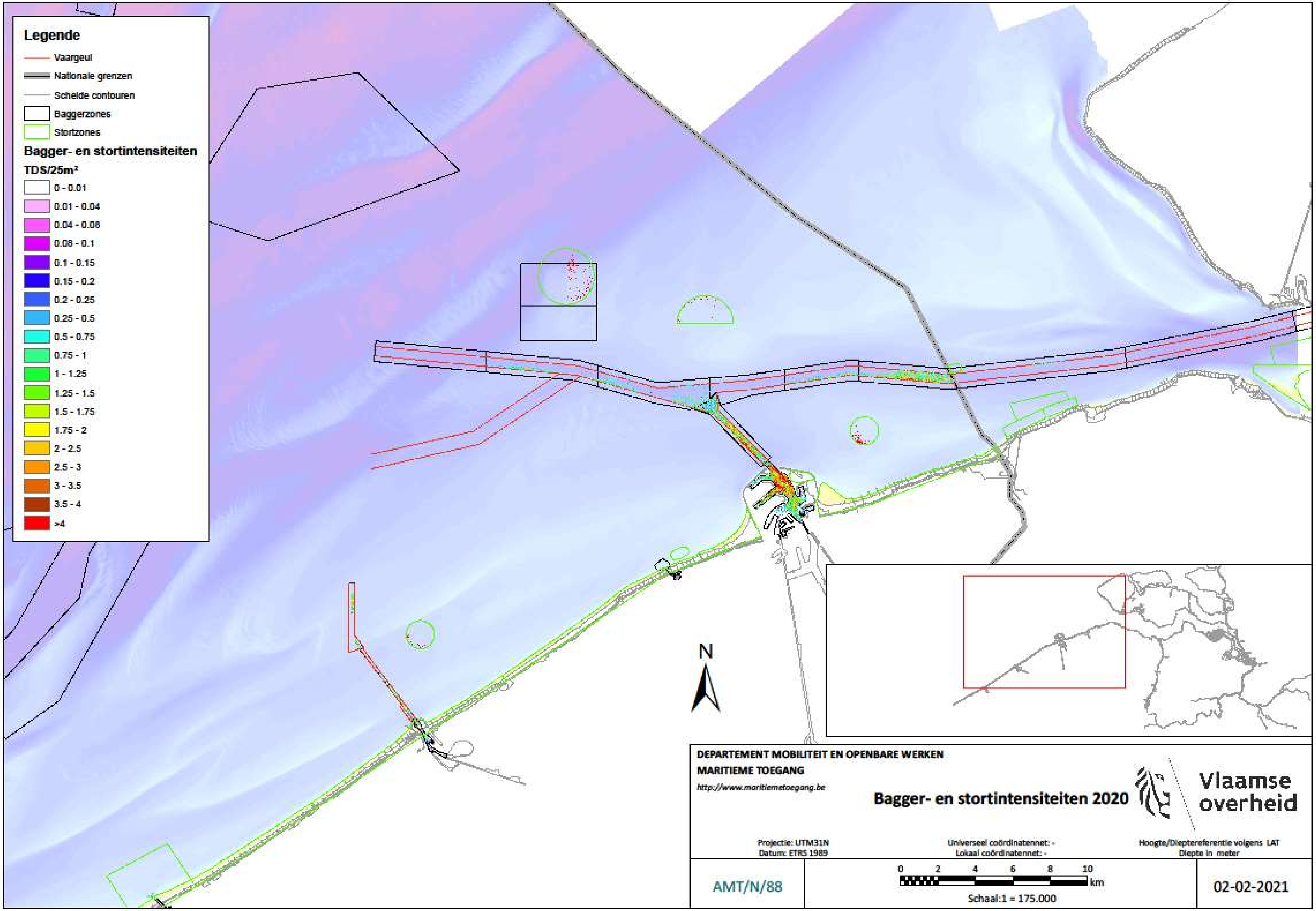
Hoogte/Dieptereferentie volgens LAT
Diepte in meter

