Academic year 2012 – 2013

IMPACT OF BEACH NOURISHMENT ON COASTAL ECOSYSTEMS WITH RECOMMENDATIONS FOR COASTAL POLICY IN BELGIUM

IMPACT VAN ZANDSUPPLETIE OP KUSTECOSYSTEMEN MET AANBEVELINGEN VOOR HET BELGISCHE KUSTBELEID

Sarah Vanden Eede

Promotors:
Prof. Dr. Magda Vincx
Prof. Dr. Steven Degraer

IMPACT OF BEACH NOURISHMENT ON COASTAL ECOSYSTEMS WITH RECOMMENDATIONS FOR COASTAL POLICY IN BELGIUM

ISBN 9789090278438
EAN 9789090278438
Marine Biology Research Group
Campus Sterre – S8
Krijgslaan 281
B-9000 Gent
Belgium

Academic Year 2012 – 2013

Publically defended on September 20th 2013

Co-authored one or more chapters:
Joke Van Tomme, Dries Bonte, Lia Laporta, Klaas Deneudt, Sofie Derous, Jeroen Speybroeck, Charlotte De Busschere, Martijn Vandegehuchte, Eric Stienen, Koen Sabbe, Steven Degraer, Magda Vincx

For citation to published work reprinted in this thesis, please refer to the original publications (as mentioned in the beginning of each chapter).

Vanden Eede S (2013) **Impact of beach nourishment on coastal ecosystems with recommendations for coastal policy in Belgium**. Ghent University (UGent), pp. 301

UNIVERSITEIT
GENT

FACULTEIT WETENSCHAPPEN

Mariene Biologie
Universiteit Gent

IMPACT OF BEACH NOURISHMENT ON COASTAL ECOSYSTEMS WITH RECOMMENDATIONS FOR COASTAL POLICY IN BELGIUM

IMPACT VAN ZANDSUPPLETIE OP KUSTECOSYSTEMEN MET AANBEVELINGEN VOOR HET BELGISCHE KUSTBELEID

Sarah Vanden Eede

Promotors	Prof. Dr. Magda Vincx
	Prof. Dr. Steven Degraer

Academic year 2012 – 2013

Thesis submitted in partial fulfilment of the requirements for the degree of Doctor in Science, Marine Sciences

Members of the reading committee

Dr. Angel Borja (AZTI, Spain)
Dr. Jeroen Speybroeck (INBO, Brussels, Belgium)
Dr. Jan Vanaverbeke (Ghent University, Belgium)

Members of the examination committee

Dr. Angel Borja (AZTI, Spain)
Dr. Jeroen Speybroeck (INBO, Brussels, Belgium)
Dr. Jan Vanaverbeke (Ghent University, Belgium)
Prof. Dr. Dries Bonte, Chairman (Ghent University, Belgium)
Prof. Dr. Magda Vincx, Promotor (Ghent University, Belgium)
Prof. Dr. Steven Degraer, Promotor (Ghent University, Belgium)
Dr. Gert Van Hoey (ILVO, Ostend, Belgium)
Prof. Dr. Ann Vanreusel (Ghent University, Belgium)

Table of contents

Dankwoord

Iedereen die mij kent, weet dat ik er nooit in zou slagen om een saai, dof en kort dankwoord te schrijven. Het volgende epistel komt dan ook niet als een verrassing, maar als een bevestiging dat jullie mij goed kennen ☺

"Either write something worth reading or do something worth writing." - Benjamin Franklin

Eerst en vooral dien ik mijn promotor Magda Vincx te bedanken voor de kans die ze mij geboden heeft om te doctoreren en voor het jarenlange vertrouwen en de vele adviezen.

Ook mijn co-promotor Steven Degraer verdient een welgemeende dank u wel. Dankzij jou is het allemaal begonnen voor mij. Ik heb weinig met je kunnen samenwerken gezien je net de Marbiol verliet toen ik er binnen wandelde, maar voor mij blijf je de pionier der Belgische strandbiologen bij uitstek.

Daarnaast wil ik graag de wetenschappers bedanken die de tijd en moeite genomen hebben om mijn doctoraat te lezen en te becommentariëren. Bedankt om er samen met mij een beter werk van te maken.

In het bijzonder en zeer uitdrukkelijk wil ik Dries Bonte bedanken. Oorspronkelijk de man van het model, maar sindsdien de drijvende kracht achter mijn doctoraat. Je had totaal geen verplichtingen ten opzichte van mij en toch nam je ridderlijk de taak op jou om mij te begeleiden in de hoofdstukken waarin ik het meeste stuurloos was. Ik heb je tot mijn officieuze copromotor gebombardeerd en ik ga je eeuwig dankbaar zijn voor je onbaatzuchtige hulp op de meest onmogelijke momenten. Je hebt van mij zonder meer een betere wetenschapper gemaakt en mijn doctoraat zou nooit geworden zijn wat het nu is zonder jou.

Joke, mijn partner-in-crime als het op strandonderzoek aankomt. Samen hebben we beslag gelegd op Dries tijdens de vele model besprekende en ontrafelende (wat doet dat model nu eigenlijk?) sessies. Ik heb de zwangerschap van je beide zoontjes van dichtbij meegemaakt. Sindsdien is mijn bewondering voor jou enkel maar gegroeid. Je combineerde je onderzoek moeiteloos met je kindjes en zelfs toen je hoogzwanger was, ging je zonder pardon mee op staalname. Ik heb intens genoten van onze samenwerking en je bent voor mij een inspiratie op meerdere vlakken.

Klaas, sinds je mij hebt leren kennen tijdens mijn thesis onderzoek ben je nooit meer van mij afgeraakt. Ik moet er regelmatig mee lachen hoe ik een hele reeks leergierige jonge vrouwen op jou heb afgestuurd met problemen over biologische waarderingskaarten. Van jou heb ik alles geleerd wat ik weet over data, databasen, gis en BVM. Ik heb de evolutie meegemaakt van de talloze programma's tot het machtige R script (1 klik… en alles gaat vanzelf???). Ik heb er nooit een geheim van gemaakt hoe intens interessant ik alles vond en zonder jou was er nooit een hoofdstuk over dit onderwerp in mijn doctoraat geraakt.

Marta, nunca olvidará el momento en que apareció en mi oficina. La conexión entre tú y yo fue sin lugar a dudas y por el tiempo que te fuiste, eran amigas. Me entiende como pocos lo hacen. ¡Gracias por todo!

The best thing about a PhD is guiding young scientists. Lia, you were sort of disappointed in the beginning since you were supposed to start under the wings of Marijn Rabaut, but by the end, I had completely won you over. I guess marine biological valuation makes young women thrive under pressure and excel when no one deems it possible (or would Klaas have something to do with it? ☺). Elisabeth, Thomas, Mick, Maaike, Kristof, Yves, Gaetan, Nina en Nathalie... Bedankt om deel uit te maken van mijn doctoraat. Ik heb even veel van jullie geleerd als jullie van mij. Ik wens jullie allemaal een interessante wetenschappelijke carriere en een magisch leven toe. Ik heb ook intens genoten van de vele begeleidende studie activiteiten, van het vermoeiende Wimereux tot de biostatistiek lessen en uiteraard ook van de EMBC Summer School in Galway met Nele en Tim. Hilarische momenten en die video... ja... die verwacht ik wel ergens op mijn doctoraatsfeestje ☺

De steun die ik binnen de MarBiol heb gekregen valt niet te onderschatten. Ik wil Guy bedanken als excellente bureaucompagnon, Karen als wijs en lichtend voorbeeld (wetenschappers zouden meer moeten zijn zoals jij ☺), Marijn omdat je bewijst dat wetenschap meer is dan publiceren en dat wetenschappelijke boodschappen een wereldlijk publiek horen te hebben, Tania (we will always have Zion National Park... de meest absurde ontmoeting in mijn leven ☺) en de vele buitenlanders (Tatiana, Giovanni, Maickel, Kwang...) omdat jullie keer op keer aantonen dat wetenschap niet stopt aan onze grenzen, Delphine omdat we zo gelijkend zijn dat het onmogelijk was voor mij om niet je vriendin te worden, Nele omdat je een schitterend mens bent en een zeer goede vriendin voor talloze mensen (merci voor de diepe gesprekken in Galway ☺), Tim omdat ik echt genoten heb van je warme gezin en je vele wijze raad, Jan omdat we samen begonnen zijn bij de windmolens en samen gefinished, zij het op wat anders ☺... ik weet nu al dat je een schitterende papa zal zijn, Liesbet omwille van de vele hulp in kleine en grote dingen. De Marbiol bestaat echter uit nog veel meer mensen die ook allemaal een pluim verdienen. Samen vormen we een team ☺

Zelf stond ik al snel gekend als de madam van de strand staalnames. Gezien ik telkens op veel vrijwilligers moest kunnen rekenen, bedacht ik een slim plan: mensen aanmoedigen door cake als traktatie te voorzien. Dat leidde uiteindelijk tot vaste vrijwilligers omdat ze fan waren van mijn cake ☺ Ik kreeg zelfs cake aanvragen voor een staalname. Ik wil drie mensen in het bijzonder bedanken om altijd paraat te staan en van mijn staalnames een superplezant gebeuren te maken.

> Bart, de vaste partner op mijn subtidale staalnames. Je hebt mijn ongelukkige havendoop meegemaakt en me supergoed opgevangen achteraf (zelfs naar het verre Kortrijk gebracht omdat ik anders nog met de trein naar huis moest). Bedankt voor de vele vele vele Van Veens ☺
>
> Niels, ja... waar te beginnen ☺ Je was steevast de entertainer van mijn strandstaalnames. Je kent de procedures nu minstens zo goed als ik. Het is altijd plezant met jou, van het luidkeels Disney liedjes zingen tot de schitterende Despicable Me imitaties (Vector! That's me, because I commit crimes with both direction and magnitude! ... Yeah! You done been shrunk!... Look at you, a little tiny toilet, for little tiny people to... Ah! Curse you, tiny toilet!...) ☺ Merci voor zo'n toffe vent te zijn. En ohja... een dikke knuffel kan uiteraard niet ontbreken ☺
>
> Annelien, de sterke madam die voor niemand hoeft onder te doen. We zijn even geschift (Als ik nu 's morgens mijn banana zit te eten kijk ik bang rond of er geen minions in de buurt zijn ☺),

maar je was toch wel de enige die achter een vermoeiende staalname dag in bikini de zee in rende, weer of geen weer. Zonder meer: respect ☺ Bovendien vormde je samen met Niels de superfans van mijn cake, getuige deze mails:

” genoeg helpende handen... maar als je mij ook mee neemt is er iemand om de cake te dragen en kan er op strand cake gegeten worden !!”

“Ikke! Ikke! alle dagen zijn mogelijk. please choose me :)”

“als je niemand kan vinden is dat goed voor mij want dan is de cake binnen bereik :)”

Ook buiten de Marbiol waren er heel wat mensen die het praktische werk van mijn doctoraat ondersteund hebben. Gilbert en Koen Allewereldt wil ik bedanken voor de vele toffe dagen aan boord van de Last Freedom. Ook de mensen van G-TEC die telkens weer voor de ondersteuning zorgden aan boord: Geert, Annelies, Elisabeth en Samuel. Dank u wel aan de mannen van het Kwartier van Lombardsijde om mij altijd toe te staan om via het militair domein van Lombardsijde het strand te betreden. Om te eindigen zijn er nog de mensen van AWK die het project financierden waarop ik uiteindelijk kon doctoreren. Bedankt voor de fijne samenwerking Miguel, Tina, Elias en Miran!

Tot zover het professionele luik. “It’s a dangerous path I bounce, but I bounce it alone.” – Tigger. Een doctoraat is zeker een eenzaam pad. Uiteindelijk doe en schrijf je het alleen. Andere mensen helpen je en sturen je, motiveren je en stuwen je, maar ze blijven buitenstaanders. Toch kan je het niet zonder hen. Ik kan me gelukkig prijzen met de vele exceptionele warme mensen die me omringen.

Ik wist al op jonge leeftijd wat ik wou worden dus voor mij lagen studiekeuzes al vrij vroeg vast. Op mijn biologisch pad doorheen het universitaire Vlaamse landschap (KULAK in Kortrijk, KUL in Leuven en UGENT in, jawel, Gent ☺) kwam ik heel wat mensen tegen die uiteindelijk gezellig meetoefden.

Nathalie, toeval bracht ons samen toen ik de aula binnenwandelde en geheel willekeurig naast jou plaatsnam. Je bleek farmacie te studeren, maar toch werden we in no-time vriendinnen. Je hart huist nu in Limburg, maar we hebben genoeg gemeen, waaronder onze liefde voor Latijns-Amerikaanse dansen, om vriendinnen voor het leven te blijven.

Sinds we samen het dreamteam vormden tijdens onze vakantiejob aan het VLIZ, vormen Heidi, Joke en ik een hecht trio, sterke madammen met een liefde voor de zee. Heidi, van al mijn vriendinnen ben jij degene die aan mij gegroeid is geraakt. Van single huismus tot getrouwde wereldreiziger... Ik heb het allemaal van heel dichtbij gevolgd en jij begrijpt mijn reishonger als geen ander. Bedankt om samen met Noeli voor mij te supporteren langs de andere kant van de wereld en weet dat ik in gedachten mee reis. Joke en Tom, want jullie zijn een team voor mij ☺. Ook gebeten door de reiskriebel, maar ik voel een andere kriebel ontwikkelen die jullie een mooie toekomst belooft. Joke, bedankt voor het luisterend oor. Tom, ik hoop dat Boulet en Frikandel er komen, want ik mis Germaine ook telkens als ik op bezoek kom ;)

Annelies ontpopte zich al snel tot mijn Marelac buddy. Samen met Gosia hebben we dat jaar echt vanalles gedaan en beleefd. Van Euro Disney tot uiteindelijk onze avonturen in Ecuador… Het is een hechte vriendschap geworden die ik koester.

Ook Arne leerde ik kennen in mijn Marelac studiejaar. Jouw nuchterheid en eerlijkheid bewonder ik enorm en ik hou van onze babbels. En ohja… in gedachten weet ik al welke geitjes er op uw grond gaan grazen ☺

Toen ik nog actief was in Kansen voor Kinderen leerde ik Sofie kennen. Zij zat in de kookploeg, ik was leidster van de middelste jongens. We zijn mekander nooit meer uit het oog verloren. Samen waren we "wanhopig" single, begonnen we aan een doctoraat (jij in de engelse literatuur en ik in de mariene biologie) en ik was erbij toen je Stani leerde kennen. Mijn hart schoot vol toen ik vernam dat je zwanger was en ik maakte er een erezaak van om Lewis zo snel mogelijk te ontmoeten. Bedankt voor je vele wijze raad en je diepe vriendschap zofia.

Mijn twee hobbies brachten mij rust tijdens de hectische periodes die het doctoraatsleven rijk is en dat niet in het minst dankzij een aantal exceptionele mensen.

Ik duik al sinds mijn 14^e^ en ruim 10 jaar geleden leerde ik Adje en John kennen. Duikpunt vormt sindsdien voor mij een thuishaven. Een speciale dank u wel gaat dan ook naar hen, Heidi, Nadieh (mijn persoonlijke cheerleader ☺) en Wim om met mij bubbels te blazen in de oneindige stilte en rust die ik telkens onder water ervaar.

Mijn liefde voor salsa bloeit nu al 6 jaar. Sinds vorig jaar behoor ik tot een vrolijke bende die de dansvloer regelmatig in vuur en vlam zet onder de deskundige leiding van Domynyck en Isabel. Salsazucar betekent voor mij plezier van de bovenste plank. Ik laad mijn batterijen altijd op met jullie allemaal samen dus dank u voor al die prachtige dansmomenten.

Kathy oftwel tante Kaat ☺ Bedankt voor je wijze raad en je onuitputbare levensenergie. Ik hoop dat ik mag ouder worden naar jouw evenbeeld… sprankelend en enthousiast.

Annelies, mijn Afro-latino buddy. Bedankt voor de vele intense gesprekken die we hebben gedeeld. Ga voor de droom meid! Het is het zo waard!

Matti, mijn salsabroer, correctie dansbroer ☺. Je weet wie ik ben tot in het kleinste detail, want we lijken op elkaar als twee druppels water. Maar… daarnaast ben je ook mijn klankbord en spiegel. Bedankt om er voor mij te zijn Matti, in goede en in kwade dagen.

Stijn… Jij bent in zekere zin mijn nuchtere evenbeeld. Ik heb jou zien evolueren van kwetsbare lieve jongen naar zelfzekere toffe vent aan de zijde van een nog zelfzekerder exemplaar, de Sandro ☺ Ook al woon je nu in Poperinge, ik koester jouw vriendschap iedere dag en jouw getuige zijn, was een grote eer voor mij. Bedankt voor wie je bent en bedankt om mij te laten zijn wie ik ben. Via jou leerde ik ook Steve en Pascal kennen. Ik zie het rode autootje nog steeds in mijn achteruitkijkspiegel…

"Take pride in how far you've come and have faith in how far you can go"

Steven, je was lang de man in mijn leven en hoewel de publieke opinie vindt dat je na een relatie geen vriendschap kan hebben, proberen we het tegendeel te bewijzen. Ik kan niet anders dan je bedanken. Je was erbij van in het begin en je bent de reden dat ik heb doorgezet op talloze momenten dat ik er de brui aan had willen geven. Zelfs tijdens die laatste zware loodjes heb je me vaak opgepept en voortgestuwd. Bedankt vanuit het diepste van mijn ziel… Bedankt.

Tom, het stond ergens wel in de sterren geschreven, denk ik dan. 13 jaar geleden zaten we samen in het 5^{e} middelbaar wiskunde-wetenschappen 8 uur, maar pas in het derde jaar unief leerden we elkaar echt kennen. Sindsdien zijn we maatjes en na alles wat we samen hebben meegemaakt, was een relatie iets wat ik niet meer zag gebeuren en toch… Ik kan niet in woorden omschrijven hoe het voelt om mijn leven met jou te delen. Het voelt vertrouwd en juist en mijn hart voelt zich thuis. Ik zie je super graag. Bedankt om mij minstens zo graag te zien als ik jou graag zie ☺

Having a place to go - is a home; Having someone to love - is a family; Having both - is a blessing - Donna Hedges

Mijn familie is niet groot, maar wel onmetelijk belangrijk voor mij. Papa, Julie en Peter, Sander, Christel, Thomas, meme en pepe, tante Mich en nonkel Mark… Bedankt om er voor mij te zijn, om te luisteren en te begrijpen, om te delen en te aanvaarden. En Basjemansje… bedankt om als het vrolijkste zonnestraaltje door mijn leven te huppelen ☺

Summary

Sandy shores or beaches line 70 % of the world's oceans, including the entire Belgian coastline. They have a multitude of ecosystem functions, constitute an important habitat for a variety of fauna and flora and hold important economic, social and cultural value as prime recreational assets. Predictions on sea level rise, intensified storms, accelerated erosion and flood risk for the North Sea have led to the drafting of the Belgian Integrated Coastal Safety Plan. In order to protect the Belgian coastline against erosion and coastal flooding on a short and long term basis (up to 2050), the Belgian sandy beaches face a multitude of beach nourishment activities over the next years. This soft coastal defence measure safeguards the natural dynamics of the coast and has little impact on the beach ecology and tourism compared to other options. However, together with the multitude of human beach functions such as tourism and economic development, beach nourishment potentially threatens the natural balance of the beach and coastal ecosystem. As management of the coastal zone is clearly a multi-faceted and complex endeavour, where the interests of several stakeholders need to be combined, coastal management desperately needs ecological dimensions. Hence, solid and meaningful biological and ecological information is needed. Clear and user-friendly management tools are essential to guide integrative and ecosystem-based strategies to sustainably manage ongoing space-use activities at the Belgian beaches and coast.

In **chapter 1** an overview of the sandy beach ecosystem, illustrated by the focus habitat of this PhD thesis, namely the Belgian coastal zone, is given. Beach ecosystem components, food web and threats are thoroughly discussed with a prime focus on coastal defence structures and their impact on the beach ecosystem. The current status of the Belgian coastal governance and policy is documented to provide for a better understanding of beach and coastal spatial planning in Belgium.

A good knowledge of the Belgian beach ecosystem in both the intertidal and shallow subtidal zone provides us with a baseline condition. This is the condition of the natural resources and ecosystem services that would have existed if no impacts had occurred, estimated on the basis of historical data, reference data or control data. Data from 1997 to 2011 were analysed in **chapter 2**, encompassing 16 intertidal and 10 shallow subtidal coastal zones, sampled over 8 years in 3 different seasons. The partitioning of the macrobenthic community structure within the Belgian beach ecosystem showed a large within beach variability, linked to elevation on the beach and median grain size of the sediment, in both the intertidal and shallow subtidal zone. Several spatial and temporal trends in abiotic factors and in macrobenthic species richness, abundance and biomass were measured. Between the detected minimum and maximum values the natural variation on Belgian beaches runs its course. Furthermore, the observed niche and interpolated occurrence of the dominant macrobenthic species of the Belgian beaches were defined, characterized by elevation on the beach and median grain size of the sediment. All these findings assess the natural variability on the Belgian beaches and increase the strength,

efficiency and accuracy of monitoring strategies to detect possible impact effects on the Belgian beaches.

Since the Belgian sandy beaches face a multitude of beach nourishment activities over the next years, optimizing the technical aspects of future nourishment projects is as such indispensable to maintain an ecologically healthy beach ecosystem. The 2009 nourishment on the Belgian beach of Lombardsijde has been scientifically evaluated through a Before After Control Impact (BACI) designed research in **chapter 3**. Following the recommendations of Speybroeck (2006) some ecological adjustments were taken into account, resulting in a phased nourishment project with nourished sand closely matching the original sediment and only moderate beach profile changes. A wider, higher and flatter intertidal beach with coarser sediment was created and no return to the pre-nourishment conditions was visible three years after nourishment. The sediment grain size distribution had changed as well, showing slow recovery in the three post-nourishment years. The analysis of the macrobenthos community structure showed that at least in some cases nourishment under ecological optimal conditions can show no significant effects on both the intertidal and shallow subtidal beach ecosystem 6 months after the nourishment. Within this time frame, the macrobenthos community had recovered from the impact of the ecological nourishment. Ecological nourishment thus proves to be the least ecologically damaging way of combating erosion, compared to all other coastal engineering activities.

By means of experiments and models, the responses of the beach ecosystem to varying environmental conditions and different beach nourishment aspects can be predicted, leading to ecologically adjusted nourishment projects. The sediment preferences of the dominant Belgian intertidal beach macrofauna were experimentally tested both in single-species and combined-species conditions in **chapter 4**. Results of the experiments indicated that *Bathyporeia pilosa* and *Eurydice pulchra* prefer the finest sediment (< 250 µm), while *Bathyporeia sarsi* (250 – 355 µm) had a broader preference and also occurred in medium-coarse sediments. Interspecific competition between the sympatrically occurring *Bathyporeia* amphipods was found to change the sediment selection of the amphipod *Bathyporeia pilosa* towards the coarser sediments where *Bathyporeai sarsi* occurred in lower frequencies. The polychaete *Scolelepis squamata* had the broadest preference (355 – 500 µm) and even showed a high occurrence in coarse sediments that are not naturally occurring on the Belgian sandy beaches. These preferences imply that beach nourishment with coarse sediment will have a major effect on *Bathyporeia pilosa* while effects of coarse sediments on *Scolelepis squamata* will be minor.

A combined envelope-mechanistic model was developed in **chapter 5** to predict short-term changes in beach macrobenthos species richness and abundance in response to changes in beach profile and grain size following beach nourishment. This model also elucidates how these changes in community composition potentially feedback on the abundance of dominant species of higher trophic levels (birds, fish and shrimp), using both the available knowledge and knowledge obtained in this PhD thesis. As different scenarios can be tested with this model, various technical aspects of beach nourishment can be compared and optimized. According to the model, the sediment grain size is the most important factor determining beach-level diversity and production, with strong deterioration of the beach ecosystem after nourishment with too coarse sediment (e.g. >> 300 µm). Therefore the gradient in sediment grain sizes that is advised for nourishment of fine-grained beaches is defined as 200 – 300 µm with the critical median sediment grain size set at 300 µm. Although the effect of nourishment slope and profile was less strong compared to the sediment, it did also affect species zonation patterns. However, patterns for higher trophic levels do not follow the decreasing patterns in macrobenthos abundance and biomass. It is advised to evaluate the beach ecosystem health by a combination of different variables (biodiversity, macrobenthos biomass) since focusing on one variable can be deceptive as opportunistic species can become very abundant on a beach impacted by nourishment.

Protected beaches could help maintain an ecologically healthy beach ecosystem during the coming years filled with coastal defence activities. In order to protect the coastal environment, one has to know what to protect. In essence we need to know which beaches are more biologically valuable than others and hence need a protection status asap. All available biological and ecological information of the shallow Belgian coastal zone was compiled in **chapter 6** for calculating an intrinsic biological value for several Belgian coastal subzones. These subzones are defined as intertidal or shallow subtidal 250 m wide zones of the Belgian coastal zone for benthic components while 3 km wide parts were chosen for highly mobile species such as birds. Spatial coverage and overall data availability were satisfactory and allowed for significant trends and patterns to be observed. Although the Belgian coast is entirely composed by sandy beaches, there is indeed biological diversity among distinct subzones. A strong mosaic pattern of biological value along the coastline and a clear lack of (benthic) data at the eastern part of the Belgian coast was detected. Around 70 % of the shallow subtidal part of the subzones scored rather *high* biological values, compared with the intertidal part and *high/very high* biological values were consistently found in intertidal zones located immediately to the east of the harbours Nieuwpoort, Oostende and Zeebrugge. The designation of marine reserves adjacent to protected beaches and reserves adjacent to the Belgian harbours is of the uttermost importance to achieve a successful and ecologically justified implementation of beach reserves. A detailed analysis of protected areas and areas under coastal flood risk indicated that biological valuation maps are a promising tool in pinpointing the impact prone beach subzones. Biological valuation maps provided a strong visual support to the proposal for the extension of some already existing nature reserves and to the need for more data to allow for significant conclusions regarding the biological value of nature reserves. These maps can be used as reliable and meaningful baseline maps for coastal spatial planning, policy and management approaches as they allow for the integration of 'natural/ecological values' at an early stage of coastal policy implementation.

In the final **chapter 7**, the conservation goals for the Belgian coastal zone are given from a biologist's perspective and beach research results are translated towards beach nourishment recommendations and policy guidelines for an ecosystem-based, integrated sandy beach and coastal management. In conclusion, beach fact sheets are provided summarizing all information gathered during this PhD thesis on the 17 intertidal and 11 shallow subtidal studied coastal locations. In a clear and easy to work with format, each beach record gives an overview of the beach's location, legal circumstances (Provincial

Spatial Implementation Plan and nature conservation status), possible coastal defence activities, biological value and current scientific knowledge. Furthermore, criteria for a good ecological beach, a plea for a multi-disciplinary, integrated beach and coastal spatial plan and some future beach research suggestions have been made.

Samenvatting

Zandstranden bakenen 70 % van alle continentale grenzen af, inclusief de Belgische kustlijn. Ze hebben vele ecosysteem functies en herbergen een specifieke fauna en flora. Dankzij hun recreatieve waarde zijn ze bovendien van groot economisch, sociaal en cultureel belang. Langetermijnsvoorspellingen voor de Noordzee maken gewag van een stijging van het zeeniveau, het voorkomen van intensere stormen, versnelde erosie en grotere overstromingsrisico's. In het laaggelegen België werd hierop geanticipeerd met het Geïntegreerde Kustveiligheidsplan. Dit plan voorziet in grote zandsuppleties gespreid over de komende jaren om de Belgische kustlijn te beschermen tegen erosie en overstromingen op de korte en lange termijn (tot 2050). Zandsuppletie is een zachte kustverdedigingstechniek die de natuurlijke kustdynamiek vrijwaart en slechts een kleine impact heeft op de strandecologie en het strandtoerisme, in vergelijking met andere kustverdedigingstechnieken. Gecombineerd met de vele menselijke strandactiviteiten en de economische groei van de Belgische kust, kan zandsuppletie echter wel de natuurlijke balans van het strand-en kustecosysteem verstoren. Aangezien het beheer van de kustzone duidelijk een veelzijdige en complexe onderneming is waarbij de noden van de verschillende belanghebbenden gecombineerd moeten worden, heeft kustbeheer dringend nood aan ecologische dimensies. Relevante en zinvolle biologische en ecologische informatie is essentieel om duidelijke en gebruiksvriendelijke management tools te creëren. Bij het uitbouwen van geïntegreerde management strategieën voor de Belgische kust zullen deze tools er dan voor zorgen dat er steeds rekening gehouden wordt met het Belgische strand-en kustecosysteem.

In **hoofdstuk 1** wordt een overzicht gegeven van het zandstrandecosysteem, geïllustreerd met het focus habitat van deze doctoraatsthesis, namelijk de Belgische kust. De strandecosysteem componenten, het voedselweb en de bedreigingen worden bediscussieerd. Er wordt een duidelijke nadruk gelegd op de kustverdediging en de mogelijke impact van deze menselijke activiteit op het strandecosysteem. Het huidige Belgische kustbestuur en de politieke cultuur hieromtrent worden eveneens gedocumenteerd om de ruimtelijke strand-en kustplanning in België beter te duiden.

Een goede kennis van het Belgische strandecosysteem in de intertidale en ondiepe subtidale zone zorgt voor een goede referentie situatie. Op basis van historische, referentie en/of controle data wordt een beeld geschetst van de oorspronkelijke natuurlijke bronnen en ecosysteem diensten voor er ooit een menselijke impact had plaats gevonden. Data van 1997 tot 2011 werd geanalyseerd in **hoofdstuk 2**. De data omvatten 16 intertidale en 10 ondiepe subtidale kustzones die werden bestudeerd over 8 jaar tijd in 3 verschillende seizoenen. De macrobenthische gemeenschapsstructuur van het Belgische strandecosysteem in de intertidale en ondiepe subtidale zone werd voornamelijk bepaald door een grote variabiliteit binnen de stranden, gekoppeld aan de hoogteligging op het strand en de mediane korrelgrootte van het sediment. Er werden verschillende ruimtelijke en temporele trends aangetroffen in de abiotische data en in de macrobenthische soortenrijkdom, abundantie en biomassa. De natuurlijke variatie op de Belgische stranden speelt zich af tussen de opgemeten minima en maxima waarden. De

geobserveerde niches van de dominante Belgische macrobenthos soorten en hun geïnterpoleerde verspreidingsgebieden werden bepaald op basis van de hoogteligging op het strand en de mediane korrelgrootte van het sediment. Al deze onderzoeksresultaten proberen de natuurlijke variatie in de Belgische kustzone in te schatten. Op basis van deze kennis zal de sterkte, de efficiëntie en de nauwkeurigheid toenemen waarmee monitoring strategieën worden opgesteld en uitgevoerd wat op zijn beurt de detectiekans van mogelijke impact effecten in de kustzone verhoogt.

Gezien de vele geplande zandsuppleties op de Belgische zandstranden is het essentieel om de technische aspecten van deze suppleties te optimaliseren, zeker als men een ecologisch strandecosysteem voor ogen houdt. In 2008 werd op het Belgische strand Lombardsijde een zandsuppletie uitgevoerd. Deze impact werd bestudeerd volgens een Before After Control Impact (BACI) onderzoeksopzet in **hoofdstuk 3**. De aanbevelingen geformuleerd door Speybroeck (2006) werden in acht genomen door middel van enkele ecologische aanpassingen, zijnde een gefaseerde suppletie met sediment gelijkend op het oorspronkelijke sediment en enkel gematige aanpassingen aan het strandprofiel. Een breder, hoger en vlakker intertidaal strand met grover sediment werd gecreëerd en er werd geen terugkeer naar de oorspronkelijke omstandigheden vastgesteld 3 jaar na de suppletie. De sediment sortering veranderde wel en toonde slechts een traag herstel in de 3 jaar volgend op de suppletie. De analyse van de macrobenthos gemeenschapsstructuur in de intertidale en ondiepe subtidale zone vertoonde geen significante effecten die gelinkt konden worden aan de suppletie 6 maanden na de suppletie. Binnen deze periode is de macrobenthos gemeenschap schijnbaar hersteld van de impact die de ecologische suppletie veroorzaakte. Een ecologische zandsuppletie is dus de ecologisch minst verstorende kustverdedigingstechniek.

Experimenten en modellen kunnen de respons van het strandecosysteem op verschillende milieucondities voorspellen en op die manier zandsuppletieprojecten op een ecologische manier bijsturen. De sedimentpreferentie van de dominante Belgische intertidale strandfauna werd experimenteel nagegaan in allotope en syntope omstandigheden in **hoofdstuk 4**. De experimentele resultaten toonden aan dat *Bathyporeia pilosa* en *Eurydice pulchra* het fijnste sediment verkiezen (< 250 µm), terwijl *Bathyporeia sarsi* (250 – 355 µm) een bredere preferentie had en dus ook in grovere sedimenten voorkwam. Er werd ook interspecifieke competitie aangetroffen tussen de twee samen voorkomende *Bathyporeia* amfipoden waardoor de sedimentpreferentie van *Bathyporeia pilosa* kan veranderen naar grovere sedimenten waarin *Bathyporeai sarsi* enkel voorkomt in lagere densiteiten. De polychaet *Scolelepis squamata* had de breedste voorkeur (355 – 500 µm) en kwam zelfs in aanzienlijke abundanties voor in grove sedimenten die van nature niet voorkomen op de Belgische zandstranden. Al deze preferenties impliceren dat zandsuppleties met grof sediment een groot effect zullen hebben op *Bathyporeia pilosa* en slechts een klein tot zelfs positief effect zullen hebben op *Scolelepis squamata*.

Er werd ook een mechanistisch-envelope model ontwikkeld in **hoofdstuk 5**. Dit model voorspelt de korte termijnsveranderingen in de strandmacrobenthos soortenrijkdom en abundanties veroorzaakt door veranderingen in het strandprofiel en in de korrelgrootte. Op basis van de algemene kennis en de kennis verworven in deze doctoraatsthesis probeert het model ook aan te tonen hoe deze veranderingen in de gemeenschapsstructuur van het macrobenthos terugkoppelen naar de abundantie van de dominante soorten van hogere trofische niveaus (vogels, vissen en garnalen). Verschillende technische aspecten van zandsuppleties kunnen vergeleken en geoptimaliseerd worden door verschillende scenario's te testen. Alle modelsimulaties toonden aan dat het gebruikte suppletiesediment de dominante factor is in het bepalen van de effecten op het ecosysteem, met een degradatie van het volledige strandecosysteem na een zandsuppletie met te grof zand (dit wil zeggen met een mediane korrelgrootte >> 300 µm). De aanbevolen gradient voor mediane korrelgrootte van het suppletiesediment werd daarom gelegd op 200 – 300 µm voor fijnkorrelige stranden. Algemeen wordt aangeraden om zand te gebruiken dat qua karakteristieken overeenkomt met de natuurlijke condities op het desbetreffende strand. Ook al was het impact effect van een wijziging in strandprofiel kleiner, het veroorzaakte ook een verandering in de zonatiepatronen van het macrobenthos. Patronen van de hogere trofische niveaus volgen echter de dalende patronen in macrobenthos densiteit en biomassa niet. Het is aangeraden om de gezondheid van het strandecosysteem te evalueren aan de hand van een combinatie van verschillende variabelen (biodiversiteit, biomassa van het macrobenthos), aangezien de focus op één variabele misleidend kan zijn. Opportunistische soorten kunnen immers erg abundant worden op een pas gesuppleerd strand.

Beschermde stranden kunnen helpen bij het handhaven van een ecologisch gezond strandecosysteem tijdens de komende rijkelijk met kustverdedigingsactiviteiten gevulde jaren. Als we het kustecosysteem willen beschermen, moeten we weten wat we willen beschermen. In wezen willen we dus weten welke stranden een hogere biologische waarde hebben aangezien deze stranden zo snel mogelijk een hogere beschermingsstatus verdienen. Alle beschikbare biologische en ecologische informatie omtrent de ondiepe Belgische kustzone werd verzameld in **hoofdstuk 6** om de intrinsieke biologische waarde van verschillende Belgische kustsubzones te berekenen. Deze subzones werden gedefinieerd als intertidale of ondiepe subtidale Belgische zones die 250 m breed zijn voor de benthische componenten en 3 km breed zijn voor de mobiele soorten zoals vogels. De ruimtelijke dekking en de hoeveelheid beschikbare data waren voldoende om de observatie van significante trends en patronen toe te staan. Ook al bestaat de Belgische kust voornamelijk uit zandstranden, toch is er een biologische diversiteit aanwezig tussen de verschillende subzones. Er werd een sterk mozaïek patroon in biologische waarde aangetroffen langsheen de kustlijn. Bovendien is er een duidelijk tekort aan (benthische) data in het oostelijke deel van de Belgische kust. Rond 70 % van de ondiepe subtidale subzones scoorden hoge biologische waarden in tegenstelling tot de aangrenzende intertidale subzones. Hoge tot zeer hoge biologische waarde werd steeds aangetroffen in de intertidale zones die zich aan de oostelijke zijde bevinden van de havens Nieuwpoort, Oostende en Zeebrugge. De aanduiding van mariene reservaten palend aan beschermde stranden en net naast de Belgische havens is dus van prioritair belang om een succesvolle en ecologisch onderbouwde implementatie van strandreservaten te bereiken. Een gedetailleerde analyse van beschermde gebieden en gebieden met een hoog overstromingsrisico toonde aan dat mariene biologische waarderingskaarten duidelijk aantonen welke strandsubzones vatbaarder zijn voor impact. De biologische waarderingskaarten zorgen ook voor een sterke visuele onderbouwing van het voorstel om een aantal natuurreservaten uit te breiden met een mariene zone. Daarnaast tonen ze ook aan dat we meer data nodig hebben om significantere conclusies te trekken omtrent de biologische waarde van reeds bestaande natuurreservaten. De biologische waarderingskaarten kunnen dus gebruikt worden als betrouwbare basislijn kaarten voor ruimtelijke planning, beleidslijnen en management benaderingen met betrekking tot de Belgische kust aangezien ze een integratie van natuurlijke en ecologische waarde toestaan in een vroege fase van kustbeleidsimplementatie.

In het laatste **hoofdstuk 7** worden beschermingsdoelstellingen voor de Belgische kust gegeven vanuit het perspectief van een bioloog. Daarnaast worden strandonderzoeksresultaten vertaald naar enerzijds aanbevelingen voor zandsuppleties en anderzijds beleidsrichtlijnen voor een geïntegreerd strand-en kustmanagement, met een focus op het kustecosysteem. Tenslotte werden strandfiches gemaakt die alle informatie bundelen voor de 17 intertidale en 11 ondiepe subtidale kustlocaties bestudeerd in deze doctoraatsthesis. In een duidelijk en gebruiksvriendelijk formaat geeft elke strandfiche een overzicht van de locatie van het strand, de legale omstandigheden (provinciale ruimtelijke uitvoeringsplannen en beschermingsstatus), mogelijke kustverdedigingsactiviteiten, biologische waarde en huidige wetenschappelijke kennis. Daarenboven werden ook criteria voor een ecologisch strand geformuleerd, werd een pleidooi gehouden voor een multidisciplinair en geïntegreerd ruimtelijk strand-en kustplan en werden enkele suggesties opgesomd voor toekomstig strandonderzoek.

Chapter 1: General introduction

In this chapter, a general overview of the physical and biological features of sandy shores, i.e. beaches, is given with a focus on the specific habitat of this PhD thesis, namely the Belgian beaches. Its ecosystem components, food web structure and threats are discussed with a prime focus on coastal defence activities and their impact on the beach ecosystem. At the end of this chapter, the current status of the governance and policy in the Belgian coastal zone is documented to provide for a better understanding of beach and coastal spatial planning in Belgium. Finally, the aims of this PhD thesis and the thesis outline are presented.

1. Coastal zones worldwide

Being the spatial interface between the land and the ocean, coastal zones are dynamic ever-changing environments under the influence of earth's natural processes. The constant sculpture of the coastline is defined by a wide array of physical factors, e.g. temperature, salinity, tides, currents, wind, wave action, light and substrate (Levinton 1995). Tides are the rise and fall of sea levels caused by the combined effects of the gravitational forces exerted by the moon and the sun and the rotation of the earth. The tidal range is defined as the vertical difference between the high tide and the succeeding low tide. It is not constant and varies globally due to tidal driving forces, geographic location, volume of water and geography of the water basin adjacent to the coast. Larger water bodies have higher tidal ranges while the geography can act as a funnel amplifying or dispersing the tide. Areas with high tidal ranges (macrotidal range > 4 m) allow waves to reach further up the shore while areas with lower tidal ranges (mesotidal range = 2 – 4 m; microtidal range < 2 m) produce deposition of sediment at a smaller elevation interval (Davies 1964). Sediment size and sorting will contribute to cross-shore beach gradient and type. With increasing grain size the beach face will steepen and the surf zone narrow. Coarse sand, cobble and boulder beaches have the steepest beach face and no surf zone (McLachlan et al. 2013). The surf zone is the most dynamic part of the beach owing to the energy released by breaking waves, which can produce onshore (wave bores), along-shore and offshore (rip currents) flows, and morphology containing single and multiple longshore and transverse bars, troughs and channels (McLachlan et al. 2013). Sand transport, driven by waves on the wet side and wind on the dry side, is highest in exposed surf zones, whereas sand storage is often greatest in well-developed dunes (Defeo et al. 2009). Given this large variety in global geomorphodynamics, a huge diversity in coastal habitats can be found from the Poles to the Equator, ranging from coastal dunes, rocky and sandy shores, mudflats, mangroves, salt marshes and tidal wetlands to estuaries, kelp forests, sea grass meadows and coral reefs.

Encompassing this broad range of habitat types, coastal ecosystems provide a wide array of goods and services (Burke et al. 2001). They store and cycle nutrients, filter pollutants from inland freshwater systems, act as sediment sinks and help to protect shorelines from erosion and storms. On the other side of shorelines, oceans play a vital role in regulating global hydrology and climate by constituting a major carbon sink and oxygen source (Beaumont et al. 2007; Beaumont et al. 2008). Coastal systems harbor unique, dynamic and fragile ecosystems with high biological productivity and genetic diversity. They are very important to the health of both marine and terrestrial environments as they are closely linked through the storage, transport and exchange of sand (Schlacher et al. 2008; Defeo et al. 2009).

However, people intensively rely on coastal regions to live as well as for trade, sea-going and recreational interests, leisure and tourism. As such, man transformed coastal ecosystems into centers of human activity. They now host the world’s primary ports of commerce, serve as a major human food source for fish, shellfish and seaweed, and they provide mankind with fertilizer, pharmaceuticals, cosmetics, household products and construction materials.

1.1 Sandy shores

Sandy shores, also called beaches, cover 70 % of all continental margins (McLachlan & Brown 2006). There is no single, agreed-upon definition for a beach. One definition refers to a beach as ‘accumulation of wave-washed, loose sediment that extends between the outermost breakers and the landward limit of wave and swash action’ (Leatherman 1979). Another definition includes ‘the area between the permanent vegetation line seaward to the point of the next geomorphic feature’ (Davis 1994).

The boundaries of a beach are never rigid. They change constantly with seasonal wave activity, tidal range and reduction in sediment supply to the beach (Pilkey & Dixon 1998). Beaches may seem globally uniform continuous sandy plains from the dunes to the sea, but in reality, a variety of beach types exists. Beaches can be defined by the degree of exposure they experience, from very sheltered to very exposed. Wave-dominated beaches in all tidal ranges can also be classified based on their morphodynamic features (Masselink & Short 1993). When conditions are calm and/or the sediment is coarse, the **reflective** beach type occurs (figure 1a). Waves flow upon the beach where a major part of the incoming wave is reflected. Three beach zones appear to be generally valid on this beach type (Reilly & Bellis 1983; Greene 2002):

(1) **supralittoral** or upper beach zone: dry sand area land inwards from the mean high water level (MHW) to primary dune, only wetted by spray, during high spring tide and storms;
(2) **intertidal** or midlittoral beach zone: wet sand area between MHW and the mean low water level (MLW), constantly moist, but not saturated, from incoming tide;
(3) **subtidal** or sublittoral beach zone: seawards from MLW to the continental slope.

The subtidal zone becomes more complex when bigger waves cut back a beach and spread out its sediments. Two subtidal subzones can then be distinguished:

(3a) swash beach zone: area where waves rush up the face of the beach and retreat seaward, usually remaining saturated;

(3b) surf beach zone: area between the water line and where breakers form as waves break.

All beach types are characterized by high temporal variability and sand storage both on the beach and in the surf zone. For instance, **low tide bar/rip-beaches** (figure 1b) have a much steeper high intertidal zone (typically 1.6 % or more) than the other beach types. If wave action is strong and/or sediment particle size is fine, the flat **dissipative** beach type (figure 1c) is created, as present on many European beaches, for instance the Netherlands, France, northwest Spain and Scotland (McLachlan & Jaramillo 1995). Sediments are stored in a broad surf zone that may have multiple sandbanks parallel to the beach. **Ultradissipative** beaches (figure 1d) are flat, wide, undistinctive beaches with a slightly tapered inclination (around 1 %). The sediment consists of fine sands (125 – 250 µm) and very fine sands (63 – 125 µm). Due to the relative lower environmental stress, these beaches harbour the most diverse and dense beach communities (McLachlan 1983; McLachlan & Jaramillo 1995; Degraer et al. 2003b). Runnels can form on an ultradissipative beach and they stay submerged over a longer period of time (Speybroeck 2007).

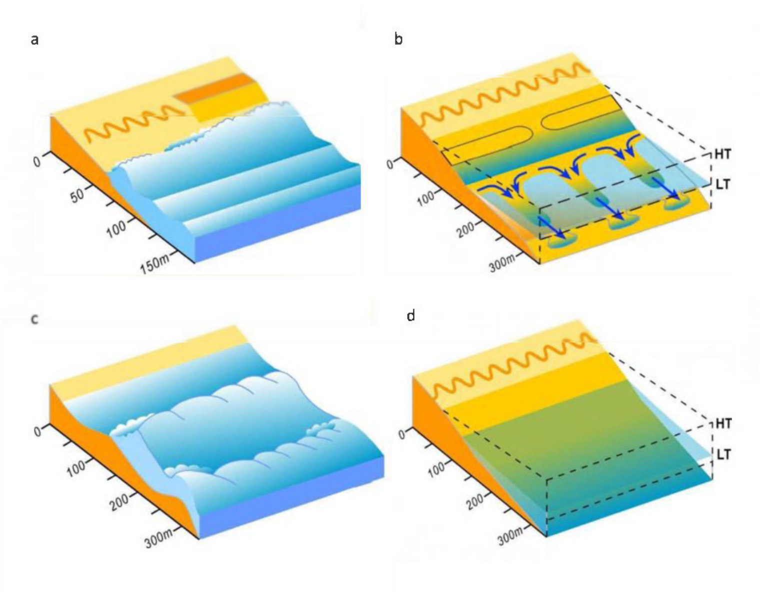


Figure 1: Schematic representation of four beach types: (a) reflective beach; (b) low tide bar/rip beach; (c) dissipative beach and (d) ultradissipative beach (at the courtesy of NIWA, the National Institute of Water and Atmospheric Research of New Zealand, based on figures in (Short 2006))

Beach sand is a naturally occurring granular material composed of finely broken-up rock and mineral particles. The composition of beach sand is highly variable, depending on the local rock sources and conditions. The most common constituent of sand in non-tropical coastal settings is silica (silicon dioxide or SiO_2), usually in the form of quartz. Because of its chemical inertness and considerable hardness, it is the most common mineral resistant to weathering. In tropical coastal settings, where reefs have dominated the ecosystem for millions of years, calcium carbonate ($CaCO_3$), for example aragonite, is the primary form of sand. Beaches in the vicinity of volcanos harbor sand consisting out of tiny fragments of volcanic lava which gives it its black color. Overall beach sand grains are smaller than 1 or 2 mm and larger than silt (< 63 µm). They are divided into five Wentworth classes (table 1) (Wentworth 1922).