AMT/N/87

Schaal:1 = 175.000

02-02-2021


Legende
Vaargeul
Nationale grenzen
Schelde contouren
Baggerzones
Stortzones
Bagger- en stortintensiteiten
TDS/25m²
0 - 0.01
0.01 - 0.04
0.04 - 0.08
0.08 - 0.1
0.1 - 0.15
0.15 - 0.2
0.2 - 0.25
0.25 - 0.5
0.5 - 0.75
0.75 - 1
1 - 1.25
1.25 - 1.5
1.5 - 1.75
1.75 - 2
2 - 2.5
2.5 - 3
3 - 3.5
3.5 - 4
>4
N
DEPARTEMENT MOBILITEIT EN OPENBARE WERKEN
MARITIEME TOEGANG
http://www.maritiemetoegang.be
Bagger- en stortintensiteiten 2020
Vlaamse overheid
Projectie: UTM31N
Datum: ETRS 1989
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Lokaal coördinatennet: -
Hoogte/Dieptereferentie volgens LAT
Diepte in meter
AMT/N/88
0 2 4 6 8 10
km
Schaal:1 = 175.000
02-02-2021

Appendix 2: Overview of executed projects

chapter 5 from Lauwaert et al. (2019)

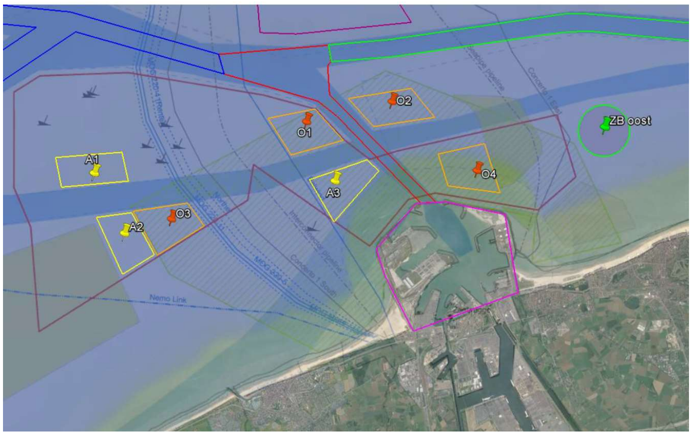
5. Overzicht van de uitgevoerde projecten

5.1 Verder traject stortproef ZBW

Volgend uit onderzoek (2009-2016) naar de efficiëntie van de bagger- en stortstrategie, wordt er verwacht dat een verplaatsing van de stortplaats ZBO naar het westen van Zeebrugge leidt tot een daling in SPM concentratie zoals beschreven in Lauwaert et al. (2016). In 2017 is hierdoor een onderzoek gestart naar de praktische implementatie van een stortlocatie ten westen van Zeebrugge. Tijdens een verkennend onderzoek zijn er eerst mogelijke locaties en exploitatiescenario's voor het storten van baggerspecie gedefinieerd, onderbouwd en afgetoetst aan randvoorwaarden. In 2018 is er vervolgens een milieunota opgesteld die te verwachten milieueffecten van de verschillende exploitatiescenario's beschrijft en dient ter onderbouwing van een nieuwe machtiging of wijziging in machtiging voor het storten van baggerspecie in zee.

a) Verkennende fase locaties en exploitatiescenario's

Tijdens de verkennende fase zijn er op basis van juridische en beleidsmatige randvoorwaarden, omgevingsrandvoorwaarden, technische randvoorwaarden, ecologische randvoorwaarden, economische randvoorwaarden en bijkomende eisen vastgelegd op het startoverleg, zevental nieuwe opties voor een stortlocatie (A1, A2, A3, O1, O2, O3 en O4). In het selectieproces om tot een de exploitatiescenario's te komen zijn enkel A1, A3, O1 en de huidige stortplaats ZBO geselecteerd omdat deze in vergelijking met andere locaties binnen de reservatiezone liggen, beter scoren op hercirculatie, een gunstigere diepgang hebben of de vaarafstand vanuit de haven gunstiger is. Hieruit zijn 5 mogelijke exploitatiescenario's opgesteld. Ieder scenario heeft bestaat telkens op twee locaties zodat er telkens kan uitgeweken worden naar een alternatieve locatie wanneer 1 van beiden locaties niet toegankelijk is, door bijvoorbeeld het broedseizoen. Er is beslist om al deze 5 scenario's mee te nemen in het milieueffecten onderzoek.