Table 1: Beach median grain size divided into Wentworth classes (Wentworth 1922)

Median grain size (mm or µm)	Wentworth class
1 – 2 mm	very coarse sand
500 – 1000 µm	coarse sand
250 – 500 µm	medium sand
125 – 250 µm	fine sand
63 – 125 µm	very fine sand

Coastal waters may be fully saline (35 – 50 ppt), brackish (0.5 – 35 ppt) or nearly fresh (< 0.5 ppt) depending on the vicinity of estuaries, river deltas or melting glaciers. Nutrients are transported by ocean currents and upwelling or they reach coastal habitats through land runoff. Oxygen levels can be increased by wave actions and decreased during algal blooms while carbon dioxide (CO_2) levels influence the acidity of the coastal systems.

For a long time, sandy beaches have been regarded as marine deserts by many biologists and were largely neglected in ecological investigations. Remane (1933) started the sandy beach research on a German beach but it took 50 more years for a first comprehensive overview of the ecological features of sandy beaches (McLachlan 1983). The physical and chemical zonation appeared to induce the apparent dynamic and variable biological zonation. Based on species characteristics and adaptations to an aquatic, terrestrial or amphibian existence, the natural beach zonation is usually most distinct but narrowest at the top of the shore and becomes less clear but widened down shore. Rhythmic migrations of highly mobile organisms shuffle and recreate zones in response to tidal, photic and semi-lunar cycles, substrate moisture, swash activity, slope temperature and turbulence. As the tide rises, zones compress while some populations move in or over the sediment and some enter the water column. The benefits of migration are to keep species (1) in optimal feeding zones with the largest prey and/or nutrient availability, (2) out of reach of bird and fish predators, by concentrating them in the swash zone and (3) in the zone of sediment reworking, reducing the chances of stranding (McLachlan & Jaramillo 1995). On dissipative, fairly undisturbed beaches, zonation patterns can even be triggered by high species abundances causing intraspecific and interspecific interactions (Defeo & McLachlan 2005).

1.2 Threats to coastal zones and sandy beaches

The popularity of coastal regions has led to profound altering of coastal habitats. Initially, the most widespread and pressing threat was habitat loss through draining, dredging and in some way converting to upland habitat, artificial substrate or open water (Crain et al. 2009). With industrialization, additional threats emerged, particularly byproducts of globalization such as invasive species, disease and nutrient pollution in nitrates and phosphates leading to eutrophication. In terms of food production, overexploitation of fish, shellfish, seaweeds and other marine organisms not only diminishes production of the harvested species but also profoundly alters the biological structure of coastal ecosystems (Maes et al. 2005a). On top of this, coastal ecosystems are strongly threatened by climate change due to expected changes in storm and wave regimes, ocean temperatures, circulation patterns, sea level rise, erosion, flooding and altered sediment budgets (Burke et al. 2001; Harley et al. 2006; Jones et al. 2008). From 1950 to 2009, measurements show an average annual global sea level rise of 1.7 ± 0.3 mm per year, with satellite data showing a rise of 3.3 ± 0.4 mm per year from 1993 to 2009 (Bindoff et al. 2007; Nicholls & Cazenave 2010). The resulting higher impact of storm surges could accelerate erosion and associated habitat loss, increase salinity in estuaries and freshwater aquifers, alter tidal ranges, change sediment and nutrient transport and increase coastal flooding. Changing concentrations of CO_2 in ocean waters will lead to acidification and may affect marine productivity or even change the rate of coral calcification (Kleypas et al. 1999).

Of all coastal habitats, sandy beaches are of the highest economic, social and cultural importance to humans as prime recreational assets. More people interact directly with beaches than with any other type of shoreline worldwide (Phillips & Jones 2006; Schlacher et al. 2008). Strong tidal currents are responsible for beach **erosion** worldwide but natural sandy beaches function as a buffer between sea and land, thus protecting the hinterland from scour, inundation and wave erosion (Young & Bryant 1992; Defeo et al. 2009). For centuries, a wide array of human disturbances has shaped and molded sandy beaches with varying impacts, e.g. pollution, eutrophication, tourism, recreation and coastal defence (Brown & McLachlan 2002; Defeo et al. 2009).

2. Belgian coastal zone

The Belgian part of the North Sea (BPNS) is situated on the northwest European continental shelf and covers 3600 km² or 0.6 % of the overall North Sea surface (figure 2). It is often referred to as Belgium's eleventh province as it comprises almost 11 % of the total Belgian surface area (Degraer et al. 2006). The gently sloping underwater landscape is characterized by a continuous succession of sandbanks and swales or gullies, making the BPNS a rather shallow marine system. Average depth is 20 m with a maximum depth of 45 m (Maes et al. 2005a). The tidal regime of the BPNS is semi-diurnal (Baeye et al. 2010) and the mean tidal range descends from about 5 m at the French border to 4.3 m towards the

Dutch border (Fremout 2002). The horizontal motion in a tide wave, i.e. the particle velocity, is called the tidal stream. Most tidal streams are rotary, although the shape of the ellipse traced out by a tidal stream vector, i.e. tidal ellipse, and the direction of rotation may vary. Tidal current ellipses are elongated in the nearshore area and become gradually more semicircular towards the offshore. Maximum current velocities are as such higher and minima lower in the nearshore area than further offshore (Fettweis et al. 2011). The prevailing tidal currents and wave action, also called hydrodynamics, keep the seawater at the BPNS in continuous motion. At high tide, Atlantic water flows through the English Channel into the North Sea. At low tide, part of this water flows back in the direction of the Atlantic Ocean. The dominant southwest-northeast directed current (> 1 m/s) is oriented parallel to the coast and results in a well-mixed homotherm and homohaline water column (Maes et al. 2005b; Degraer et al. 2006; Fettweis et al. 2011). Combined with the constant water supply by major river systems surrounding the North Sea, e.g. Rhine, Meuse, Scheldt, Yser, Authie, Canche and Somme (north to south), and numerous other processes, including wave action, these currents result in a clear seawater gradient: from turbid and nutrient-rich near-coastal water to more transparent and nutrient-poor offshore water (Fettweis et al. 2010). The combination of a complex bathymetry, hydrodynamics and meteorological conditions is also responsible for a high diversity of sediment types on the BPNS varying from very fine mud to coarse sand (Verfaillie et al. 2009).

The coastal zone stretches over 67 km, is southwest to northeast directed and consists mostly of sandy beaches with sea walls in front of the cities and dunes in between. For the purposes of this PhD, the Belgian coastal zone has been defined to include the intertidal and shallow subtidal areas between MHW and the 1 nautical mile from the 0 m depth bathymetric contour. Wherever appropriate, we also included the dune area and its very specific ecosystem (figure 2).

Degraer et al. (2003) stated that a gradual transition is visible from west to east, from ultradissipative beaches, with occasional runnels, to low tide bar/rip beaches (see also figure 1 for the beach types). According to Speybroeck et al. (2008a), Belgian beaches are (ultra-)dissipative, macrotidal, and wide from a morphodynamic perspective. Due to the relative lower environmental stress, the western beaches closest to the French border harbor the most diverse and dense beach ecosystem (Cattrijsse & Vincx 2001; Degraer et al. 2003b). The width of the intertidal zone varies from 200 to 700 meters, decreasing towards the east. Belgian beach sand mainly consists of quartz and has an average median grain size varying between 200 µm and 220 µm, with a minimum of 160 µm and a maximum of 380 µm (Speybroeck 2007). It can be said that the surficial sediments along the shallow subtidal Belgian coastal zone fine in a northeast direction although this is largely dependent on the interaction between the morphological features that may cause an enhanced flow-topography interaction (Degraer et al. 2003a). The waves and tidal currents give rise to a residual coastal sediment drift towards the northeast in the subtidal and intertidal beach zones (Van Lancker et al. 2007; Baeye et al. 2010). The dominant southwestern winds induce a northeastern aeolian drift in the supralittoral (Speybroeck 2007). As coarser sediments deposit in places with a strong current or with strong wave action, the grain size increases from west to east and from MLW to MHW (Deronde et al. 2006; Verfaillie et al. 2006; Van Lancker et al. 2007; Speybroeck et al. 2008a).

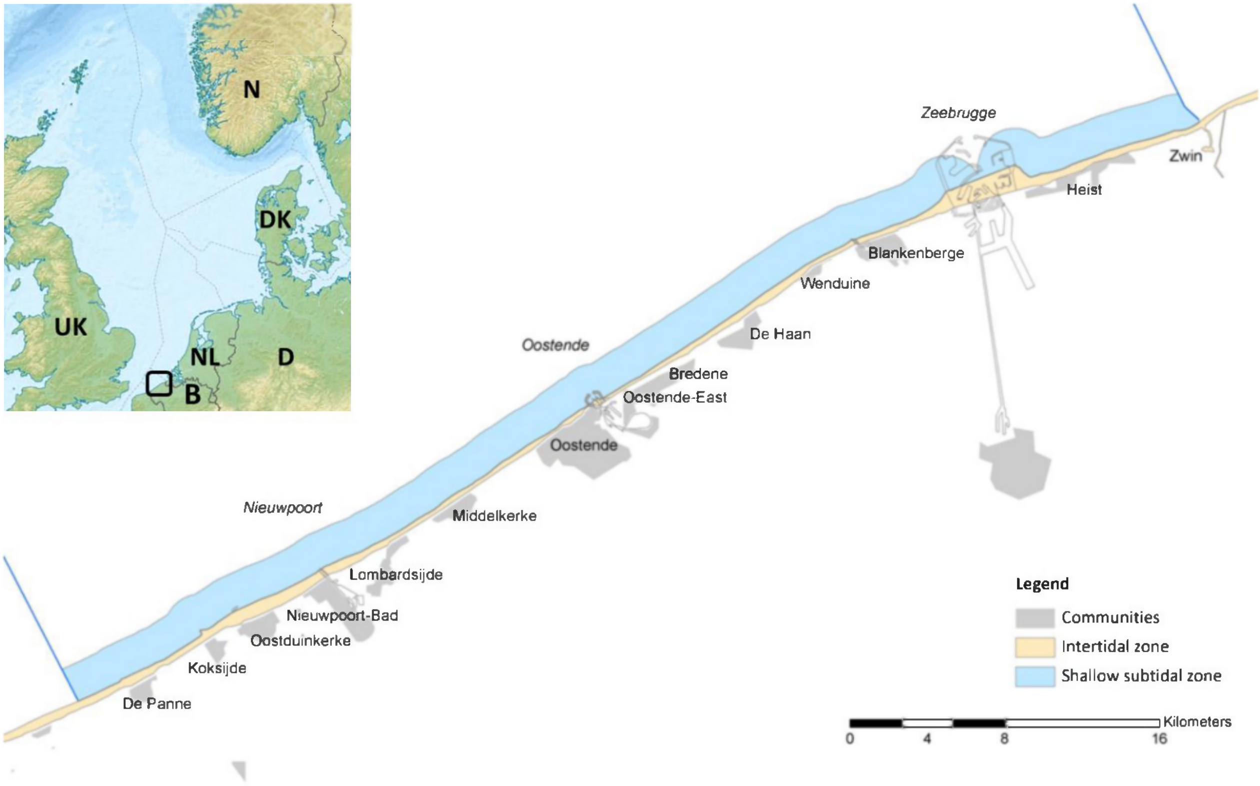


Figure 2: Location of the Belgian part of the North Sea (black box in insert picture) and the Belgian coastal area with its intertidal and shallow subtidal subdivisions; the most important coastal communities and/or sampling locations used in this PhD thesis have been indicated on the map, the three prime harbors are named above their respective locations

3. Belgian coastal ecosystem

Marine biologists have intensively studied the BPNS since the early seventies. A complete overview of all biota, from the plankton to the higher trophic levels still lacks but several components are well known, e.g. benthos and fish populations (Vandepitte et al. 2010a; Vandepitte et al. 2010b). According to the general knowledge, the Belgian marine food web consists of at least 27 mammal species (four are spotted regularly), 75 seabird species, 120 fish species and a huge species diversity in bacteria and viruses, plankton and benthos (Copejans & Smits 2011). Species richness and biodiversity are lower on Belgian beaches but every beach zone has its own characteristic ecosystem with specifically adapted biological components (Speybroeck 2007). The intertidal zone is an unstable environment, prone to repetitive small-scale impact by fast fluctuations in its physical and chemical conditions and irregular large-scale impacts, e.g. storms. In the following paragraphs, the Belgian coastal ecosystem will be illustrated per beach zone, e.g. supra-littoral, intertidal and shallow subtidal zone (figure 3).

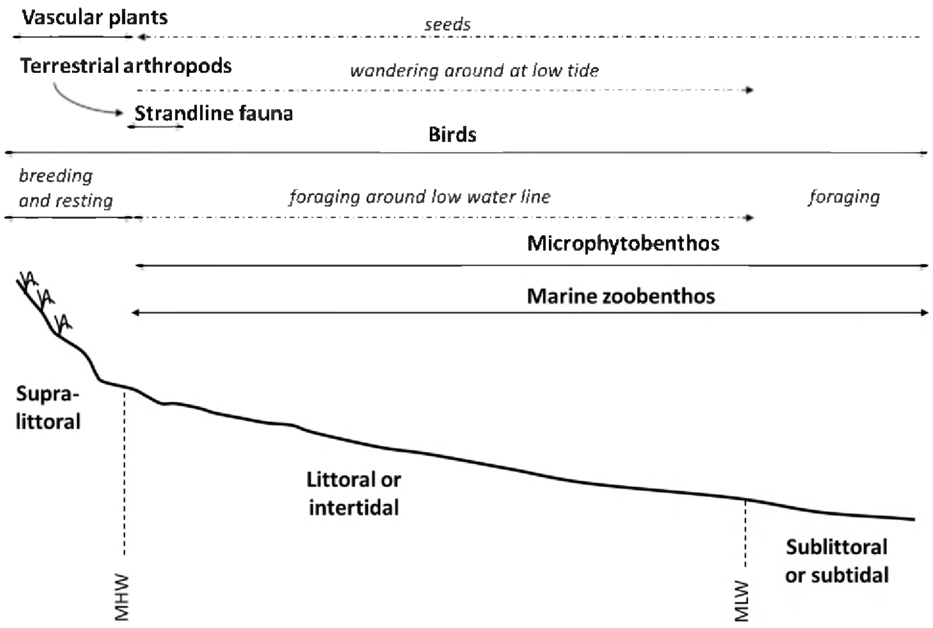


Figure 3: Belgian beach zones with characteristic fauna and flora zonation pattern (adapted from Speybroeck 2007); MHW: mean high water level; MLW: mean low water level

3.1 Supralittoral zone: vascular plants and strandline fauna

The supralittoral ecology, mainly characterized by vascular plants and arthropods (see figure 3), is determined by geomorphodynamics (Provoost et al. 2011). A positive sand budget is essential in this highly dynamic environment. On erosive beaches, strandlines appear and disappear too quickly to settle and develop plant or invertebrate communities. On sedimentary beaches however, vascular plants can sprout, grow and execute an entire phenological cycle, allowing for primary production, a more diverse ecosystem and highly evolved supralittoral vegetation (Speybroeck 2007). Stranded material or wrack, mainly kelp and brown algae, enable rich invertebrate strandline communities. The nutrient influx of decomposing wrack even elevates the vascular plants' vitality and fixates the sediment firmly.

The Belgian supralittoral zone consists of embryonic dunes, dry beach area and strandline. Most vascular plant species are short-lived and adapted to the dynamic nature of this biotope. They disperse and easily colonise strandlines by means of floating seeds that resist seawater for a long time (thalassochory) (Rappé 1996; Rappé 1997). The most common species along Belgian beaches is sea rocket (*Cakile maritima*), often accompanied by prickly saltwort (*Salsola kali*) and sea sandwort (*Honckenya peploides*). All typical species of the supralittoral zone are classified on the Red List as rare to (highly) endangered (Van Landuyt et al. 2006). Phytosociologically, Belgian grey dune vegetation includes moss dunes and dune grasslands mixing with the moss-dominated and lichen-dominated Cladonio-Koelerietalia vegetation. Wild asparagus (*Asparagus prostrates*) is strictly Atlantic and, at least regionally, rare and limited to grey dunes, defined as fixed coastal dunes with herbaceous vegetation in the Coordination of Information on the Environment biotope classification (Evans 2012) in Flanders and Belgium (Provoost et al. 2004).

From an invertebrate fauna perspective, grey dunes are the most endangered dune habitat, certainly within a Belgian context. Fragmentation of grey dunes due to tall grass and scrub encroachment causes substantial loss of regional biodiversity in Flemish coastal dunes (Provoost et al. 2004) and decreases the number of typical invertebrate species within each isolated patch (Bonte et al. 2002; Grootaert & Pollet 2004; Bonte & Hoffmann 2005). Most characteristic dune invertebrates however, are found in more dynamic habitats such as mobile dunes and young dune slacks (Provoost 2004, Bonte 2005). Besides the sandhopper (*Talitrus saltator*), a number of fly and beetle species make up the most typical Belgian supralittoral fauna. The sandhopper is a dominant species (Lincoln 1979) and plays an important role as primary consumer of the organic strandline matter (Robertson & Mann 1980; Griffiths et al. 1983; Stenton-Dozey & Griffiths 1983; Adin & Riera 2003). Other typical decomposers living in and near the strandline are flies and mosquitoes (Diptera, (Grootaert & Pollet 2004)) and their larvae, predators and parasites, two benthic springtails (Collembola: *Folsomia sexoculata* and *Isotoma maritima*) (Janssens 2002), predator mites (*Gamasina*), feeding on springtails and other invertebrates (Koehler et al. 1995; Salmane 2000) and 14 beetle families (Coleoptera) (Haghebaert 1989) with 46 strandline species. If a natural connection between the dunes and the beach is (still) present, some common dune species (isopods, spiders and carabids) can be encountered in the wrack. Beach restricted spiders (Aranea) are absent on Belgian beaches although Red List dune species may sometimes be found (Maelfait et al. 1998).

3.2 Intertidal and shallow subtidal zone: benthos

Benthic species inhabiting the highly dynamic intertidal and shallow subtidal environment possess a high tolerance towards several forms of environmental stress. Normal seasonal fluctuations within their composition, numbers and biomass are an adaptive feature to physical variation in their habitat (Oliver & Slattery 1976; Buchanan et al. 1978; Adriaanse & Coosen 1991). The onshore – offshore and eastern coast – western coast gradients have been reported for all benthic assemblages (Cattrijsse & Vincx 2001) and for pelagic communities such as the phytoplankton and zooplankton (M'Harzi et al. 1998).

The lower layer of the water column, 1m above the seabed, is inhabited by the **hyperbenthos**, a group mainly consisting of small crustaceans, like shrimp (e.g. *Crangon crangon*) and mysids. They consume detritus, algae and zooplankton and serve as prey for young fish and shrimp (Dewicke et al. 2003). High

densities are reached in regions with a strong input of organic matter to the bottom environment (Mees & Jones 1997; Dewicke et al. 2003; DFO 2004).
In the Belgian coastal zone, hyperbenthos is dominated by mysids (Mees et al. 1994; Mees & Jones 1997; Beyst et al. 2001a). They migrate with the tide in and out the intertidal zone to feed and to escape predators from deeper waters (McLachlan 1983; Gibson & Robb 1996; Beyst et al. 1999b; Gibson & Yoshiyama 1999; Wilber et al. 2003).

The seabed surface is the habitat of the **epibenthos**, a community of large, active organisms, including sea stars, brittle stars, crabs, lobsters, bottom fish and cephalopods. The surf zone supports abundant fish resources comprised of small species and juveniles (Modde & Ross 1981; Ross et al. 1987; Brown & McLachlan 2002; Beck et al. 2003). The diet of these fish changes with their developmental stage and prey availability. Populations are generally denser and more diverse in the summer and early fall (Naughton & Saloman 1978; Saloman & Naughton 1979; Modde & Ross 1981).
The Belgian intertidal zone serves as a nursery for the common littoral crab (*Carcinus maenas*) and a whole range of juvenile flat fish species, e.g. plaice (*Pleuronectes platessa*), sole (*Solea solea*), brill (*Scophtalmus rhombus*), turbot (*S. maximus*) and dab (*Limanda limanda*) (Beyst et al. 1999b). These juvenile flat fish migrate with the tide in and out the high intertidal zone to feed on epibenthos (Mees & Jones 1997; Hostens & Mees 1999) and macrobenthos and to escape predators from deeper waters (McLachlan 1983; Gibson & Robb 1996; Beyst et al. 1999b; Gibson & Yoshiyama 1999; Wilber et al. 2003).

The surf zone and nearshore regions are important migratory routes used by both hyperbenthos and larval and juvenile fish (epibenthos). They travel parallel to the coast and move easily in and out inlets and estuarine nurseries or back and forth between shallow and deeper waters (Hackney 1996; Beyst et al. 2001a; Beyst et al. 2002).

The **microbenthos** consists of unicellular organisms, namely diatoms, ciliates and bacteria, living between and on the sand or silt grains. The **microphytobentos** (MPB) are microscopic algae living on benthic surfaces at the photic marine zone. They are the most important primary producers on apparently unvegetated coastal zones and their biomass supports higher trophic levels.
On Belgian beaches, the MPB consists of dinoflagellates, euglenoids and both epipsammic (<10µm, living on sand grains) and epipelic (free living, forming biofilms) diatoms (Sabbe 1997; Speybroeck 2007; Maria et al. 2011a). The composition and occurrence of MPB on Belgian beaches depends on season, beach height position, hydrodynamics and grain size. The availability of inorganic nutrients, like nitrogen and phosphor is also important but remains poorly studied (Underwood & Kromkamp 1999). The highest levels of MPB appear in summer, due to optimal temperature and light conditions. On muddy sediment this peak shifts to the spring (Sabbe 1997).

Meiobenthos groups all organisms smaller than 1 mm but bigger than 38 µm, living buried in the seabed. This group is characterised by a large variety of invertebrates, including copepod crustaceans (coarser sand) and roundworms or nematodes (finer sands) (McLachlan 1983). Nematodes feed on bacteria, microphytobenthos, other meiofauna, detritus and dissolved organic matter (McLachlan 1983) while copepods prefer microphytobenthos (Granéli & Turner 2002).

In Belgium, knowledge on intertidal meiofauna is restricted to the meiofaunal community of the western part of the coastline (De Panne / Koksijde) (Gheskiere et al. 2002; Gheskiere et al. 2004; Gheskiere et al. 2005; Maria et al. 2011b; Maria et al. 2012a). In general 15 meiofauna taxa were recorded with Nematoda, Harpacticoida and Turbellaria (Martens 1983, 1984; Martens & Schockaert 1986) being the dominant ones. Higher densities were found in the lower intertidal zone while diversity peaked in the middle of this zone. Three semi-separated Nematoda species associations were detected: (1) supralittoral: *Rhabditis sp.* and *Axonolaimus helgolandicus*; (2) high intertidal zone: *Trissonchulus sp.*, *Dichromadora hyalocheile* and *Parachromadorita sp.*; (3) low intertidal zone: e.g. *Odontophora phalarata*, *O. rectangula*, *Cyartonema elegans* and *Chaetonema riemanni* (Gheskiere et al. 2004).

Macrobenthos is generally defined as the organisms measuring over 1 mm long and living buried in the seabed. Of the marine zoobenthos, this group of bivalves, polychaetes, crustaceans and echinoderms is best investigated. On a world-wide scale, crustaceans tend to be most abundant on exposed beaches, while molluscs and polychaetes abound on sheltered beaches (McLachlan & Jaramillo 1995; Elliott et al. 1997). Macrobenthos performs well as an indicator of pollution and stress and plays a key role in the wider beach ecosystem and food web. They feed primarily on faunal detritus and to a lesser extent on algal benthos and detritus (Sundbäck & Persson 1981; Josefson et al. 2002) and they are a major food source for birds and epibenthos. Macrobenthos is less abundant on sandy beaches than meiobenthos but comprises a larger part of the total biomass (Greene 2002).

Around 265 macrobenthic species have been discovered in the BPNS (Elliott et al. 1997; Degraer et al. 1999b; Cattrijsse & Vincx 2001; Degraer et al. 2003b). Their spatial distribution shows variability along the cross-shore gradient (figure 4), from sparse high intertidal to diverse shallow subtidal communities (Speybroeck 2007). A very narrow high intertidal zone is the habitat of the amphipod *Bathyporeia pilosa* and the polychaete *Scolelepis squamata*. The isopod *Eurydice pulchra* and the amphipod *Bathyporeia sarsi* live in a wide zone in the middle of the intertidal area. In the lowest parts of the intertidal zone, several polychaetes, e.g. *Nephtys cirrosa*, and bivalves have to share the space. The current zonation, distribution, abundance and species characteristics of *Bathyporeia pilosa* and *Bathyporeia sarsi* on Belgian beaches are likely to be the result of both niche diversification and character displacement (Van Tomme 2013). Although abiotic factors are defining the upper zonation limits of both amphipods, as has been generally accepted for sandy beaches (McLachlan 1996; McLachlan 2001), recently it was shown that the lower limits can only be explained by a combination of abiotic and biotic forces (Van Tomme 2013). The subtidal zone is a more buffered system, both physically and biologically controlled, mainly by sedimentology and geomorphology. More than 100 species are adapted to several subtidal microhabitats (Van Hoey et al. 2004), making the subtidal zone a species diverse system where competition for food and place reign (Van Hoey et al. 2007b; Van Hoey et al. 2010).

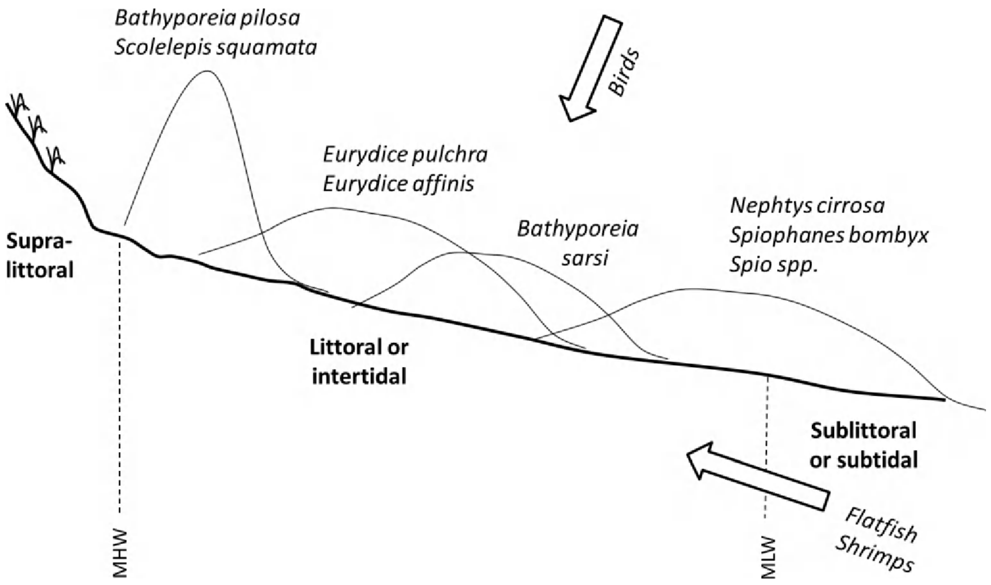


Figure 4: Zonation pattern of different Belgian macrobenthic species (adapted from Van Tomme 2013, originally adapted from Speybroeck 2007); MHW: mean high water level; MLW: mean low water level

Runnels on ultradissipative beaches were largely neglected in macrobenthos beach research because they need a different sampling strategy than generally applied. They contain a benthic fauna resembling the subtidal communities (Boulez 2002). Both in abundance and diversity, this fauna exceeds that of neighbouring sand banks since the runnels stay submerged over a longer period of time and contain higher levels of organic matter (Speybroeck 2007). Recent meiobenthos studies indicated that nematode communities from runnel and sandbar habitats are significantly different, illustrating the importance of microhabitat heterogeneity (Maria et al. 2012b).

3.3 From supralittoral to shallow subtidal: birds and seals

Some bird species use one or several beach zones for resting, nesting, moulting, breeding, foraging or any combination of these activities. The supralittoral zone is an important area in our region for birds, especially in winter and during migration. The intertidal and shallow subtidal zones form an important foraging area for many birds that feed primarily on macrobenthos. Seal activity on beaches is closely related to tidal cycles. At low tide, seals like to rest on sand banks, sand flats, hard defence structures, like groins, or even floating devices, like pontoons and buoys that allow for easy escape in case of hazard or danger. If the sand banks or sand flats offer enough peace and quiet, they could even be used as spawning areas.

In the Belgian supralittoral zone, turnstone *Arenaria interpres* feeds on strandline material (Smit & Wolff 1981; Becuwe et al. 2006). Only three Red List species can breed here, e.g. Kentish plover *Charadrius alexandrinus,* little tern *Sternula albifrons* and common ringed plover *Charadrius hiaticula*. These three breeding birds are threatened with extinction (Vermeersch et al. 2004) and their breeding distribution is limited to Zeebrugge port and the adjacent reserve 'Baai van Heist' (Stienen & Van Waeyenberge 2002; Courtens & Stienen 2004; Stienen & Van Waeyenberge 2004; Stienen et al. 2005). Intertidal macrobenthos of easily penetrable, wet substrates along the edges of gullies and along the MLW are the primary food source for many gulls and wading birds, e.g. oystercatcher *Haematopus ostralegus*, dunlin *Calidris alpina* and sanderling *Calidris alba* (Engledow et al. 2001; Stuer 2002; Speybroeck et al. 2005a; Speybroeck et al. 2005b). At high tide, they use the supralittoral to rest or gather before moving to high water roosts located on groins or near larger tidal flats, e.g. reserves 'IJzermonding' and 'Baai van Heist' (figure 2). The shallow subtidal Belgian waters and their associated food resources are of international importance for a number of seabirds, at least in a specific season (Seys 2001; Van Waeyenberge et al. 2001; Stienen & Van Waeyenberge 2002). These species are common scoter *Melanitta nigra*, crested grebe *Podiceps cristatus*, little gull *Larus minutus*, little tern, common tern *Sterna hirundo* and sandwich tern *Sterna sandvicensis*. Internationally of less importance but with a strong coastal connection are black-headed gull *Chroicocephalus ridibundus* and common gull *Larus canus* (Spanoghe 1999; Spanoghe & Devos 2002).

Seals are only sporadically spotted in the Belgian coastal waters, with harbor seal *Phoca vitulina* and grey seal *Halichoerus grypus* as most common visitors (De Smet 1978; Rappé 1983; Van Gompel 1983; Haelters 1999; Degraer et al. 2009; Jonckheere 2011). Both seals preferentially prey on benthic fish and small crustaceans in shallow waters. Since 2008, a small (maximum count of 16) but resident group of harbor seals stays at the Belgian coast. They currently prefer the estuaries of both the Schelde and Ijzer rivers as resting areas though one groin in Koksijde (Ster der Zee) seems to have served the same purpose (Jonckheere 2011). All Belgian beaches are too heavily used and too gently sloped to accommodate seals as these two factors both hinder fast escape into the sea in case of hazard or danger (Degraer et al. 2009).

4. Threats to the Belgian sandy beach ecosystem

An inventory of coastal evolution in the European Union showed 55 % of the coastline to be stable, 19 % to be suffering from erosion problems and 8 % to be depositional. The remaining 18 % of the coastline cannot be assigned to any of the categories (Airoldi et al. 2005). Research based on sequential beach profiling revealed that a natural cycle explains the periodical behavior of erosion and accretion on Belgian beaches (De Moor 1979; De Moor & Bloome 1988). Strong tidal currents are responsible for beach erosion at more than 50 % of the Belgian coastline (Deronde et al. 2006). Along the western coast, beaches are mostly stable and accreting. Further east to Oostende, beach sedimentology does not evolve in accordance with any clear trend. Beaches from Bredene to Wenduine and beaches in front of dykes with a rather pushed-forward position, e.g. Knokke, show increasingly irregular beach profiles. They have been and are still subjected to severe management measures which provoked a permanent erosive situation (De Moor & Bloome 1988; De Wolf et al. 1993; De Wolf 2002; Speybroeck 2007). The harbor walls of Belgian's biggest harbor, Zeebrugge, have profoundly altered the beach morphodynamics and morphology of all beaches situated at its eastern side, creating a deviation from the gradual transition in beach types along the Belgian coast (Deronde et al. 2008). Beaches closest to the Dutch border are more or less stable. Erosion only occurs on the beach at the mouth of the nature reserve 'Zwin' (figure 2).

4.1 Coastal defence along the Belgian coastline

Since the Middle Ages, man has strived to keep the Belgian coastline at its position or even move it seaward by drastically altering beaches up to the point where they are no longer capable of providing their natural coastal defence services. The low elevation of the Belgian beaches makes them even more vulnerable to detrimental erosive forces, sea level rise, storms and the consequent higher possibility of hinterland flooding. Unfortunately, human retreat in areas of low-value land and relocation is not achievable because of resistance to regulation of this coastal defence approach by both the general public and politics and the high economic value placed on coastal and port properties (Grober 1992). Instead, a large part of Belgium's only 67 km of shoreline is protected by hard defence constructions, like groins, concrete dykes, seawalls and revetments (figure 5) (Hanson et al. 2002). However, the construction and enhancement of these structures enhanced beach erosion (Airoldi et al. 2005; Defeo et al. 2009) and destroyed important ecosystem functions (Martin et al. 2005). These hard barriers also lead to the 'coastal squeeze' phenomenon whereby less and less space is available for natural coastal processes to accommodate eroding forces or adjust to the changes in sea level, storms and tides (Doody 2005; Schlacher et al. 2007; Nicholls & de la Vega-Leinert 2008). Nowadays, confidence has been established in soft coastal defence techniques, like nourishment (Dankers et al. 1983; Adriaanse & Coosen 1991; Charlier et al. 1998; Basco 1999; Peterson et al. 2000; Brown & McLachlan 2002; Finkl & Walker 2002; Greene 2002; Hamm et al. 2002; Hanson et al. 2002). The philosophy behind **nourishment** is based on the consideration that when a stretch of coast is sediment-starved, it could be more appropriate to import sediment and let nature do its job, rather than desperately try to counteract natural forcing factors to keep the remaining sediment. There are however, no universal nourishment concepts. Examples of basic design objectives for a well-defined period are: (1) improving coastal stability by keeping the MLW position seawards of a selected position, (2) improving coastal protection to maintain a certain amount of sand (m^3 per m) or (3) increasing and maintaining a certain beach width.

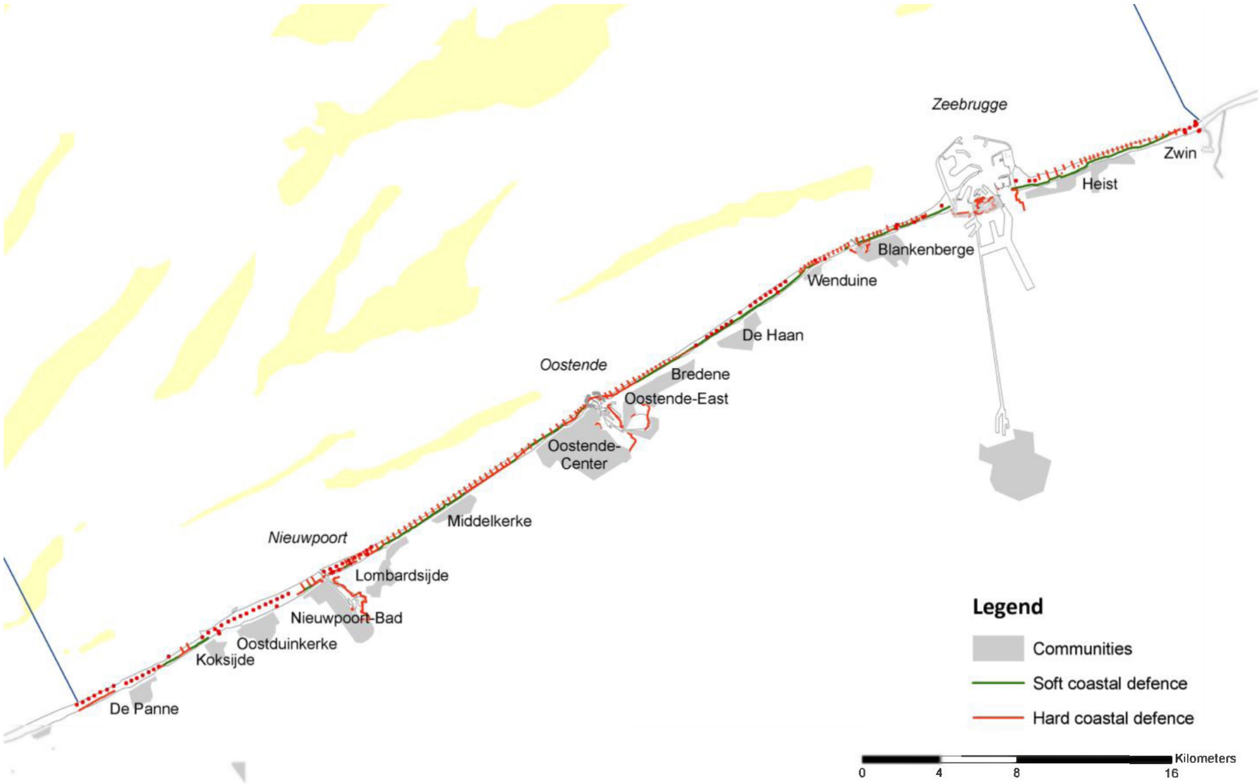


Figure 5: Hard and soft coastal defence structures along the Belgian coastline (shapefiles from (Belpaeme & Konings 2004)

There is some debate as to the most effective nourishment position to achieve optimum protection. Possible locations include the upper beach and dune face (i.e. backshore nourishment), the intertidal beach (i.e. beach or profile nourishment) and the shallow foreshore zone (i.e. foreshore nourishment). Backshore nourishment benefits are immediate, but if the sediment is primarily sand, it will be rapidly redistributed along-shore or across the beach face by waves and currents to form a new equilibrium profile (Greene 2002). Beach nourishment anticipates this redistribution, and provides shoreline protection by helping to dissipate wave energy before it reaches the dunes. Foreshore nourishment also anticipates the gradual redistribution of sand into the beach system but the technique is mostly used in areas where coastal protection measures have steepened the coastal profile or in areas with a long-term sediment deficit. Periodic **beach nourishment** has rapidly become a widely applied protective measure worldwide, for both short-term emergencies (i.e. storm-induced erosion) as well as long-term issues (i.e. structural erosion and sea level rise). It is generally considered as the less harmful beach management option because it safeguards the natural dynamics of the coast (Hamm et al. 2002). Beach nourishment projects can be augmented with dune construction and hard structures to provide a desired level of protection at the site (Greene 2002). In 1956, Belgium undertook the largest beach nourishment project in the world at that time on the beach of Knokke (De Moor & Bloome 1988). Since then, the entire Belgian coastline is regularly nourished to physically maintain and safeguard its beaches. Unfortunately, there are no reliable data available to provide an overview of the whereabouts or regularity of the multiple nourishment projects performed since 1956. Acute and maintenance nourishment projects are performed on a regular basis on several Belgian beaches but the information concerning these projects remains poorly documented. The problem lies with the defragmentation of duties and responsibilities along the Belgian coastline with information scattered between coastal communities, local governments, the Flemish and Belgian government (prior to federated Belgium). Figure 5 gives an overview of the current position of several hard and coastal defence structures. On 10 June 2011, the Flemish government approved The Integrated Coastal Safety Plan. This plan contains a series of measures and alternatives to be taken between now and 2050, guarding against the danger of a superstorm and preventing present and future flooding (Mertens et al. 2008). For the next years, Belgian beaches will thus face a multitude of coastal defence activities, including large-scale long-term beach nourishment projects.

However, being an impact, beach nourishment does put a pressure on the biota living on, in and around sandy beaches (Speybroeck et al. 2006a). Peer-reviewed impact studies and adequate information on the consequences of nourishment however are scarce (Jones et al. 2008; Leewis et al. 2012; Schlacher et al. 2012). It remains difficult to predict the impact of nourishment on the beach ecosystem and to suggest possible ecological adjustments to nourishment projects. Species and their habitat could be impacted directly, indirectly or even via cumulative effects in a number of different ways including direct mortality, sublethal impairment and degraded habitat (Essink 1999; Greene 2002). The ecological effects of nourishment can generally be divided into three main groups (Speybroeck et al. 2006a): effects related directly to aspects of the construction phase of the nourishment project (1) and effects related to quality (2) or quantity (3) characteristics of the nourishment sediment (figure 6). Sand for beach nourishment operations is mostly obtained near shore or offshore although nearby channel dredging can also provide the necessary sediment. The total impact effect is influenced by place, time and size of the nourishment project next to the chosen nourishment technique and strategy.

Greene (2002) stressed the possibility of temporal and spatial cumulative effects and the synergisms between them both. Since nourishment is a temporary solution, periodic long lasting additions of sand every three to ten years are constantly required to maintain the width of the beach (Grober 1992). Currently, little is known of the physical and ecological effects of these repeated nourishment projects but they can initiate compaction of the beach sediment, long term elevated turbidity and permanently altered sediment composition and beach morphology. The combined effects of simultaneous nourishment projects along an entire coastline should be considered as well. Several short projects are advised over long lasting huge ones, especially in areas where short term morphological changes are unpredictable (Adriaanse & Coosen 1991; Löffler & Coosen 1995; Peterson et al. 2000).

Furthermore, interpretation of nourishment study results must be done with caution. Natural disturbances, like storms, spatial patchiness and natural variability of sandy beach organisms complicate analyses of beach nourishment impacts (Hamm et al. 2002; Hanson et al. 2002; McLachlan & Dorvlo 2005). Natural variation in temperature, salinity, wave climate and weather can mediate changes in benthic diversity, possibly masking or preventing detection of nourishment effects (Greene 2002). Subsequently, most beach nourishment studies could not differentiate natural variation from nourishment impacts (Reilly & Bellis 1983; Peterson & Manning 2001; Van Dalfsen & Essink 2001; Greene 2002; Hamm et al. 2002; Hanson et al. 2002; Wilber et al. 2003; Kuang et al. 2011). It is also important to clearly state and define the term 'recovery' within this context. Full recovery should be defined by (long-term) biological, ecological and physical processes controlling recolonization and succession. The time scale for achieving populations similar to those found prior to nourishment is in any case at least one year, mainly due to pronounced discontinuities in distribution of populations and seasonal fluctuations (Leewis et al. 2012; Schlacher et al. 2012). Benthic recovery will depend on the nourishment survival rate of already present organisms, migrating and recolonizing capabilities of adults and recruitment of young stadia of nearby populations through dispersal. One might consider a benthic community to be recovered when at least 80 % of the species diversity and biomass has been restored (Essink 1999). One should bear in mind that biomass recovers at a more rapid rate than the diversity of species (Adriaanse & Coosen 1991) and pioneer populations, such as polychaetes and other annelid worms, may temporarily exceed the original populations in numbers of individuals and diversity. The severity of the temporary disruption caused to birds and seals is dependent on their activities during the nourishment period as well as on their reliance degree on the nourishment areas. In general, it is assumed that human activities disturb birds within a 500 m radius and seals within a 1500 m radius (Adriaanse & Coosen 1991). Little remains known of recovery rates, overall ecological effects, effects on subtidal communities, large-scale and long-term effects, preferably studied over several seasons and years (Greene 2002). In combination with insufficient baseline data, these knowledge gaps hinder conclusive measurement of beach nourishment effects beyond one year (Grober 1992).

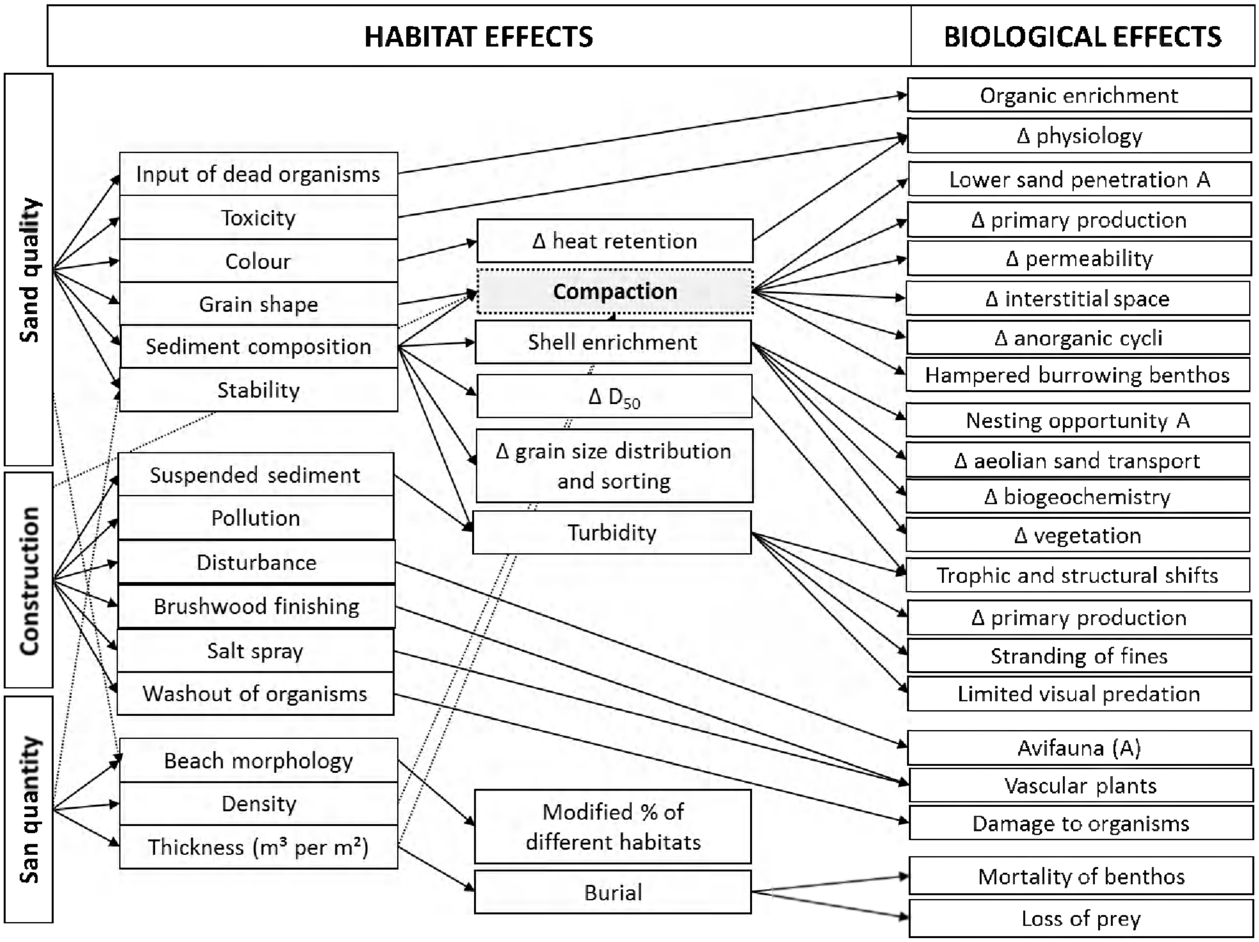


Figure 6: Integrated network of the ecological effects of beach nourishment (Speybroeck et al. 2006a)

4.2 Guidelines for ecologically good practice of beach nourishment along the Belgian coastline

Speybroeck (2006) formulated general guidelines for an ecologically good practice of beach nourishment. From a technical point of view coarser grain sizes produce a steepe, more stable and longer lived nourishment (Finkl 2002). However, to limit the ecological impact, nourishment sands with a **comparable sediment composition** to that of the natural sediment should be used, to allow swift recovery of the benthic fauna (Parr et al. 1978; Nelson 1993; Löffler & Coosen 1995; Peterson et al. 2000) and to avoid a sharp transition from dissipative to reflective beaches (Anfuso et al. 2001). While the impact of sediment color is largely unknown it seems precautionary to use sands with the **same color** as the original sediment. **Toxic substances** should be absent (Adriaanse & Coosen 1991). The preferred time of nourishment is entirely site-specific, depending on the nature and location of the beach and the species inhabiting or exploiting it (Speybroeck et al. 2006a). Spring and early summer provide for least restricted work windows and the nourishment has the greatest chance to stabilize before winter storms start to erode and redistribute the beach. When aiming at a minimal ecological impact in mesothermic zones of the northern hemisphere, nourishment should be completed within a **single winter**, starting after October and ending around March. A number of **smaller projects** (<800 m in length of shore) is preferred over a single large nourishment project (Adriaanse & Coosen 1991; Löffler & Coosen 1995; Peterson et al. 2000). The short distance between nourished and unnourished beach strips allows swift recolonization, depending on species-specific dispersal capacities. Planktonic larvae can disperse over distances well beyond 1000 km. Unfortunately, dispersal distances of permanent meiofauna are limited to 10 km and postlarvae and juveniles are restricted to 10 m. Adult macrobenthos can cover only 1 m or less so their dispersal capacities are very low (Günther 1992). Most benthic species will be less seriously affected if **sediment deposition** is restricted to 0.2 to 0.3 m. No clear best choice can be made among backshore, beach and foreshore nourishment. It seems advisable to decide on the **nourishment technique** in respect to the local natural values of the beach ecosystem. Each strategy has its major impact on a different part of the beach. Choices will have to be made in view of the vulnerability of the organisms residing in each beach zone. Moreover, nourishment needs to be as **cost-effective and efficient** as possible. Costs for a nourishment scheme depend on the source of material, transport methods, volumes required, the need for hard control structures, like groins and breakwaters, the need for secondary defences, expected scheme life before topping up and the amount of minor works undertaken to enhance the dune system (Charlier et al. 1998). Hence, it is indispensable to strive for ecological and economic beach nourishment projects.