Exploitatiescenario	Stortlocatie(s)	Omschrijving
1	O1 + ZBO	O1 wanneer toegankelijk, anders ZBO
2	O1 (noord/zuid)	O1 zuid wanneer toegankelijk, anders O1 noord
3	O1 + A1	O1 wanneer toegankelijk, anders A1
4	A3 + A1	Verdeling A3 / A1 op basis van jaargemiddelde hercirculatie en gemiddelde vaarafstand
5	ZBO + A1	A1 primair (25 %) en ZBO (75 %) ter reductie van jaargemiddelde hercirculatie

b) Milieunota voor beoordeling van de alternatieve baggerstortlocaties

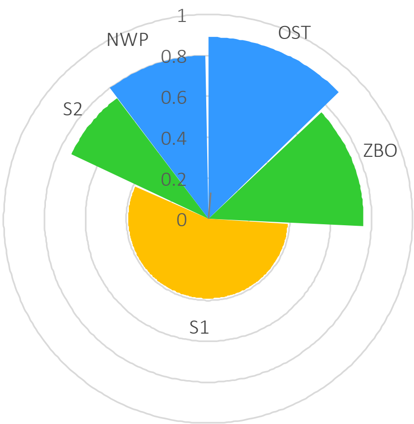
In deze vervolgstap zijn de vijf mogelijke scenario's beoordeeld op milieueffecten. De belangrijkste effecten van het verplaatsen van de baggerstortlocatie hebben betrekking tot de disciplines bodem, water en fauna & flora (marcrobenthos), garnaalvisserij en scheepvaart. De verlaging van de hercirculatie veroorzaakt in verhouding met de huidige exploitatie op ZBO voor scenario 3 en 4 een significant positieve score, scenario 1 is positief, scenario 4 is gering positief en scenario 5 is verwaarloosbaar. Door bedekking van de toplaag treedt biotoopverlies op voor macrobenthos dit treedt vooral op in het Albra alba habitat (A1 in scenario 3, 4 en 5). Baggerstorten heeft een negatieve invloed op de garnaalvisserij inspanning, dit kan echter deels met behulp van milderende maatregelen worden opgevangen, zoals uitwijkingen gedurende het garnaalseizoen. Er is een gedeeltelijke overlap en kruisingen met diverse secundaire scheepvaarroutes en gekende verkeersstromen. Dit is voornamelijk ter hoogte van A3, ter hoogte van deze locatie valt bijgevolg een grotere hinder te verwachten. Voor de meeste disciplines wordt er echter vrijwel geen effect verwacht in vergelijking met het huidige exploitatiescenarios. Op basis van dit effectenonderzoek gecombineerd met enkele praktische expertise is beslist een langdurige test op te starten waar scenario 1 zal worden getest (zie punt 6.4).

5.2 Beleidsinput MSFD

De Belgische MSFD-rapportage omvat in verschillende hoofdstukken informatie komende van het baggerstortsonderzoek. In dit hoofdstuk worden de resultaten weergegeven. Meer details (kader, methodologie) staan in het uitgebreid MSFD artikel 8 rapport (Belgische staat, 2018) (https://odnature.naturalsciences.be/msfd/nl/). Daarnaast zijn er voor een aantal hoofdstukken uit het rapport, gegevens uit de baggermonitoring geïncorporeerd in de algemene analyse (Vb. visziektes, voorkomen belangrijke benthische soorten, niet-inheemse soorten, …).

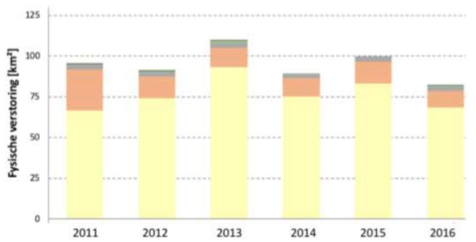
a) Toestand benthische habitats (zachte substraat) (D1, D6)

Activiteit 'Dumpen van gebaggerd materiaal': Voor de status van het benthisch habitat op de vijf stortlocaties, is per locatie een gemiddelde Ecological Quality Ratio (EQR)-waarde (via BEQI; www.beqi.eu) bepaald over de periode 2010-2014. De methodologie voor deze monitoring en analyses is beschreven in Lauwaert et al. (2016). Vervolgens is een gemiddelde EQR berekend over alle locaties heen, rekening houdend met de oppervlakte van de respectievelijke stortplaatsen. De benthische habitat condities zijn zeer goed vergelijkbaar tussen de impact- en controlelocatie voor de stortplaatsen ter hoogte van Oostende (EQR=0.89) en Nieuwpoort (EQR=0.80), goed vergelijkbaar voor de stortplaats S2 (EQR=0.74) en Zeebrugge Oost (EQR=0.76) en zwak vergelijkbaar voor stortplaats S1 (EQR=0.39). Dit wijst op een gedegradeerd habitat op locatie S1 gelegen in het infralitoraal zand. De algemene benthische toestand voor het totale gebied dat beïnvloed wordt binnen een bepaald habitat door het storten van gebaggerd materiaal wordt berekend door de EQR-scores per site uit te middelen in relatie tot de grootte van de stortplaats. Voor de stortplaatsen (S1, S2, Nieuwpoort) gelegen in het infralitoraal zand geeft dit een matige beoordeling (EQR=0.496). Dit is volledig toe te schrijven aan de veranderde toestand van de habitat op site S1, de grootste stortzone (72% van het totale beïnvloedde gebied). De twee stortplaatsen (ZBO, Oostende) in het infralitoraal slib krijgen een zeer goede beoordeling (EQR=0.825). Dit betekent dat 0.64% van het infralitoraal zand een ongunstige benthosstatus heeft door het storten van gebaggerd materiaal.