5. Governance in the Belgian coastal zone

In the Belgian coastal zone, different government levels exercise competences, including international institutions, the federal government, the Flemish Region, the Province of West Flanders and ten coastal municipalities (figure 7). The federal and regional responsibilities are exclusive, having legal responsibilities only within their precise geographical boundaries, and equivalent with no hierarchy between the standards issued by each group (Cliquet et al. 2007; Cliquet & Decleer 2007). This complex institutional context can cause substantial problems through overlap, conflicts and gaps (Cliquet 2001; Maes et al. 2005b; Douvere 2008). The federal government has jurisdiction over the entire BPNS, including the Exclusive Economic Zone (EEZ, more than 12 nautical miles) and the Territorial Sea (between the MLW and 12 nautical miles). Within this regard, the shallow subtidal coastal zone (between MLW and 1 nautical mile) falls under federal jurisdiction. Federal competences include, among others, environmental policy and protection of the marine environment, wind farms at sea, shipping, military activities, aggregate extraction, cables and pipelines. The Flemish regional authority governs the inland territory, estuaries and inland waters, and the coastal waters above the MLW, including the intertidal coastal zone. The Flemish Region is competent for policy areas such as nature policy on the beach and the hinterland, recreation, ports, fishing, dredging, piloting and coastal defence (Cliquet 2001; Maes et al. 2005b; Cliquet et al. 2007). Within the BPNS, the environmental policy competences are thus shared between the federal and regional levels (Herrier et al. 2005). Dialogue is vital to ensure that Belgium speaks with a single voice within numerous international organisations and bodies (Cliquet et al.

2010). This is the prime task of the steering group on seas and oceans of the Coordination Committee for International Environmental Policy (CCIEP 1995) (Cliquet 2001; Cliquet & Maes 2001). On a day-to-day basis, this group is steered by the Directorate General for the Environment at the Federal Public Service. Specifically for the Belgian coastal zone, the Coordination Centre for Integrated Coastal Zone Management encourages and promotes sustainable and integrated management by allowing a platform to discuss cross-sectorial themes between the federal, Flemish and provincial policy levels (Cliquet 2001).

Figure 7: Coastal legal system in Belgium; 1NM: 1 nautical mile; MLW: mean low water level; MHW: mean high water level; WFD: Water Framework Directive, MSFD: Marine Strategy Framework Directive; ICZM: Integrated Coastal Zone Management (Laporta 2012)

Some of the most important international expectations and obligations in Belgium are to be found in the Convention on Biological Diversity, the Ramsar and OSPAR Conventions, the combined Birds (BD) and Habitats Directive (HD) (Natura 2000), the Water Framework Directive (WFD), the Marine Strategy Framework Directive (MSFD) and the Integrated Coastal Zone Management Directive (ICZM) (see Appendices – Chapter 1 for a more detailed description). On 13 March 2013, the European Commission launched a proposal to improve the planning of maritime activities at sea and the management of coastal areas (Commission 2013a). The proposal now takes the form of a draft directive and it will aim to establish a common European framework for maritime spatial planning and integrated coastal management in EU Member States, with a view to ensure that the growth of maritime and coastal activities and the use of resources at sea and on coasts remain sustainable. The Commission proposal will be considered by the Council of the European Union and the European Parliament. Once adopted, the new initiative will become EU law. In Belgium, a public consultation of the draft directive runs from 2 June 2013 to 29 September 2013 includingly.

In order to meet these international obligations, Flanders drafted the Decree on the Protection of Coastal Dunes (1993) and the Decree concerning Nature Conservation and Natural Environment (1997) (Herrier & Killemaes 2001). The main purpose of these Decrees was to protect the diversity in habitats and species and the entire dune environment with its natural characteristics. Up to that point, the ever expanding construction and urbanization of the Belgian coastline, also known as the Atlantic Wall, threatened to destroy the entire Belgian dune ecosystem. According to the Decrees, designated dune areas have to be at least 2 ha, with a high biological value or they should be deemed irreplaceable areas on the basis of their shape and geomorphological characteristics. A construction ban in the dunes was implemented with nature conservation and coastal defence as its only conceptions.

In 1999, the law of the protection of the marine environment in sea areas under Belgian jurisdiction (Wet Marien Milieu, MMM, in 1999, amended in 2005) established the legal basis for the conservation, restoration and development of nature and the protection of the BPNS against sea-related pollution (Maes et al. 2005b). The general principles of environmental law are summarized in this important act:

(1) prevention principle: prevention is better than cure;
(2) precautionary principle: preventive measures must be taken if there are grounds for concern regarding pollution;
(3) principle of sustainable management: human activities must be managed in such a way that the marine ecosystem remains in a condition which ensures the continued use of the sea;
(4) polluter pays principle: the costs of measures to prevent and fight pollution are to be borne by the polluter;
(5) principle of restoration: if the environment is damaged or disrupted, the marine environment must be restored to its original condition as far as is possible; and
(6) principle of objective liability: the party having caused the damage to or disruption of the environment in sea areas as a result of an accident or an infringement of the law is obliged to remedy this, even if they are not at fault.

In addition, the MMM law established the basis for creating five types of marine reserves: (1) integral marine reserves, (2) specific marine reserves, (3) Special Protection Areas (SPAs) and Special Areas of Conservation (SACs), (4) closed zones for certain activities all year or part of the year and (5) buffer zones in which the restrictions on the activities are less strict than in the marine reserves. For each designated marine protected area, a policy plan must be drawn up (Cliquet et al. 2007). Before and during activities requiring a permit, there is a general obligation to prepare a report on the environmental effects (at the initiative of the applicant) and to undertake environmental assessment (carried out by the government).

The following implementing decrees have been issued in the context of the MMM act:

(1) Royal Decree of 12 March 2000 on the procedure for dumping certain substances and materials in the North Sea (Belgian Official Journal of 4 April 2000);
(2) Royal Decree of 21 December 2001 on the protection of species (Belgian Official Journal of 14 February 2002): complete protection of sea mammals and offshore seabirds (Table 2);
(3) Royal Decree of 7 September 2003 on the procedure for permits required for certain activities in sea areas (Belgian Official Journal of 17 September 2003);
(4) Royal Decree of 9 September 2003 on the assessment of environmental effects (Belgian Official Journal of 17 September 2003);
(5) Royal Decree of 8 July 2005 on the simplified procedure for assessment of environmental effects (Belgian Official Journal of 14 July 2005);
(6) Royal Decree of 14 October 2005 on the installation of special protection areas and special zones for nature conservation (Belgian Official Journal of 31 October 2005), i.e. Natura 2000;
(7) Royal Decree of 14 October 2005 on the conditions for community agreements concerning special protected marine areas (Belgian Official Journal of 31 October 2005);
(8) Royal Decree of 5 March 2006 on the establishment of a focused marine reserve in the sea areas under the Belgian jurisdiction and amending the Royal Decree of 14 October 2005 imposing special protection areas and special areas for conservation in marine areas under the jurisdiction of Belgium (Belgian Official Journal of 27 March 2006);
(9) Royal Decree of 23 June 2010 on the establishment of a framework for achieving good surface water status (Belgian Official Journal of 13 July 2010), i.e. Water Framework Directive;
(10) Royal Decree of 23 June 2010 on a marine strategy for the Belgian marine areas (Belgian Official Journal of 13 July 2010), i.e. Marine Strategy Framework Directive;
(11) Royal Decree of 20 July 2012 on the drafting of a marine spatial plan for the BPNS, excluding the intertidal zone. This plan will organize all spatial and temporal human activities based on a long term vision and clear economic, social and ecological objectives (Belgian Official Journal of 28 November 2012).

More recently, the Flemish Decree on the establishment of the updated monitoring of the water status pursuant to Article 67 and 69 of the Decree of 18 July 2003 relating to the integral water policy was drafted on 26 April 2013 and published on 23 July 2013.

To the present day, there is still no comprehensive legal framework or code specific for the whole Belgian coastal zone nor for either its marine (the federal level) or land part (the Flemish level). There are even no legal instruments on the integrated management of the coastal zone. However, the Belgian federal government is working on a marine spatial plan (as stated in the Royal Decree of 20 July 2012).

6. Current coastal governance status in Belgium and Flanders

In the BPNS, excluding the intertidal zone, Belgium currently counts one SAC (Vlaamse Banken), one contested SAC (Vlakte van de Raan), 3 SPAs (Western coast, Poldercomplex and Zwin), one integral marine reserve (Baai van Heist) and one Ramsar site (Vlaamse Banken) (figure 8) (Cliquet & Decleer 2007; Cliquet et al. 2010). In the coastal zone, including the intertidal and supralittoral zone, Flanders designated five SPAs (Westkust, IJzermonding, Poldercomplex, Kustbroedvogels Zeebrugge-Heist and Zwin), two SACs (Polders and Duingebieden, including Ijzermonding and Zwin), two beach reserves (Baai van Heist and Ijzermonding), two Ramsar sites (Zwin and Ijzermonding) and scattered protected dune areas, mostly incorporated in the SACs (figure 8) (Decleer 2007; Degraer et al. 2010).

For Belgium and Flanders, the annexes to the HD and BD list 16 marine and coastal habitat types (table 2). Different federal and Flemish jurisdiction recognize some level of protection for 17 coastal seabirds, 5 sea mammals, 2 bats, 6 fish, 2 reptiles, 2 amphibians, 2 invertebrates and 2 plants (table 3). Under the Natura 2000 obligation, Flemish and Belgian conservation objectives and measures (regional; G-IHDs) have been drafted to represent the reference situation of its threatened habitats and species (Degraer et al. 2009; Regeringsbesluit 2010). For each SPA and SAC, a set of fine-tuned measures (site-specific; S-IHD) and a management plan are to be drafted.

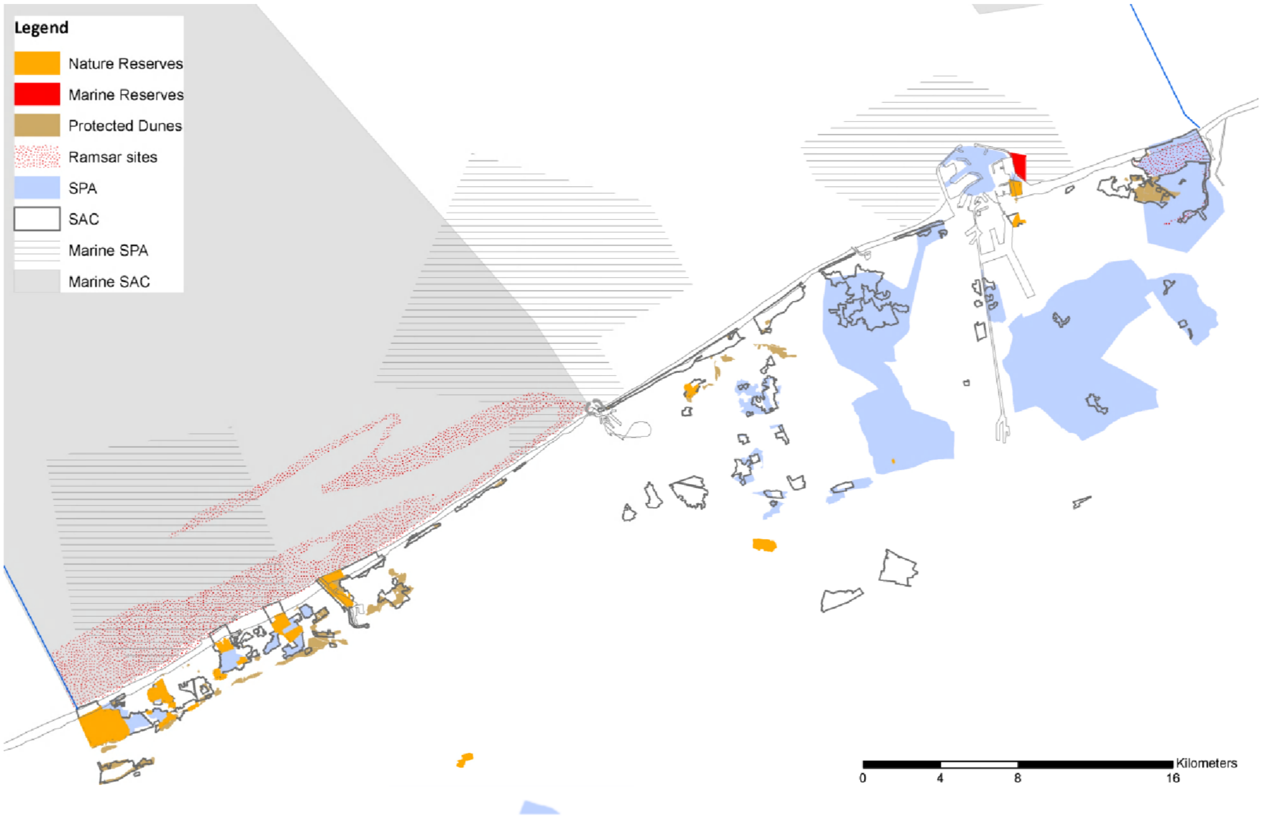


Figure 8: Protected areas along the Belgian coastline, both under federal and Flemish jurisdiction

Table 2: Overview of marine and coastal habitat types in Belgium and Flanders, according to the Habitat Directive

Habitat Directive code	**Habitat description**
1110	Sandbanks which are slightly covered by sea water all the time
1130	Estuaries
1140	Mudflats and sandflats not covered by seawater at low tide, including sandy beaches
1170	Reefs
1310	Salicornia and other annuals colonizing mud and sand
1320	Spartina swards *Spartinion maritimae*
1330	Atlantic salt meadows *Glauco-Puccinellietalia maritimae*
2110	Embryonic shifting dunes
2120	Shifting dunes along the shoreline with *Ammophila arenaria* (white dunes)
2130	Decalcified fixed dunes with fixed coastal dunes with herbaceous vegetation (grey dunes)
2150	Atlantic decalcified fixed dunes (*Calluno-Ulicetea*)
2160	Dunes with *Hippophaë rhamnoides*
2170	Dunes with *Salix repens ssp. argentea* (*Salicion arenariae*)
2180	Wooded dunes of the Atlantic, Continental and Boreal region
2190	Humid dune slacks
3140	Hard oligo-mesotrophic waters with benthic vegetation of Chara spp.

Table 3: Overview of Belgian coastal species protected under different federal and Flemish jurisdiction

MMM	Scientific name	Common name
Reptiles	*Caretta caretta*	Loggerhead sea turtle
	Chelonia midas	Green sea turtle
Fish	*Acipenser sturio*	European sea sturgeon
	Alosa alosa	Allis shad
	Alosa fallax	Twait shad
	Coregonus oxyrhynchus	/
	Lampetra fluviatilis	European river lamprey
	Petromyzon marinus	Sea lamprey
Mammals	*Halichoerus grypus*	Grey seal
	Lutra lutra	European otter
	Phoca vitulina	Harbour seal
	Phocoena phocoena	Harbour porpoise
	Tursiops truncatus	Atlantic bottlenose dolphin

Natura 2000	Scientific name	Common name
Plants	*Apium repens*	Creeping marshwort
	Liparis loeselii	/
Invertebrates	*Vertigo angustior*	Narrow-mouthed whorl snail
	Vertigo moulinsiana	Desmoulin's whorl snail
Amphibians	*Triturus cristatus*	Great Crested newt
	Epidalea calamita	Natterjack toad

Natura 2000	Scientific name	Common name
Bats	*Myotis mystacinus*	Whiskered bat
	Plecotus auritus	Brown long-eared bat
Birds	*Caprimulgus europaeus*	European nightjar
	Charadrius alexandrinus	Kentish plover
	Ciconia ciconia	White Stork
	Dendrocopos medius	Middle spotted woodpecker
	Egretta garzetta	Little egret
	Larus melanocephalus	Mediterranean Gull
	Larus minutus	Little gull
	Lullula arborea	Woodlark
	Luscinia svecica	Bluethroat
	Nycticorax nycticorax	Black-crowned night heron
	Pernis apivorus	European honey buzzard
	Platalaea leucorodia	Eurasian Spoonbill
	Podiceps cristatus	Great crested grebe
	Recurvirostra avosetta	Pied avocet
	Sterna albifrons	Little tern
	Sterna hirundo	Common tern
	Sterna sandvicensis	Sandwich tern

Regarding the WFD, only the Federal River Basin District Management Plan (RBMP) for the coastal waters falls within the scope of this PhD. Based on the existing monitoring of the OSPAR Convention, a total of six monitoring sites are regularly surveyed for hydromorphological parameters and biological (macrobenthos and phytoplankton: chlorophyll a and *Phaeocystis*) and abiotic Quality Elements (oxygen, salinity, pH and nutrients: e.g. dissolved inorganic nitrogen DIN and phosphorus DIP). No heavily modified or artificial water bodies have been designated in this RBMP. The Program of Measures lists and defines in general terms the current and future basic and supplementary measures necessary to improve the ecological and chemical status of the RBMP. Even specific supplementary measures are included for those water bodies likely to fail in the achievement of the environmental objectives by 2015. Within the scope of ICZM, Belgium has pioneered with the development and implementation of a set of 24 coastal sustainability indicators by broad public participation (Maelfait et al. 2006; Maelfait & Belpaeme 2007; Maelfait & Belpaeme 2009). The implementation of the MSFD in Belgium showed a lot of progress in 2012. Next to an initial assessment and a socio-economic analysis of the Belgian marine waters (subtidal waters, MLW to offshore), the 'Good Environmental Status (GEnvS) and defined environmental targets were described. A monitoring program (2014) and a measures program (2016) should be drafted in the near future. To meet all these EU directives, Belgium is trying to unify the regional conservation objectives and measures (G-IHDs, Natura 2000), good ecological status (WFD) and good environmental status (MSFD).

7. Aims of the PhD thesis

The overall aims and underlying research questions of this PhD study are:

(1) to investigate the *in situ* impact effects of an ecological beach nourishment on the macrobenthos of Belgian sandy beaches
  a. What is the natural spatial and temporal macrobenthic variability within the Belgian beach ecosystem and what are the main macrobenthic zonation patterns on Belgian sandy beaches?
  b. What is the relationship between relevant abiotic factors, such as beach elevation, sediment structure, total organic carbon and total organic matter, and the macrobenthos on Belgian sandy beaches?
  c. What are the *in situ* effects of ecological beach nourishment on macrobenthos?
  d. What sediment type can be recommended for use in beach nourishment projects, based on the sediment preference of the four dominant macrobenthic species (*Scolelepis squamata, Eurydice pulchra, Bathyporeia pilosa and Bathyporeia sarsi*) of the Belgian sandy beaches?

(2) to use this knowledge in order to provide protocols and tools for managing the Belgian beach ecosystem in a sustainable way, such as:
  a. a prediction model, based on the relationship between abiotic factors and the occurrence of benthos, birds, fish and shrimp.
  b. a biological valuation analysis of the intertidal and shallow subtidal Belgian coastal zone and an exploration of its applications for marine spatial planning of two space-use conflicts at the Belgian coast, being flood protection, by means of beach nourishment, and nature conservation.
  c. a series of guidelines based on scientific results from monitoring data, experiments, BVMs and model predictions.

In order to achieve these goals, following topics and aspects were investigated :

Examining the natural spatial and temporal variation on Belgian beaches
The natural variability and spatial patchiness of organisms on the sandy beach complicate the study and analyses of impacts on the macrobenthos. In order to understand both trends and disturbances and quantify possible impact effects against the natural fluctuations, long time series of abiotic and macrobenthic data of 16 Belgian beaches have been analysed, sampled over 14 years.

Unraveling possible impact effects of an ecological beach nourishment with monitoring data

Beach nourishment has rapidly become a widely applied coastal defence measure on Belgian sandy beaches, because it effectively safeguards the beach ecosystem against structural erosion when applied under certain ecological conditions. A Before After Control Impact (BACI) design has been set up to scientifically evaluate the *in situ* impact effects of the ecological beach nourishment on the soft sediment macrobenthos of the Belgian beach of Lombardsijde.

Protocols and tools for ecologically adjusted beach nourishment

Optimizing technical aspects of beach nourishment remains essential in order to minimize the impact effects on the natural ecology of the beaches. By means of experiments and model predictions, benthic responses to varying environmental conditions and different beach nourishment aspects can help in ecologically adjusting nourishment projects. To this end, a nourishment simulation model for the Belgian beach ecosystem has been created. The simulation model predicts short-term changes in beach macrobenthos species richness in response to changes in beach profile and grain size following beach nourishment and elucidates how these changes in community composition potentially feedback on the abundance of dominant species of higher trophic levels (birds, fish and shrimp). Furthermore, all available biological and ecological information for the shallow Belgian coastal zone was compiled for calculating an intrinsic biological value for several subzones of the Belgian beaches. These biological valuation maps (BVMs) can be used as reliable and meaningful baseline maps for spatial planning, marine policy and management approaches. These maps allow for the integration of ‘natural/ecological values’ at an early stage of policy development and implementation. Both model predictions and BVMs are valuable decision support tools as they represent the consequences of different management decisions in an illustrative way.

Guidelines for ecologically good practice of beach nourishment and sandy beach management

Management of sandy beaches is a multi-faceted and complex endeavor, where the interests of several stakeholders need to be combined. Based on scientific results from monitoring data, experiments, BVMs and model predictions, a series of guidelines has been provided.

8. Outline of the PhD thesis

Apart from the general introduction and discussion, this thesis is a compilation of research articles, either published or to be submitted for publication. Each chapter is therefore intended to be an autonomous part, which can be read separately from the other chapters. Inevitably, there is some overlap between the introduction and discussion sections of the different chapters but given the variety of topics, this overlap should not hinder the general readability of this PhD thesis. Cited literature is compiled in a single list at the end of the thesis. Chapter 4 has been carried out and written in close co-authorship with the first author Joke Van Tomme. Chapter 5 has a shared first authorship with Joke Van Tomme. All other chapters have the candidate as first author.

Chapter 1 (general introduction) gives an overview of the sandy beach ecosystem, illustrated by the focus habitat of this PhD, namely the Belgian beaches. Its ecosystem components, food web and threats are thoroughly discussed with a prime focus on coastal defence structures and their impact on the beach ecosystem. The current status of the governance in the Belgian coastal zone is documented to provide for a better understanding of beach spatial management in Belgium. The study described in **Chapter 2, "Assessment of the ecological characteristics of the Belgian beaches prior to the implementation of the Belgian Master Plan for Coastal Safety",** tries to describe and update our knowledge of the intertidal and shallow subtidal Belgian coastal areas on both spatial and temporal scales. For this study, data from 1997 to 2011 were analysed, encompassing 16 intertidal and 10 shallow subtidal coastal zones, sampled over 8 years in 3 different seasons. The main aims were to partition the macrobenthic variance and describe both the relationship between five abiotic factors and the macrobenthos and the macrobenthic zonation patterns. In **Chapter 3 "The monitoring of 'ecological' beach nourishment on macrobenthos, within a Special Area of Conservation (SAC) along the Belgian coast"**, a Before After Control Impact (BACI) design was set up to scientifically evaluate the possible impact effects of the ecological beach nourishment on the soft sediment macrobenthos of the Belgian beach of Lombardsijde. The information in Chapter 2 is used as a broad reference scale against which possible impact effects can be measured. Beach nourishment typically alters the sediment grain size and beach profile of the nourished beach. **Chapter 4**, titled **"Macrofaunal sediment selectivity considerations for beach nourishment programmes"**, examines the sediment preferences of Belgian sandy beach macrofauna both in single-species and combined-species conditions. This information can help in adjusting the technical beach nourishment aspects to minimize ecological impact.

Since beach nourishment has become generally applicable on Belgian beaches, and an ecosystem based management is indispensable, information on the response of the complete sandy beach ecosystem to the altered physical environment is needed. Therefore a model was developed in **Chapter 5, "Assessing the impact of beach nourishment on the intertidal food web through the development of a mechanistic-envelope model"**, predicting responses of all ecosystem components after nourishment using both the available knowledge and knowledge obtained in this PhD study. As different scenarios can be tested in this model, optimizing various technical aspects of beach nourishment will be one of the model's main advantages. In **Chapter 6, "Marine biological valuation of the shallow Belgian coastal zone: a space-use conflict example within the context of marine spatial planning"**, we used the marine biological valuation method in order to assess the marine biological value of the shallow Belgian coastal zone for the support of ecosystem-based marine spatial planning. The resulting biological valuation maps were then used to explore the applications of BV on two space-use conflicts at the Belgian coast, mainly flood protection, by means of beach nourishment, and nature conservation. **Chapter 7, "An ecosystem approach towards Belgian coastal policy",** is a general discussion. The conservation goals of sandy beaches are given from a biologist's perspective and translated towards beach nourishment recommendations and policy guidelines for an ecosystem-based, integrated sandy beach management.

Chapter 2: Assessment of the ecological characteristics of the Belgian beaches prior to the implementation of the Belgian Master Plan for Coastal Safety

Sarah Vanden Eede, Dries Bonte, Steven Degraer, Magda Vincx (in prep.) Assessment of the ecological characteristics of the Belgian beaches prior to the implementation of the Belgian Master Plan for Coastal Safety.

In preparation for submission

Abstract

Sandy shores or beaches line 70 % of the world's oceans, including the entire Belgian coastline (67 km). They have a multitude of ecosystem functions, constitute an important habitat for a variety of fauna and flora and hold important economic, social and cultural value as prime recreational assets. In order to protect the Belgian coastline against erosion and coastal flooding on a short and long term basis (up to 2050), the Belgian sandy beaches face a multitude of beach nourishment activities over the next years, as stipulated in the Belgian Master Plan for Coastal Safety. Monitoring the ecological characteristics of the Belgian beaches will aid in the detection of possible impact effects of past, current and future beach nourishment activities.

To this end, 686 intertidal and 582 shallow subtidal samples from the period 1997 – 2011 were analysed to describe the macrobenthic communities and corresponding abiotic patterns on 16 Belgian beaches. The 15 intertidal and 9 shallow subtidal locations were treated separately. The partitioning of macrobenthic community structure within the Belgian beach ecosystem showed a large within beach variability, linked to elevation on the beach (intertidal: 44 % and shallow subtidal: 50 %) and median grain size of the sediment (intertidal: 35 % and shallow subtidal: 23 %), in both the intertidal and shallow subtidal zone. Large scale along-shore spatial (intertidal: 14 % and shallow subtidal: 13 %) and long-term temporal (intertidal: 5% and shallow subtidal: 12 %) variability clearly explained less variation. Several spatial and temporal trends in abiotic factors (overall median grain size between 150 and 300 µm) and in macrobenthic species richness (intertidal: 0 – 19 species; shallow subtidal: 0 – 28 species), abundance (intertidal: 0 – 3988.75 individuals.m^{-2}; shallow subtidal: 0 – 1949.32 individuals.m^{-2}) and biomass (intertidal: 0 – 6.95 g AFDW.m^{-2}; shallow subtidal: 0 – 246.14 g AFDW.m^{-2}) were measured. The mean macrobenthic abundance in the intertidal and shallow subtidal zone fluctuates between 0 and 350 individuals.m^{-2} over the years. Furthermore, the realized niches of the dominant macrobenthic species of the Belgian beaches were defined as the area where these species really live during low tide, characterized by elevation on the beach and median grain size of the sediment.

Since all sampling locations were considered to be outside the influence of major impacting activities, these findings improve our knowledge of the natural abiotic and macrobenthic variability of the Belgian beaches. As such, this study can be used as a preconceived basis (t0 situation) of 'natural' macrobenthic variability on the Belgian beaches.

Keywords: natural variation, mesoscale patterns, sandy beaches, macrobenthos

1. Introduction

Sandy shores, also called beaches, cover 70 % of all continental margins (McLachlan & Brown 2006). They have a multitude of ecosystem functions as they are an important habitat for a variety of fauna and flora, and are concurrently of immense economic, social and cultural importance to humans as prime recreational assets. Moreover, natural sandy beaches function as a buffer between land and sea, thus protecting the hinterland from scour, inundation and wave erosion (Young & Bryant 1992; Defeo et al. 2009).

The Belgian coastline is 67 km long and entirely composed of sandy beaches. However, the ecological continuum naturally expected on this type of ecosystem, from the intertidal zone to the foredunes, is disrupted by stone breakwaters and concrete dykes (De Ruyck et al. 2001). These constructions were built as a response to erosion and coastal flood risk (Speybroeck et al. 2006a; Roode et al. 2008). In order to protect the Belgian coast (up to 2050) a multitude of beach nourishment projects are planned over the next years (Mertens et al. 2008). Monitoring the ecological characteristics of the Belgian beaches will aid in the detection of possible impact effects of past, current and future beach nourishment projects.

Belgian beaches are wide and (ultra)dissipative (Speybroeck et al. 2008a), displaying a semi-diurnal macrotidal regime (De Moor 1979, 1986; De Moor & Bloome 1988; De Moor 2006). The sandy sediment has an across-shore average grain size ranging from 160 µm to 380 µm (Speybroeck et al. 2008a). It becomes coarser from west to east and from the subtidal zone up to the supralittoral dry beach. A natural gradient of slightly increasing beach slopes and consequently decreasing beach width also exists from west to east (Depuydt 1972). Sandy beaches have across-shore, defined as perpendicular to the water line, and along-shore, defined as parallel to the water line, dimensions (James & Fairweather 1996), giving rise to small-scale morphodynamic and associated macrobenthic gradients (Degraer et al. 2003b).

Macrobenthos is generally defined as the organisms measuring over 1 mm long and living buried in the seabed. This group of bivalves, polychaetes, crustaceans and echinoderms performs well as an indicator of pollution and stress and plays a key role in the beach ecosystem and foodweb. They feed on detritus, algae and/or other benthic animals (Sundbäck & Persson 1981; Josefson et al. 2002). In turn, they are a major food source for birds and epibenthos (crabs, shrimp, juvenile fish). Owing to the highly dynamic nature of their environment, mainly determined by waves, tides and winds, macrobenthic species have a high tolerance towards several forms of environmental stress. Some key adaptations to their habitat are mobility, burrowing ability, rhythmic (e.g. tidal, circadian, (semi)lunar, seasonal) behavior, orientation mechanisms and flexibility to cope with rapidly changing conditions (McLachlan & Jaramillo 1995; Defeo et al. 2009).

The distribution of macrobenthos on sandy beaches has been well documented in many parts of the world (e.g., (Dexter 1979; Knott et al. 1983; Jaramillo & McLachlan 1993; McLachlan & Jaramillo

1995; Fernandes & Soares-Gomes 2006; Veloso et al. 2006), including the Belgian sandy beaches (Degraer et al. 1999a; Degraer et al. 2003b; Speybroeck et al. 2008a). Along the Belgian across-shore gradient, variability can be detected within the macrobenthic spatial distribution (figure 1), from sparse high intertidal to somewhat diverser shallow subtidal communities (Speybroeck 2007). In general, a very narrow high intertidal zone is the habitat of the amphipod *Bathyporeia pilosa* and the polychaete *Scolelepis squamata*. The isopod *Eurydice pulchra* and the amphipod *Bathyporeia sarsi* live in a wide zone in the middle of the intertidal area. In the lowest parts of the intertidal zone, several polychaetes, e.g. *Nephtys cirrosa*, and bivalves have to share the space. The shallow subtidal zone has never been investigated in detail before, but different subtidal communities can be distinguished from the French to the Dutch border (Degraer et al. 2003b; Van Hoey et al. 2004). The dominant subtidal species are the bivalves *Abra alba*, *Macoma balthica* and *Kurtiella bidentata*, the polychaetes *Cirratulidae* species, *Glycera lapidum*, *Magelona* species, *Nephtys cirrosa*, *Ophelia borealis* and *Spiophanes bombyx*, and the amphipod *Urothoe brevicornis*.

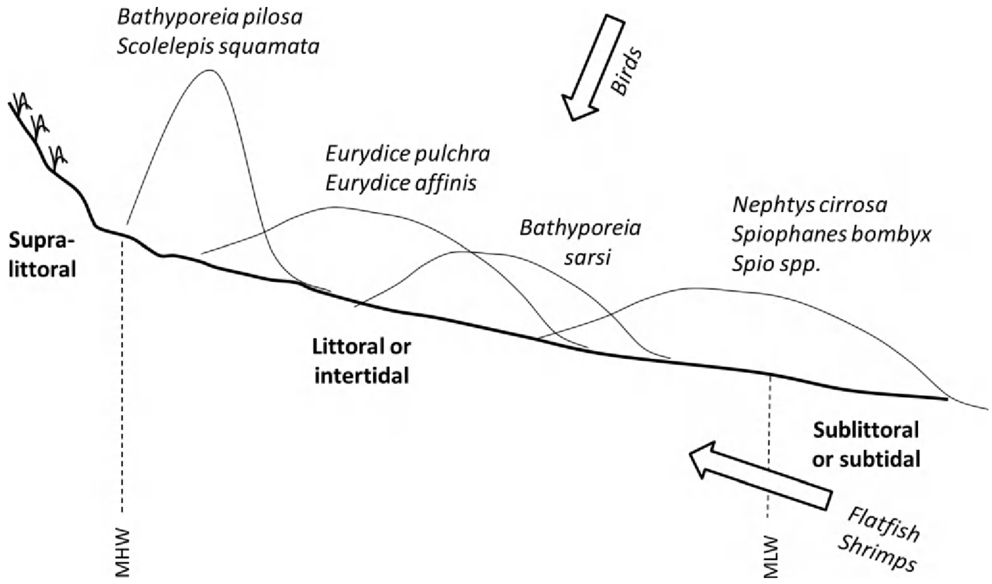


Figure 1: Zonation pattern of different Belgian macrobenthos species (adapted from (Van Tomme 2013)); MHW: mean high water level; MLW: mean low water level

Due to temporal variations within the zonation patterns (Haynes & Quinn 1995) and morphodynamic differences among beaches (McLachlan & Jaramillo 1995), attempts to apply universal zonation schemes in the intertidal and shallow subtidal beach zones (Dahl 1952; Salvat 1964) have often failed. Intertidal sand is a fundamentally unstable habitat for infauna as it is subjected to both regular small-scale disturbance and irregular, often frequent, large-scale disturbance resulting from storms. It also suffers from rapid fluctuations in physical and chemical characteristics. Subtidal sandy deposits are equally unstable but not subjected to the additional stress of twice daily inundation and exposure by the tides (Hayward 1994). In the end, the species composition, richness, abundance (McLachlan & Jaramillo 1995; Veloso & Cardoso 2001) and biomass (Adriaanse & Coosen 1991; McLachlan et al.

1996a; McLachlan & Dorvlo 2005) always feedback with the abiotic environment. Although currents, salinity and temperature may be basic steering variables (Govaere et al. 1980; Creutzberg et al. 1984), the most important morphodynamic beach characteristics on Belgian beaches are grain size, slope or beach elevation and total organic matter (Degraer et al. 2003b; Speybroeck et al. 2006a). This natural variability and spatial patchiness of macrobenthic organisms complicates the study and quantification of possible impact effects of, for instance, beach nourishment activities.

Previously, the macrobenthos of Belgian beaches has been surveyed in detail at the ultra-dissipative beach of De Panne (Elliott et al. 1997; Degraer et al. 1999a) and on eleven intertidal beaches that were regarded as being pristine (Speybroeck et al. 2005b; Speybroeck et al. 2008a). The general aim of this study is to update our knowledge of the Belgian beach ecosystem in both the intertidal and shallow subtidal zone. An extensive list of longitudinal Belgian macrobenthic data is used to investigate (1) the natural spatial and temporal macrobenthic variability within the Belgian beach ecosystem, (2) the relationship between relevant abiotic factors, such as beach elevation, sediment structure, total organic carbon and total organic matter, and the macrobenthos on Belgian sandy beaches and (3) the observed niche and interpolated occurrence of the dominant macrobenthic species on Belgian sandy beaches.

2. Material and methods

2.1 Study Area

The subdivision of the shallow Belgian coastal zone follows the ecological zonation, focusing specifically on the intertidal and the shallow subtidal zones (figure 2). It is therefore defined by a landward boundary that follows the high water mark obtained by airborne Light Detection And Ranging (LIDAR) observations of the Belgian coast in 2011 (data provided by the Agency for Maritime and Coastal Services: Coastal division – MDK(Deronde et al. 2008) and a seaward boundary for the shallow subtidal foreshore of 1 nautical mile from the zero depth (0 m) bathymetric line (figure 2).

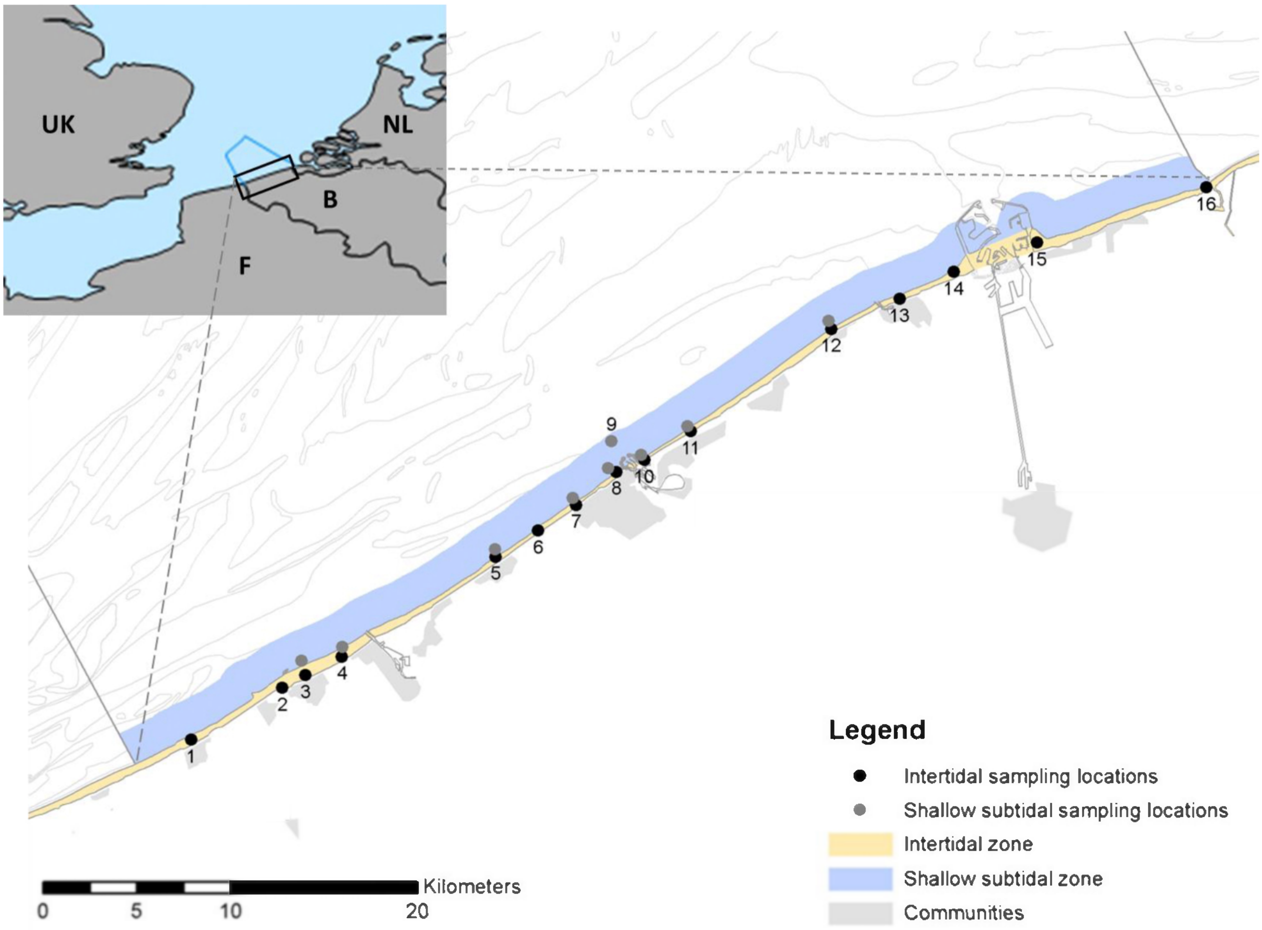


Figure 2: The Belgian coastal zone, with a distinction between the intertidal (light brown) and shallow subtidal zone (blue), showing the sixteen sampling locations (code specifications in table 1)

2.2 Data

Since 1997, the Marine Biology Research Group of Ghent University coordinated numerous macrobenthic studies along the Belgian coastline. An integrated database comprises all available relevant data gathered during the period 1997 – 2011 in the intertidal and shallow subtidal zones (table 1). Sixteen intertidal locations were investigated, together with 9 nearby shallow subtidal locations. Intertidal and shallow subtidal samples were also taken at the beach of Lombardsijde. As these samples were taken to monitor an actual nourishment (performed in 2009), they will be analysed in the next chapter.

Table 1: Sampling locations used for the integrated macrobenthic database. Only data collected in intertidal and shallow subtidal zones of the Belgian coast (key: S=spring; A=autumn; a (Degraer et al. 2003b); b (Speybroeck et al. 2003); c (Welvaert 2005); d (Van Ginderdeuren et al. 2007); e (Vanden Eede et al. 2008); f (Vanden Eede & Vincx 2010); g (Vanden Eede & Vincx 2011b); h (Vanden Eede et al. 2013))

INTERTIDAL ZONE		**1997**	**2002**	**2004**	**2006**	**2008**	**2009**	**2009**	**2010**	**2010**	**2011**	**2011**
		A	**A**		**A**	**S**	**S**	**A**	**S**	**A**	**S**	**A**
Study	Codes	a	b	c	d	e	f	f	g	g	h	H
De Panne	1	28										
Schipgatduinen	2	22										
Koksijde-Oostduinkerke	3			15	26							
Nieuwpoort	4					26	26	26	15	15	15	15
Middelkerke	5		18	15	26							
Raversijde	6	20										
Mariakerke	7											15
Oostende-Center	8		33	45	25							
Oostende-East	10			25	25				15	15	15	15
Bredene	11								15	15	15	15
Wenduine	12			15	26							
Blankenberge	13											15
Fonteintjes	14	22										
Heist	15	22										
Zwinduinen en Polders	16	20										
Total amount of samples		**134**	**51**	**115**	**128**	**26**	**26**	**26**	**45**	**45**	**45**	**75**

SHALLOW SUBTIDAL ZONE		**1997**	**2002**	**2004**	**2006**	**2008**	**2009**	**2009**	**2010**	**2010**	**2011**	**2011**
		A	**A**		**A**	**S**	**S**	**A**	**S**	**A**	**S**	**A**
Study	Codes	a	b	c	d	e	f	f	g	g	h	H
Koksijde-Oostduinkerke	3			15	25							
Nieuwpoort	4					25	25	25	15	15	15	15
Middelkerke	5		18	15	25							
Mariakerke	7											15
Oostende-Center	8		42	45	25							
Oostende-Fairway	9			15	12							
Oostende-East	10			25	25				15	15	15	15
Bredene	11								15	15	15	15
Wenduine	12			15	25							
Total amount of samples		**0**	**60**	**130**	**137**	**25**	**25**	**25**	**45**	**45**	**45**	**60**

2.3 Sampling method

Intertidal sampling always started at high tide and followed the receding water down the beach, as the organisms tend to cluster very close to the water line, ending at low tide. Samples were taken by excavating a quadrat frame (surface area, 0.1026 m^2) to a depth of 0.15 m. The samples were immediately sieved (ø 1 mm) and preserved in an 8 % formaldehyde-seawater solution. Next to each biotic sample, one core sample (ø 3.6 cm) for sediment analysis was collected. At every sampling location, a beach profile and the geographic position (geographic wgs84) were noted. As we always sample at the water line, we can deduce the real elevation of the sample locations to the water surface (MLW, mean low water level) using the M2 reduction model (Van Cauwenberghe et al. 1993). The shallow subtidal zone was sampled with a small vessel at high tide. At every sample location, a Van Veen grab was lowered to take a sediment sample (surface area, 0.1026 m^2). Simultaneously, the coordinates (UTMwgs84), time and depth of each sample were determined. The depth was afterwards corrected (compared to MLW) using the M2 reduction model. Before opening the Van Veen grab, a core sample (ø 3.6 cm) for sediment analysis was taken. The samples were immediately sieved (ø 1 mm) and preserved in an 8 % formaldehyde-seawater solution.

2.4 Laboratory procedures

The sieved samples for faunal analysis were stained with Rose Bengal and elutriated ten times to collect the macrobenthos. The remaining material was examined to collect the larger organisms that were too heavy to be floated off by elutriation. All macrobenthic organisms, except for Oligochaeta, Actiniaria and Nematoda, were identified to species level, where possible, and counted. Faunal abundance was extrapolated to the number of individuals per m^2. Biomass (gram Ash Free Dry Weight or g AFDW) estimates were obtained by loss of mass on ignition (480 °C for 2 h) of oven-dried samples (110 °C for 24 h) (Van Ginderdeuren et al. 2007; Vanden Eede et al. 2008; Vanden Eede & Vincx 2010, 2011b; Vanden Eede et al. 2013). The biomass was calculated on higher taxon level except for the dominant species, being *Bathyporeia pilosa and Bathyporeia sarsi*, *Ensis* juveniles, *Eurydice* species, *Lanice conchilega, Macoma balthica* juveniles, *Nephtys cirrosa*, *Owenia fusiformis, Scolelepis squamata* and *Spisula subtruncata*.

After drying the sediment samples, the grain size distribution was determined with a Malvern Mastersizer 2000G laser with diffraction module (measuring range: 0.02 – 2000 µm). In this paper, sediment was characterized by median grain size (µm) and silt fraction (< 63 µm). The percentage of shell fragments was determined by means of the percent volume of sediment remaining on the 1 mm sieve (carbonate content). The values for Total Organic Carbon (TOC) were calculated with an automatic element analyzer 1500 Carlo Elba. The percentage of Total Organic Matter (TOM) was obtained by loss of mass on ignition, using the biomass analysis strategy.

2.5 Data analysis

In total, 721 intertidal and 582 shallow subtidal samples were gathered between 1997 and 2011. 35 intertidal samples were excluded from the analyses as they contained no macrobenthos. In total, 207 species were identified from which 89 species were taken into account (79 species in the intertidal zone and 86 species in the shallow subtidal zone). The species excluded from these analyses clearly attributed to other benthic groups, e.g. meiobenthos, hyperbenthos or epibenthos, and/or were gathered in a non-representative way, e.g. a frequency of occurrence of less than 0.5 % and a maximum of 5 individuals per sample (Appendices – Chapter 2 – table A). The sampling location maps were created with ESRI ArcMap Version 9.3. Multivariate analyses in Primer v6 were used to detect patterns in the distribution of macrobenthic assemblages, without transforming data. Multidimensional scaling was based on Bray-Curtis similarity, a statistic used to quantify the compositional dissimilarity (abundance data per species) between different sampling locations.