b) Fysische verstoring en verlies van zeebodem (D6)

Fysische verstoring wordt vooral veroorzaakt door het baggeren en storten van gebaggerd materiaal, de zandwinning, de installatie van windmolenparken en bijhorende kabellegging, en door bodemberoerende visserij. Voor de periode van 2011 tot en met 2016 blijkt de fysische verstoring van de zeebodem nagenoeg constant. Sommige activiteiten zijn permanent, andere kennen een jaar-tot-jaar variatie. Visserij buiten beschouwing gelaten, wordt gemiddeld een oppervlakte van 2.5 à 3% van het BCP per jaar verstoord door menselijke activiteiten. Een evaluatie van fysische verstoring per grootschalig habitattype toont dat baggerwerken en het storten van gebaggerd materiaal vooral samenvalt met de circalittorale slib- en zandhabitats; mariene aggregaatextractie met de circalittorale

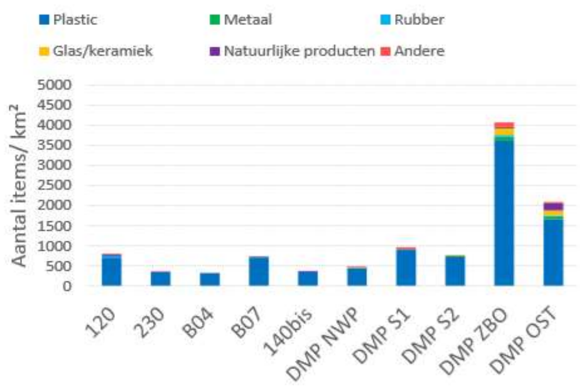


	2011		2012		2013		2014		2015		2016	
	km²	%	km²	%	km²	%	km²	%	km²	%	km²	%
Windmolens	0.19	0.01	0.19	0.01	0.57	0.02	0.00	0.00	0.00	0.00	0.39	0.01
Energiekabels	0.27	0.01	0.60	0.02	0.91	0.03	0.00	0.00	0.00	0.00	0.41	0.01
Munitiestortplaats	3.03	0.09	3.03	0.09	3.03	0.09	3.03	0.09	3.03	0.09	3.03	0.09
Baggeren en lossen	25.21	0.73	13.28	0.38	12.15	0.35	11.08	0.32	13.36	0.39	9.87	0.29
Extractie van zand	66.67	1.93	74.11	2.15	93.08	2.69	75.22	2.18	83.21	2.41	68.39	1.98
Totaal	**95.38**	**2.76**	**91.20**	**2.64**	**109.74**	**3.18**	**89.33**	**2.59**	**99.60**	**2.88**	**82.09**	**2.38**
Visserij	BDNZ		BDNZ		BDNZ		BDNZ		BDNZ		BDNZ	

tot zeewaartse zandige en grofkorrelige habitats. De oorlogsmunitiestortplaats 'Paardenmarkt' valt vooral samen met infralittorale slibhabitats. Het voorkomen van de zeewaarts gelegen gemengde sedimenten en slibhabitats worden fragmentarisch verstoord door mariene aggregaatextractie (merkbaar in 2013, 2014 en 2016).

c) Afval op de zeebodem (D10)

In de figuur wordt het aantal afvalitems/km² weergegeven op basis van de monitoring van de stortplaatsen op het BCP (Lauwaert et al., 2016). De vijf in gebruik zijnde stortplaatsen hebben een oppervlakte variërend tussen 0.9 en 7.1 km² (zie Figuur 2.61). De drie stortplaatsen nabij Zeebrugge worden intensief gebruikt, met in 2015 een gestorte hoeveelheid tussen 3.0 en 5.5 miljoen TDS per stortplaats. Stortplaatsen Oostende en Nieuwpoort werden veel minder intensief gebruikt, met respectievelijk 0.5 en 0.2 miljoen ton droge stof gestort in 2015 (Lauwaert et al., 2016). De locaties 120, 230, 140bis, B04 en B07 zijn referentieslepen.