Samples within a cluster are more similar than samples of different clusters. To partition the abiotic and macrobenthic variance on Belgian beaches, the adonis function of the Vegan package in R 2.14.2 was used. It fits permutative linear models (e.g., factors, polynomial regression) to Euclidean (abiotic data) and Bray-Curtis (macrobenthic data) distance matrices and allows for nonparametric analysis of variance using these distance matrices. The Spearman's rank correlation coefficient was calculated in R 2.14.2 to assess the relationship between median grain size (µm) and elevation (m versus MLW). Based on these two abiotic variables, the realized niche of the dominant intertidal and shallow subtidal species of the Belgian beaches could be illustrated by contour plots (R2.14.2, with the akima and lattice packages). All tables and basic graphs were made in Microsoft Excel 2010.

3. Results

Minimum, maximum and mean intertidal and shallow subtidal values for the measured abiotic and biotic factors are given in table 2. Between these minimum and maximum values the natural variation on Belgian beaches runs its course.

Table 2: Mean, minimum and maximum values for abiotic and biotic factors in the intertidal and shallow subtidal zone (MLW: mean low water level; AFDW: ash-free dry weight)

	Beach elevation (m versus MLW)	**Median grain size (µm)**	**Silt fraction (%)**	**Total organic matter (%)**	**Total organic carbon (%)**	**Carbonate content (%)**	**Species richness (number of species)**	**Abundance (number of individuals.m^{-2})**	**Biomass (g AFDW.m^{-2})**
INTERTIDAL									
mean	2.32	241.55	0.02	0.39	0.47	8.71	4.42	104.62	0.34
minimum	0.00	175.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
maximum	5.41	464.00	2.25	1.84	19.00	36.68	19.00	3988.75	6.95
SHALLOW SUBTIDAL									
mean	-3.70	181.44	5.47	0.93	0.63	10.09	8.08	107.57	7.85
minimum	-10.00	17.45	0.00	0.00	0.00	0.00	0.00	0.00	0.00
maximum	-2.67	319.73	89.30	11.75	4.85	36.10	28.00	1949.32	246.14

Multidimensional scaling (MDS) of all sampling data did not indicate a clear separation of the macrobenthic data based on spatial (different beaches) or temporal (different years or seasons) variables. There is an overlap visible between samples from the intertidal and shallow subtidal zone (Figure 3). However, since these two habitats are substantially different to harbor other macrobenthic communities, as described in Chapter 1 and in Van Hoey et al. (2004), the intertidal and shallow subtidal data were treated separately.

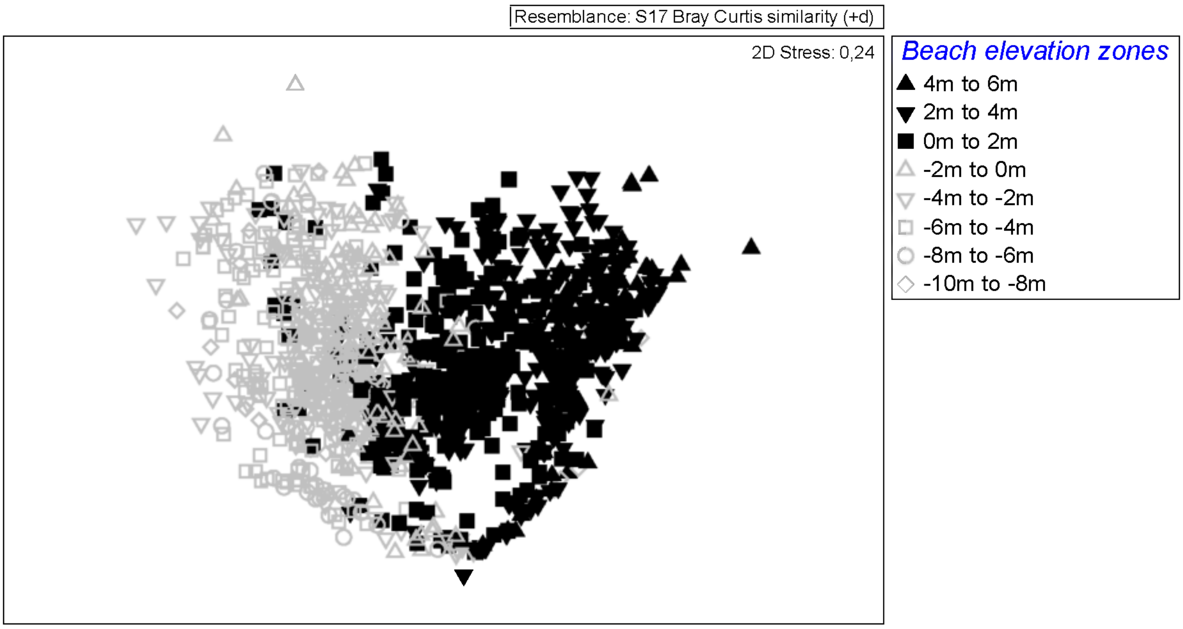


Figure 3: MDS plot of all data, showing a partitioning into two groups, being intertidal (black) and shallow subtidal (grey) samples with overlap around mean low water level, and to a lesser extent into beach elevation zones

3.1 Partitioning of abiotic and macrobenthic variance on Belgian beaches

The partitioning of abiotic variance (Appendices – Chapter 2 – tabel B1) is based on six abiotic factors, being beach elevation (m versus MLW), median grain size (µm), silt fraction (%), carbonate content (%), total organic carbon (%) and total organic matter (%). Figure 4 shows an apparent quantitative difference between beaches (intertidal: 57 % and shallow subtidal: 45 %) and between years (intertidal: 22 % and shallow subtidal: 28 %), rather than between seasons (intertidal: 0.5% and shallow subtidal: 1 %).

The partitioning of macrobenthic variance (Appendices – Chapter 2 – tabel B2) is based on the macrobenthic abundance data and all six abiotic factors (figure 4). Large scale spatial (intertidal: 14% and shallow subtidal: 13 %) and temporal (intertidal: 5 % and shallow subtidal: 12 %) variability explains less variation in community structure than within beach elevation (intertidal: 44 % and shallow subtidal: 50 %) and median grain size (intertidal: 35 % and shallow subtidal: 23 %). There appears to be a large within beach variability in macrobenthic abundance, linked to elevation on the beach and median grain size of the sediment.

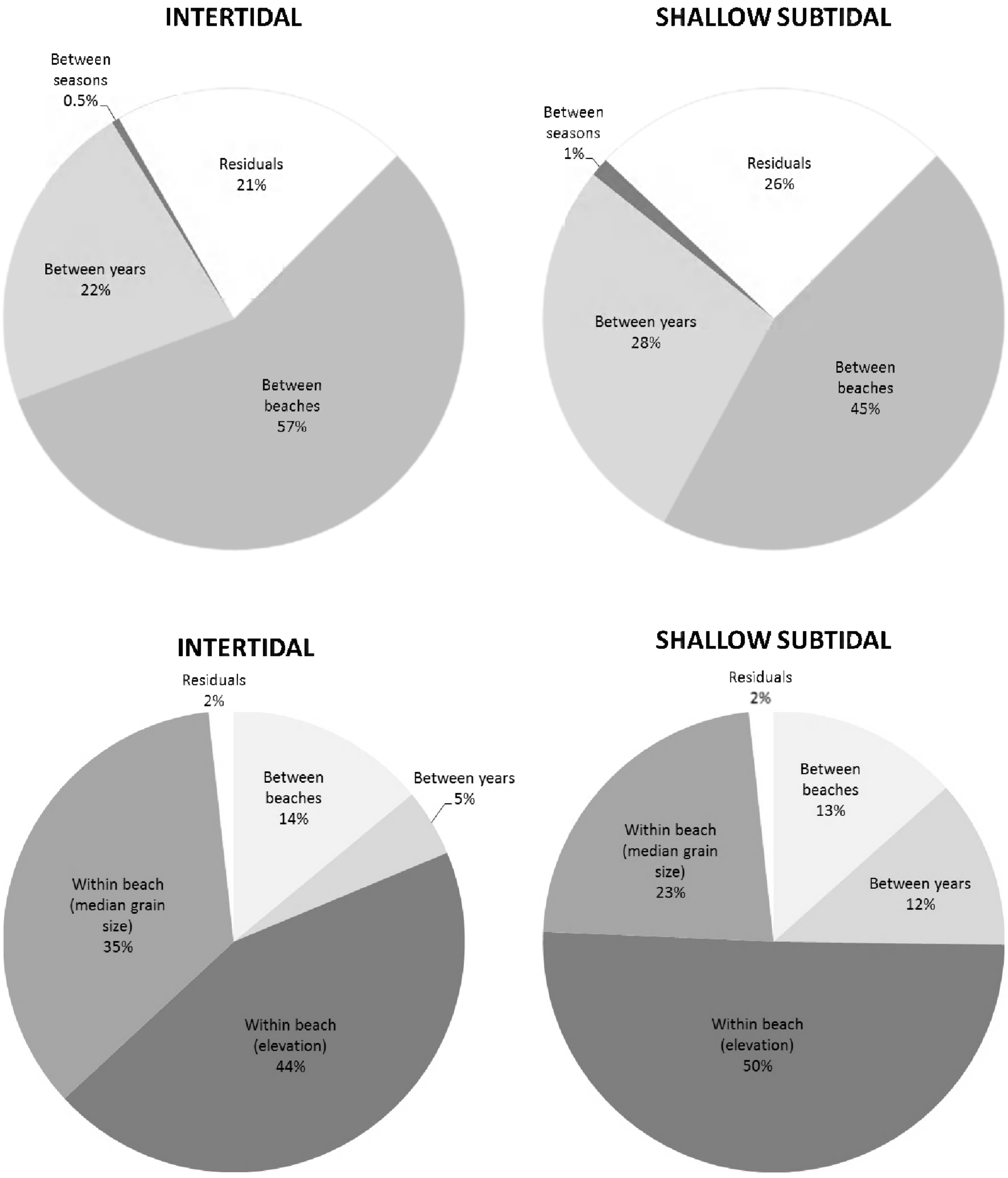


Figure 4: Partitioning of abiotic (top) and macrobenthic variance (bottom) in the intertidal and shallow subtidal zone along the Belgian coastal area, based on data from 1997 – 2011, representation of the R^2 values (as percentages) given in Appendices – Chapter 2 – Table B1 and B2

3.2 Spatial and temporal macrobenthos variation

The variation between beaches and the variation between years has been analyzed in great detail and the tables in Appendices – Chapter 2 – table C and D give an overview of the mean abiotic and biotic values, respectively per beach over all the sampled years and per year over all the sampled beaches. The two most important abiotic characteristics according to figure 4, e.g. grain size (µm)a dn elevation on the beach (m versus MLW), and TOM (%) are analyzed per beach per year (figure 5), next to the three biotic factors, e.g. species richness (number of species), abundance (number of indviduals.m^{-2}) and biomass (g AFDW.m^{-2}) (figure 6). The mean intertidal median grain size coarsens when going from west (200 – 250 µm) to east (250 – 330 µm). A peak mean value of 420.54 µm was only registered once in 2006 on the beach of Oostende-Center. The shallow subtidal zone holds much finer sediment (120 – 230 µm) which appears to become even finer when going from west to east. The intertidal mean beach elevation values vary between 1.5 and 3 m relative to the MLW. The mean depth of the shallow subtidal samples shows greater variation (-1 and -9 m) with the deepest values recorded in 2006. The shallow subtidal zone also holds much higher mean TOM values (0 - 3.5 %) than the intertidal zone (0 – 0.8 %). In 2004, the lowest mean TOM values were recorded in both the intertidal and shallow subtidal zone. Figure 7 shows that the mean species richness is higher in the shallow subtidal (2 – 16 species) than in the intertidal zone (2 – 9 species). The highest mean species richness was recorded in 2011 in the shallow subtidal zone of Nieuwpoort (15 species). The intertidal mean abundance fluctuates between 20 and 200 individuals.m^{-2} with a peak mean abundance value of 456 individuals.m^{-2} recorded in De Panne in 1997. Mean shallow subtidal abundance varies between 120 and 230 individuals.m^{-2}. The mean biomass in the intertidal zone (0 – 0.8 g.m^{-2}) is significantly lower than in the shallow subtidal zone (0 – 45 g.m^{-2}). One mean intertidal peak value can be seen in 2004 in Wenduine (1.7 g.m^{-2}). In the shallow subtidal zone, the mean 2010 biomass value of Nieuwpoort (33.1 g.m^{-2}) and the mean 2011 biomass value of Mariakerke (45.2 g.m^{-2}) are the only outliers.

When looking at the evolution of median grain size in time (figure 7), the intertidal values (200 – 280 µm) are always higher than the shallow subtidal values (170 – 210 µm). In 2006, the highest mean median grain size could be detected in the intertidal zone (277.39 ± 3.27 µm; without Oostende-Center: 242.64 ± 1.49 µm). The mean shallow subtidal median grain size peaks in 2010 (205.79 ± 1.31 µm). Figure 8 also shows the temporal evolution of the macrobenthic abundance. Two intertidal peak values (1997: 193.07 ± 16.54 individuals.m^{-2} – 2008: 178.38 ± 10.05 individuals.m^{-2}) and one shallow subtidal peak value (2009: 312.04 ± 17.05 individuals.m^{-2}) can be seen. In 2002, 2006 and 2008, the mean intertidal abundance was higher than the shallow subtidal values.

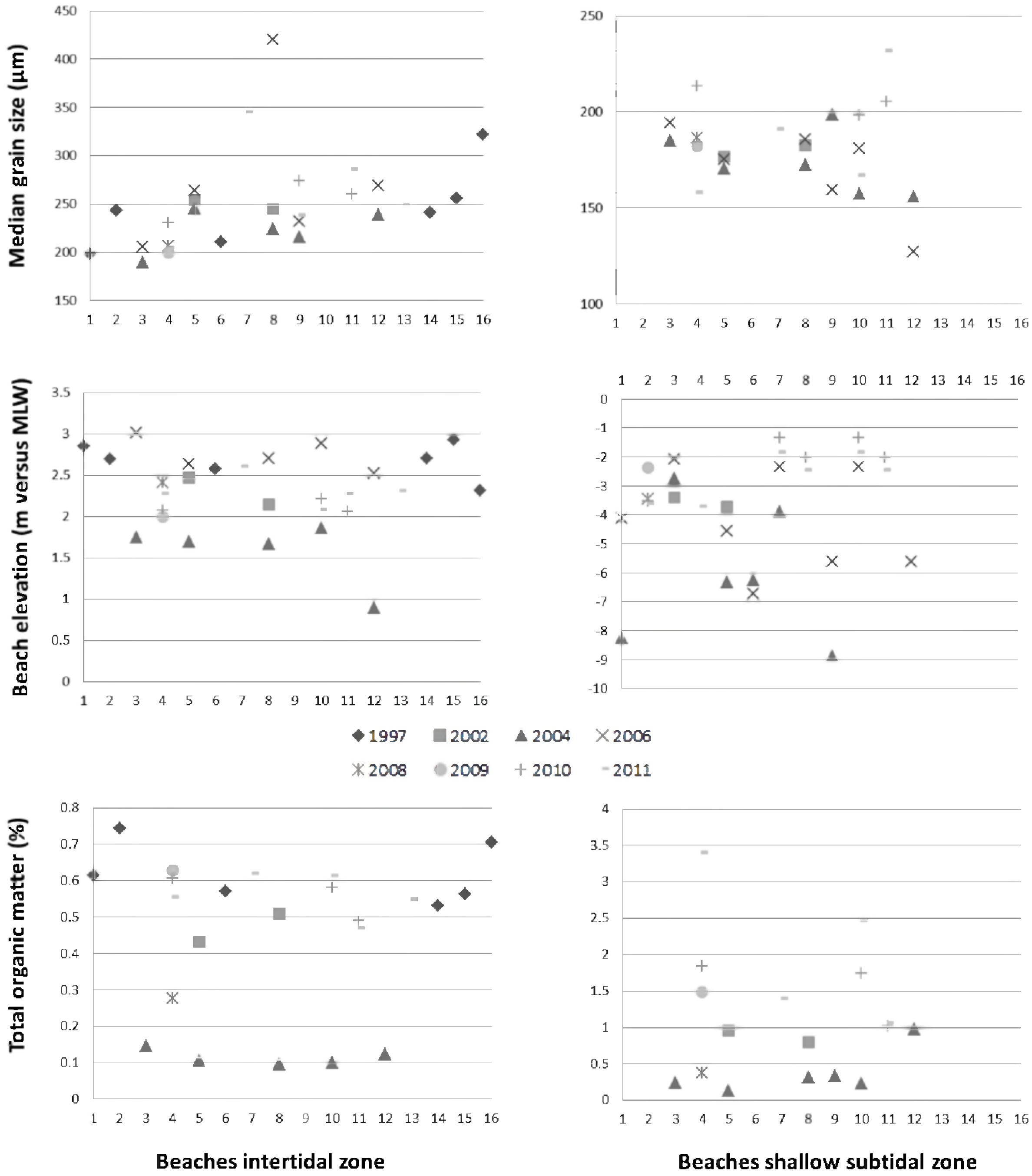


Figure 5: Mean median grain size (µm), beach elevation (m versus MLW) and total organic matter (%) per beach and per year, for the intertidal (left) and shallow subtidal zone (right) separately; with codes in the X-axis representing the following beaches: (1) De Panne, (2) Schipgatduinen, (3) Koksijde-Oostduinkerke, (4) Nieuwpoort, (5) Middelkerke, (6) Raversijde, (7) Mariakerke, (8) Oostende-Fairway, (9) Oostende-East, (10) Bredene, (12) Wenduine, (13) Blankenberge, (14) Fonteintjes, (15) Heist, (16) Zwinduinen en Polders

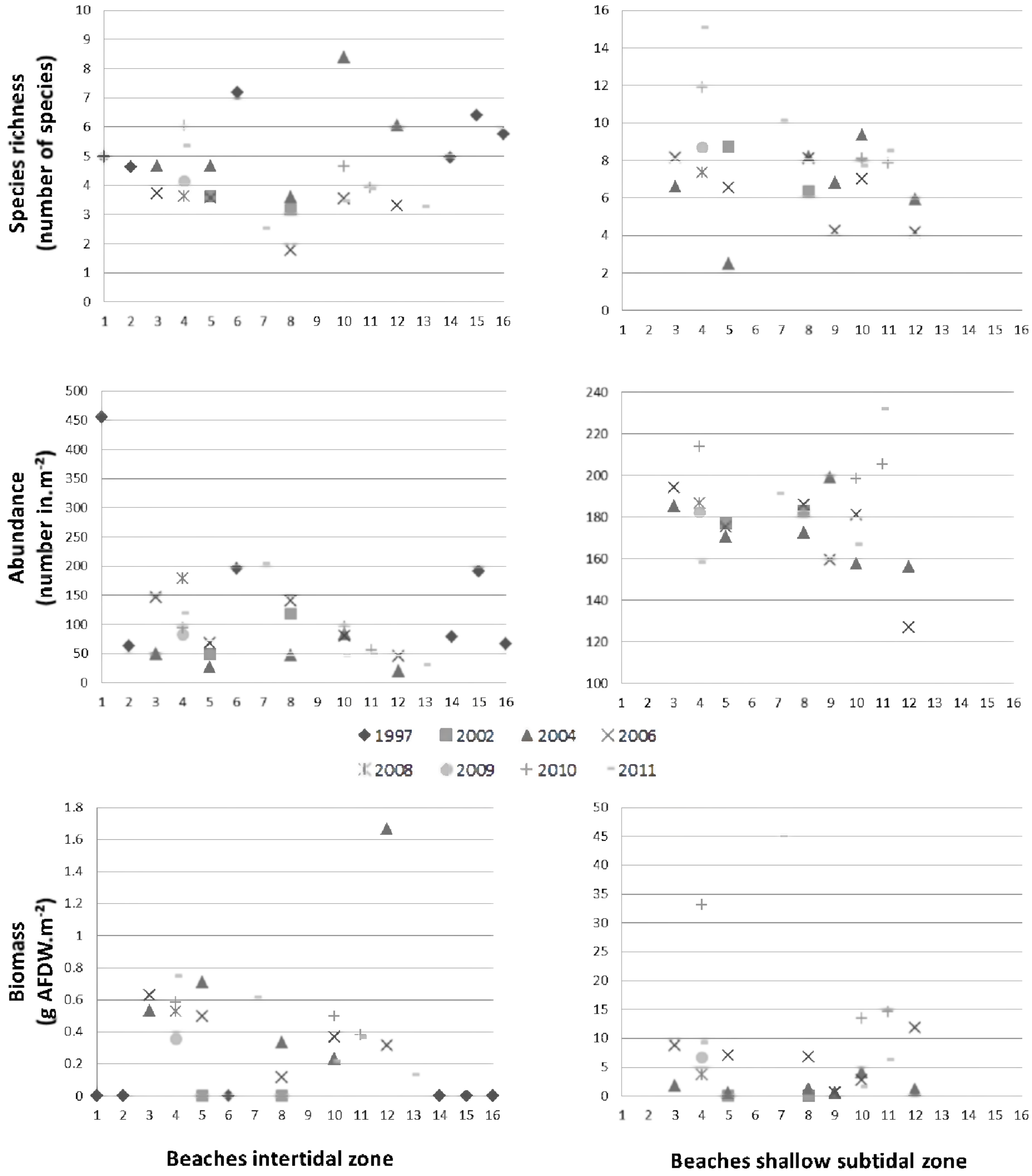


Figure 6: Mean species richness (number of species), abundance (number of individuals.m^{-2}) and biomass (g AFDW.m^{-2}) per beach and per year, for the intertidal (left) and shallow subtidal zone (right) separately; with codes in the X-axis representing the following beaches: (1) De Panne, (2) Schipgatduinen, (3) Koksijde-Oostduinkerke, (4) Nieuwpoort, (5) Middelkerke, (6) Raversijde, (7) Mariakerke, (8) Oostende-Fairway, (9) Oostende-East, (10) Bredene, (12) Wenduine, (13) Blankenberge, (14) Fonteintjes, (15) Heist, (16) Zwinduinen en Polders

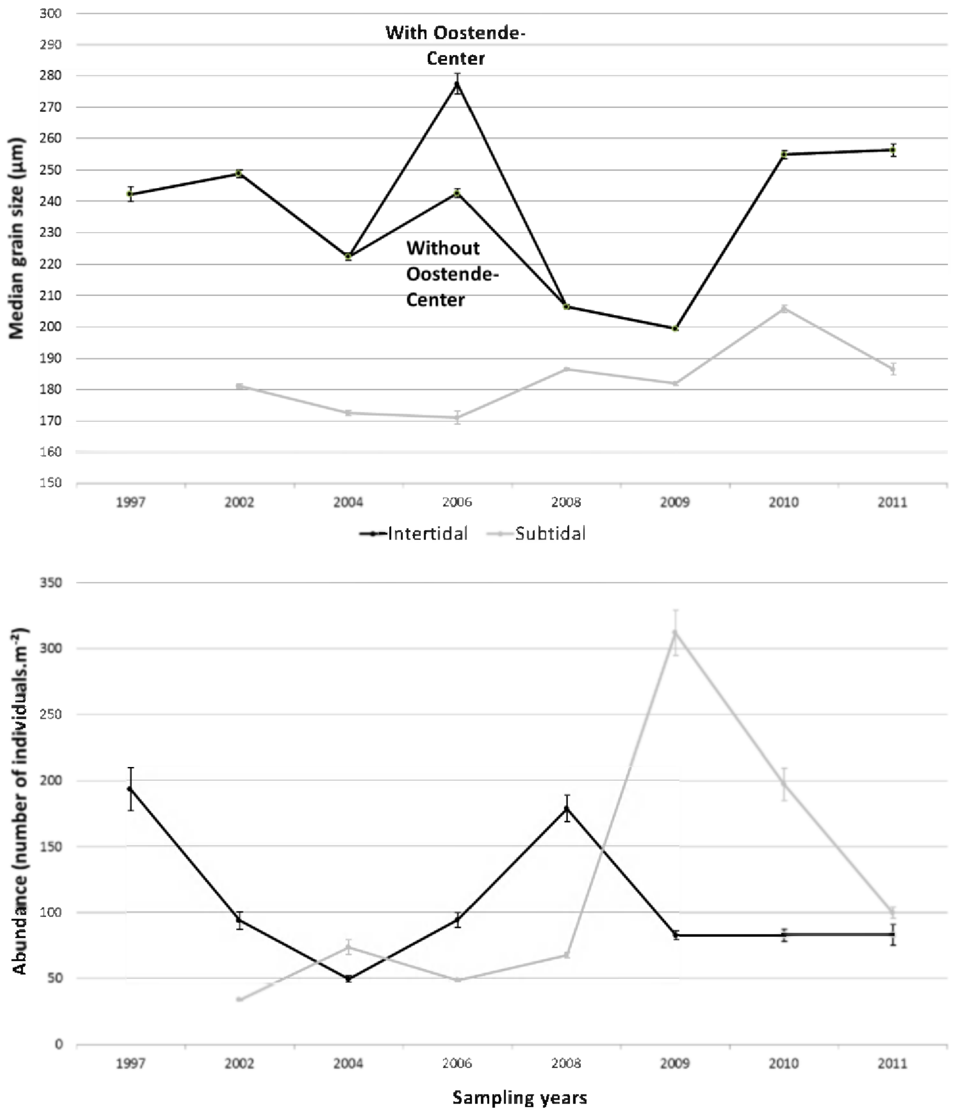


Figure 7: Mean median grain size (µm; top) and abundance (number of individuals.m^{-2}; bottom) of all samples gathered per year with standard error, for the intertidal (black) and shallow subtidal zone (grey) separately

3.3 Variation within beaches

By taking the results of all beaches together and focusing on the different beach elevation zones, it was possible to describe the spatial variation on the beach perpendicular to the water line. High up on the intertidal beach, near MHW, the median grain size is the highest, the silt fraction the lowest, the total organic matter the lowest and the carbonate content the lowest (table 3). The total organic carbon decreases from the MHW to MLW and increases from MLW to -8 m. The species richness increases as the median grain size and beach height decreases, except for the two deepest zones (6.73 and 7.47 species). This trend cannot be detected in the abundance or biomass values, although the biomass values also increase from MLW to -8 m.