Op basis van dit baggermonitoringsonderzoek varieerde het aantal afvalitems in de Belgische kustzone in de periode 2013-2016 tussen 330±140 en 4100±6500 items/km². Op de baggerstortplaatsen Zeebrugge Oost en Oostende werden de hoogste aantallen afvalitems waargenomen. Dit is waarschijnlijk niet alleen een effect van het storten van baggerspecie. Beide baggerstortplaatsen liggen namelijk in zogenaamde sedimentatiegebieden in het BCP (Fettweis et al., 2009) en zijn bijgevolg semi-natuurlijke verzamelplaatsen voor marien afval op de zeebodem. In de referentieslepen en de andere drie baggerstortplaatsen lag het aantal afvalitems beduidend lager, gemiddeld 620±670 items/km2. In alle slepen van het baggermonitoringsonderzoek was plastic de overwegende afvalcategorie (92-96%).

Het aantal afvalitems in de baggermonitoringsslepen lijkt aanzienlijk hoger dan in de slepen van het BTS-visserijonderzoek op het BCP. Dit kan verklaard worden omdat de baggerstortplaatsen en -slepen allemaal in de nabije kustzone liggen, maar vooral omdat er gevist wordt met een kleinere maaswijdte, waardoor meer kleine afvaldeeltjes kunnen worden opgevist. Daardoor is het niet mogelijk beide onderzoeken direct met elkaar te vergelijken.

Op basis van het baggermonitoringsonderzoek, dat met een boomkor met fijnere maaswijdte wordt uitgevoerd, kan geconcludeerd worden dat meer dan 90% van het afval in de Belgische kustzone uit plastic items bestaat, met een gemiddelde van 1050±2300 plastic items/km². Op baggerstortplaats Zeebrugge Oost en Oostende worden de hoogste aantallen afvalitems genoteerd, respectievelijk 3 en 7 maal hoger dan in de andere slepen in de kustzone, wat kan gerelateerd worden aan het storten van baggerspecie uit de havens maar ook aan de natuurlijke sedimentatieprocessen in deze zones.

5.3 Optimalisatie baggerwerken kusthavens (MDK)

Voor het onderzoeksproject voor de optimalisatie van de baggerwerken in de kustjachthavens werden twee pistes verder onderzocht, namelijk de optimalisatie van de baggerstortlocatie en de uitvoeringsmethode.

a) Optimalisatie van de stortlocatie met behoud van de stortmethode.

Binnen de opmaak van het MRP 2020-2026 werd gezocht naar een optimalisatie van de bestaande stortlocaties. Dit heeft geresulteerd in de aanduiding van een zoekzone als alternatief voor de baggerstortplaats Nieuwpoort voor de te storten specie afkomstig van de baggerwerken in de haven van Nieuwpoort. De zoekzone voor Zeebrugge West ter vervanging/aanvulling van Zeebrugge Oost, waarvoor het projectonderzoek loopt door aMT, levert ook voor de baggerwerken in de jachthaven van Blankenberge een rendementswinst op. Hierdoor wordt niet verder gezocht naar een eigen alternatieve stortlocatie.

b) Optimalisatie van de stortmethode.

Met verschillende marktpartijen werden de uitvoeringsmodaliteiten afgewogen. Het werken met een vaste stortleiding in plaats van het kleppen met onderlossers lijkt pas haalbaar indien de stortlocatie zich op minder dan 1 km uit de kust bevindt. De impact op het landgebeuren is vrij groot door de aanwezigheid van bovengrondse leidingen wat vanuit ruimtelijk oogpunt niet wenselijk is. De rendementswinst die zou kunnen gehaald worden door de uitvoering van de baggerwerken wordt beperkt door het tempo waarmee de haven kan vrijgemaakt worden. Deze beperkende factor zorgt ervoor dat de baggertuigen suboptimaal zouden ingezet worden. De beoogde rendementswinst wordt hierdoor niet gehaald.

c) Conclusie

Gelet op de gedetecteerde randvoorwaarden en de beperkte rendementswinst wordt de alternatieve stortmethode voorlopig niet verder onderzocht. De alternatieve stortlocaties worden wel verder onderzocht.