Table 3: Mean of the abiotic and biotic factors per beach height zone, over all the sampled beaches and years; MHW: mean high water level; MLW: mean low water level

BEACH ELEVATION ZONES		**Beach elevation (m versus MLW)**	**Median grain size (µm)**	**Silt fraction (%)**	**Total organic matter (%)**	**Total organic carbon (%)**	**Carbonate content (%)**	**Species richness (number of species)**	**Abundance (number of individuals.m^{-2})**	**Biomass (g AFDW.m^{-2})**
4m to 6m (MHW)	INTERTIDAL ZONE	4.40	256.01	0.05	0.30	0.61	6.72	3.57	322.12	0.40
2m to 4m		2.90	253.28	0.05	0.41	0.55	8.40	4.24	118.97	0.33
0m (MLW) to 2m		1.44	226.72	0.01	0.44	0.38	9.45	4.73	49.53	0.37
-2m to 0m	SHALLOW SUBDTIAL ZONE	-1.30	191.98	2.05	1.04	0.53	9.39	6.97	89.86	5.97
-4m to -2m		-2.91	183.57	2.48	0.93	0.58	9.68	8.50	118.28	7.75
-6m to -4m		-4.94	178.03	8.63	1.00	0.69	9.47	9.47	135.02	10.04
-8m to -6m		-6.46	168.38	13.84	0.29	0.97	11.39	6.71	78.18	12.04
-10m to -8m		-9.02	170.72	6.29	0.60	0.55	15.29	7.49	64.04	1.73

According to figure 4, median grain size and slope or elevation on the beach account for about 77 % of macrobenthic variance on Belgian beaches. Moreover, the median grain size decreases when the beach height decreases (Spearman rank correlation: $r_s = 0.6439136$; $p<0,01$; figure 8). This is clearly visible in the intertidal zone, with only 4 samples showing a median grain size larger than 450 µm: one sample taken in 1997 in Schipgatduinen (464.00 µm) and 3 samples taken in 2006 in Oostende-Center (486.63 µm, 560.34 µm and 594.29 µm). In the shallow subtidal zone, the median grain size generally fluctuates between 150 and 200 µm with some very low values (median grain size smaller than 100 µm) between -1 and -8 m: 2 Oostende-Center samples taken in 2002, 2 Oostende-Fairway samples taken in 2006, 12 Wenduine samples taken in 2006, 2 Oostende-East samples taken in Spring and Autumn 2011 and 4 Nieuwpoort samples taken in Spring 2011. Overall, 96 % of all data points have a median grain size between 150 and 300 µm.

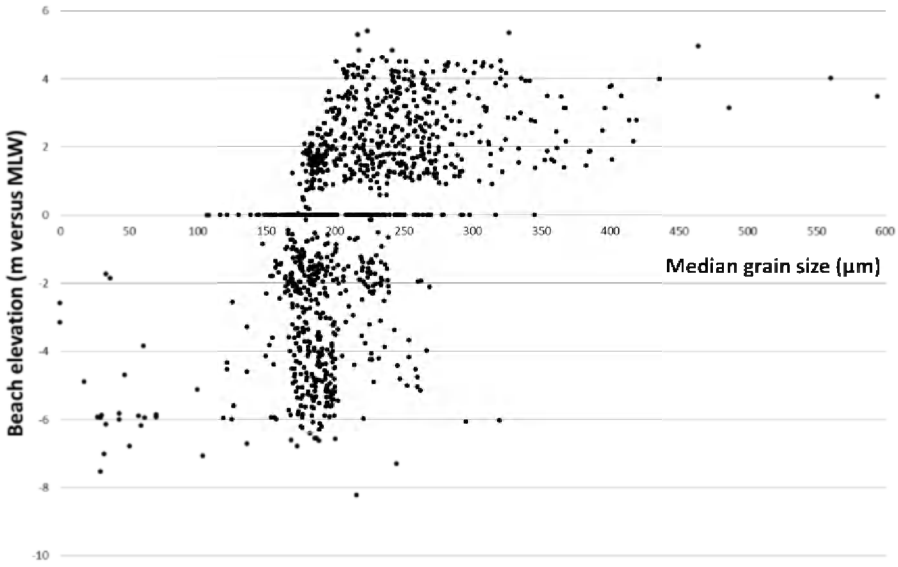


Figure 8: Beach elevation (m versus MLW) versus median grain size (µm)

In figure 9, the highest values of species richness (6 samples: 20 to 28 species), abundance (13 samples: 1000 to 3988 individuals.m^{-2}) and biomass (88 samples: 10 to 246.14 g AFDW.m^{-2}) are not depicted as they were scarce and masked the overall data pattern (1268 samples). However, they are still represented by the white spots in figure 9. Figure 9 shows that the highest species richness could be found in the shallow subtidal zone between -1 and -5 m (150 – 200 µm) and between -7 and -8 m (300 – 350 µm). The abundance is highest between 2 and 5 m when the median grain size varies between 200 and 250 µm. In the shallow subtidal zone, the highest abundance can be found between -2 and -5 m in sediment with a median grain size around 150 µm. The biomass is much higher in the shallow subtidal than in the intertidal zone, reaching its highest values (200 – 250 g AFDW.m^{-2}; white spots in graph) between 1 and -8 m in sediment with a median grain size between 150 and 200 µm.

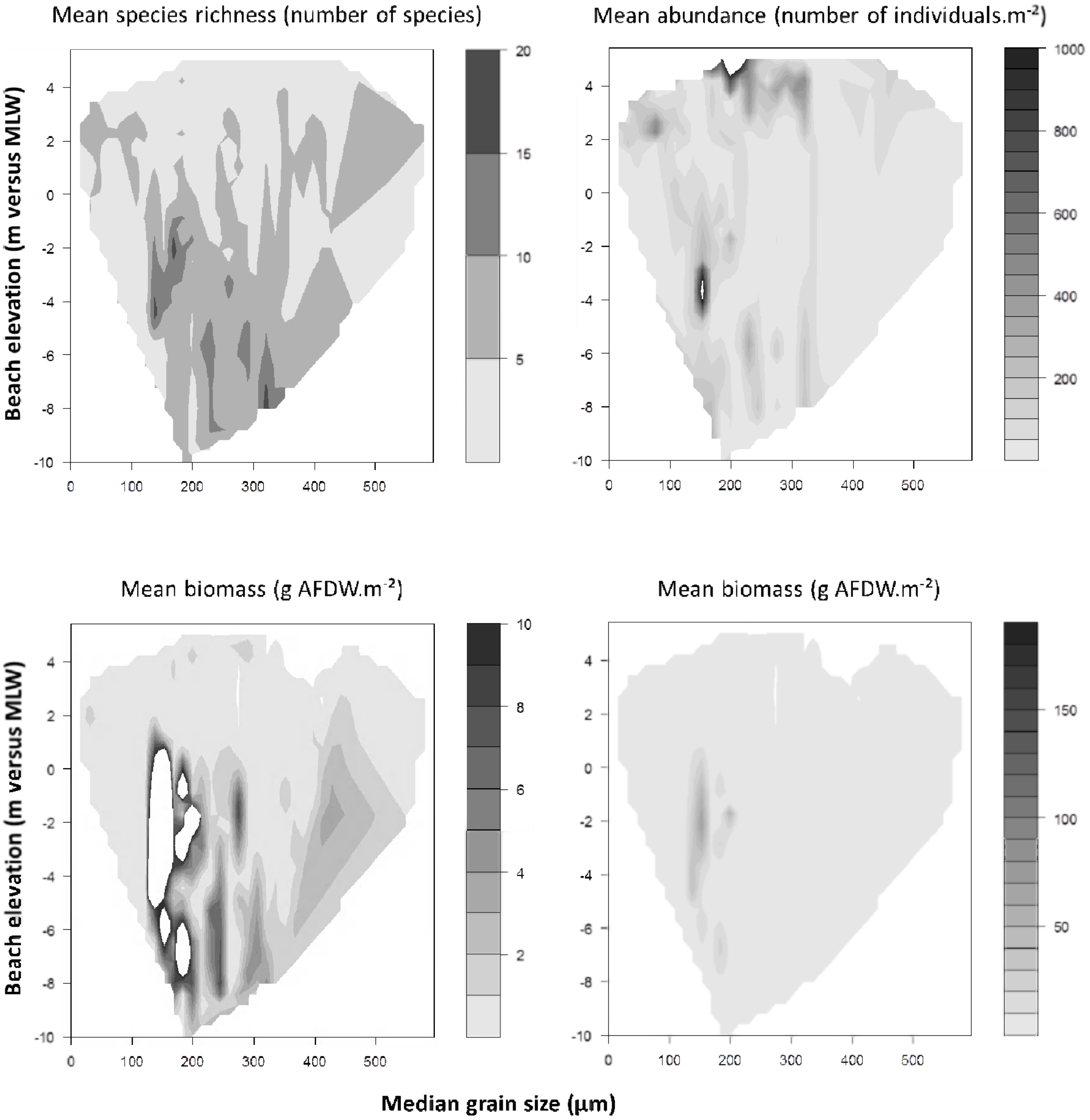


Figure 9: Contour plots of species richness (number of species), abundance (number of individuals.m^{-2}) and biomass (g AFDW.m^{-2}) over beach elevation (m versus MLW) and median grain size (µm); the highest values of each factor are not depicted but they are still visible in the figure as white spots. The contour plot with the highest biomass values is included to confirm the significance of the white spots.

3.4 Niche properties of the dominant species

The dominant species in the intertidal and shallow subtidal zone were determined by taking the five species that contributed the most to the total abundance (Table 4). The juvenile status of the bivalves *Ensis* species and *Macoma balthica* is clearly stated as they were significantly smaller than the adult individuals. Intertidal and shallow subtidal contour plots were made for each of these dominant species to unravel their habitat preferences and niche properties, based on beach elevation and median grain size.

Table 4: The dominant species in the intertidal and shallow subtidal zone over all sampled beaches and years

Intertidal zone	% of total abundance	Shallow subtidal zone	% of total abundance
Bathyporeia pilosa	25.04	*Ensis* juveniles	40.70
Scolelepis squamata	21.01	*Lanice conchilega*	7.64
Bathyporeia sarsi	13.36	*Cirratulidae species*	7.57
Eurydice pulchra	7.60	*Spio species*	6.25
Spiophanes bombyx	5.27	*Macoma balthica* juveniles	6.21

In figure 10, the highest values of *Bathyporeia pilosa* (2 samples taken in the high intertidal zone of De Panne in 1997 with 9522 and 15010 individuals.m^{-2}), *Bathyporeia sarsi* (1 sample taken in the middle of the intertidal zone of Nieuwpoort in spring 2009 with 2583 and 1 sample taken in the high intertidal zone of Raversijde in 1997 with 4766 individuals.m^{-2}), *Eurydice pulchra* (2 samples taken in the high intertidal zone of De Panne in 1997 with 1774 and 2554 individuals.m^{-2}; represented by white spot in figure 11), *Scolelepis squamata* (1 sample taken in the high intertidal zone of Mariakerke in autumn 2011 with 1969 individuals.m^{-2} and 1 sample taken in the high intertidal zone of Raversijde in 1997 with 2105 individuals.m^{-2}) and *Spiophanes bombyx* (3 samples taken in the low intertidal zone of Oostende-East in 2004 with 2500, 2558 and 5274 individuals.m^{-2}) are not depicted as they were scarce and masked the overall data pattern (686 samples). *Bathyporeia pilosa* and *Eurydice pulchra* prefer the highest intertidal beach zone (4 m – 5.5 m) and the finest sediment (200 – 250 µm). *Bathyporeia pilosa* also shares the middle of the intertidal zone (2.5 m – 4 m around 200 µm and 1.5 – 3.5 m around 280 µm) with *Bathyporeia sarsi*. On its own, *Bathyporeia sarsi* occupies the zone between 3.5 and 4.5 m (250 – 280 µm). Of the two polychaetes, *Spiophanes bombyx* prefers the lowest beach zone, close to MLW (1 m – 2 m) with sediment around 200 µm. *Scolelepis squamata* on the other hand thrives all over the intertidal beach but prefers the upper and middle area (2 m – 5.5 m). On Belgian sandy beaches, this species occurs in a wide range of sediment grain sizes (200 – 600 µm). Not only do the abundance values peak around 400 µm but they seem to increase again from 500 µm onwards.

In the shallow subtidal zone (figure 11), only the highest values of *Lanice conchilega* (1 sample taken in the shallow subtidal zone of Oostende-Center in 2004 with 9503 individuals.m^{-2}) is not depicted, because it masked the resolution of the classes. The *Ensis* juvenile bivalves prefer the zone between 2 m and -6 m with sediment between 250 and 300 µm. The abundance of *Macoma balthica* juveniles reaches peak levels in three different zones: (1) -2 m to -6 m (50 – 100 µm); (2) 0 m to -2 m (200 µm) and (3) 2 m to -4 m (250 µm). The highest abundance values of *Lanice conchilega* can be found between 0 and -6 m in sediment of 150 – 200 µm. The abundance of *Cirratulidae* species reaches peak values in three different zones: (1) -3 m to -5 m (50 – 100 µm); (2) 0 m to -2 m (200 µm) and (3) 2 m to -6 m (250 – 280 µm). They seem to co-occur with *Macoma balthica* juveniles. *Spio* species prefer two zones: (1) -6 m to -10 m (180 µm) and (2) 1 m to -6 m (150 – 300 µm).

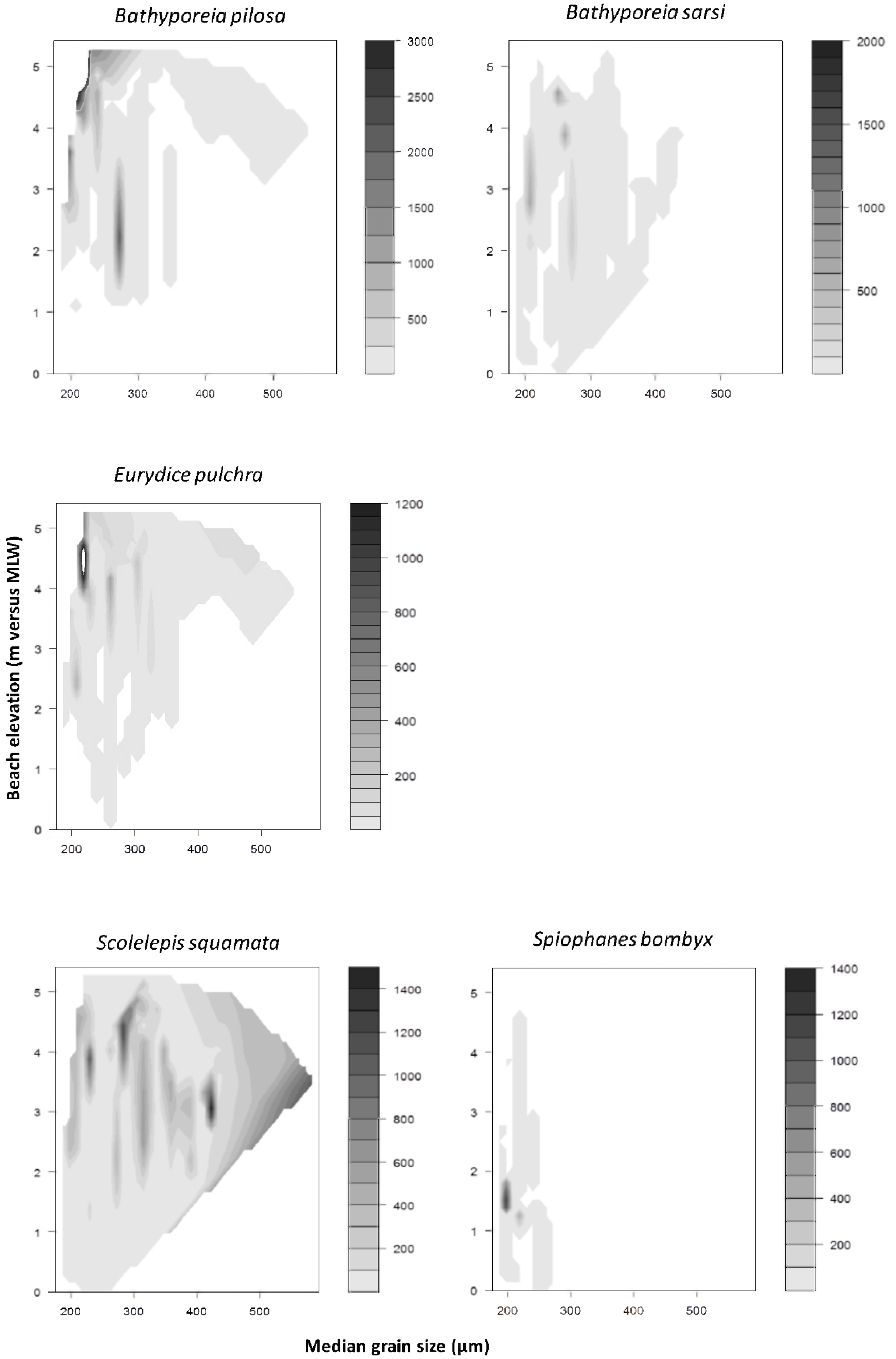


Figure 10: Contourplots showing the abundance of the five dominant macrobenthic species in the intertidal zone versus beach elevation (m versus MLW) on the Y axis and median grain size size (µm) on the X axis

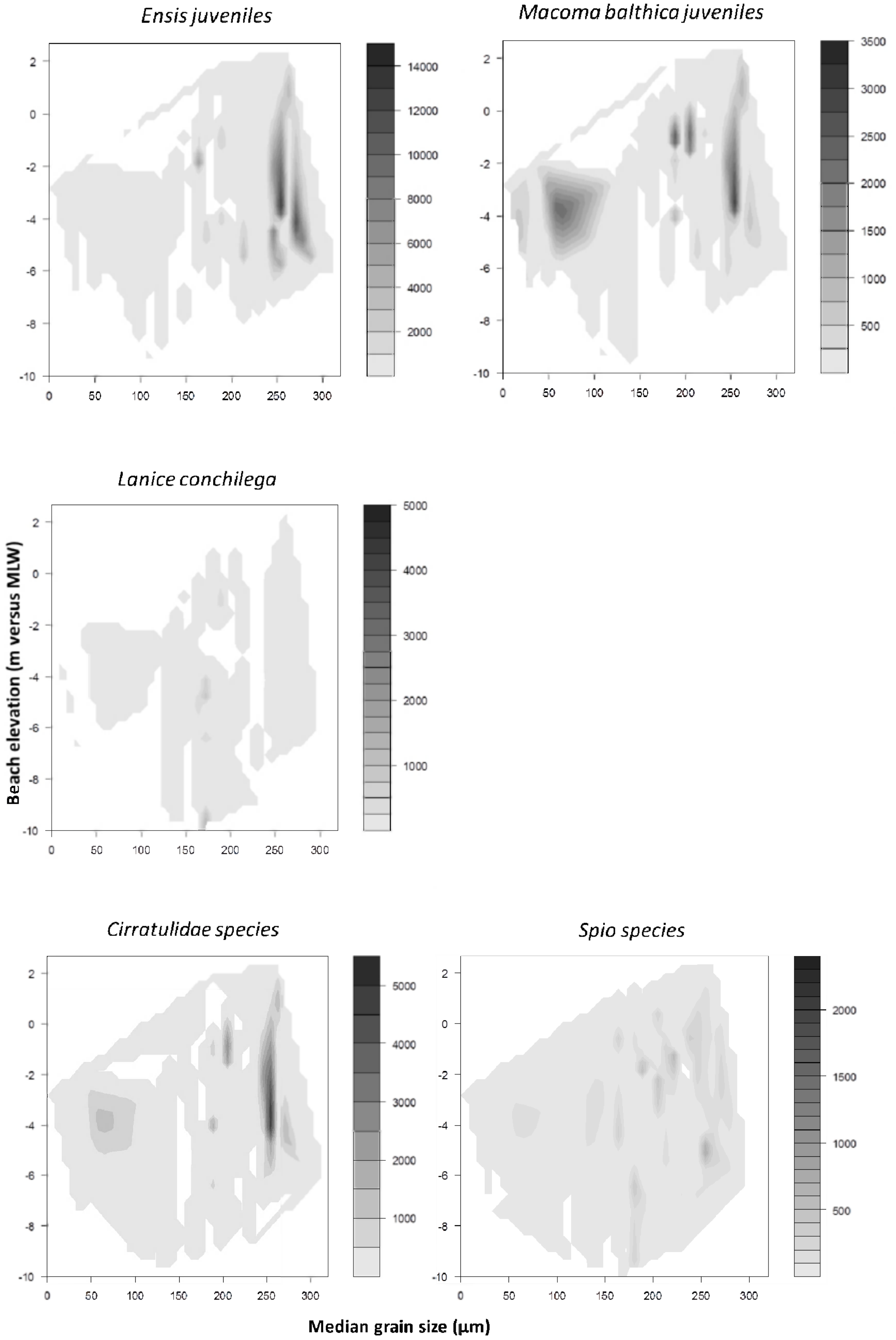


Figure 11: Contourplots showing the abundance of the five dominant macrobenthic species in the shallow subtidal zone versus beach elevation (m versus MLW) on the Y axis and median grain size size (µm) on the X axis

4. Discussion

In this study, we aimed to assess the ecological characteristics of the Belgian beaches prior to the implementation of the Belgian Master Plan for Coastal Safety and the concurrent multitude of beach nourishment projects over the next years. First a quantitative assessment of spatiotemporal variation in abiotic and biotic parameters was performed, followed by a more descriptive synthesis on niche properties of biodiversity metrics and the dominant species. All sampling locations were considered to be outside the influence of major impacting activities, except for Oostende-Center. This beach received a nourishment with very coarse sediment in 2006 and should yield some aberrant values compared to the other beaches. All other data represent the current state of the Belgian beaches. If it is assumed that changes at these locations are the result of natural factors rather than associated with field related disturbances, the abiotic and biotic variance across this group of locations at any one time may be taken to represent the normal limits to naturally induced change (Pearson & Mannvik 1998) in the Belgian coastal zone. As the main aim is to distinguish and allocate natural macrobenthic variation within the Belgian beach ecosystem to possible spatial or temporal trends, focus was placed on the overall trends and not on particular differences between beaches or between samples within beach zones.

4.1 Partitioning of abiotic and macrobenthic variance on Belgian beaches

The irregular and patchy distribution of sediments in the coastal zone (Ruddick et al. 1998), combined with its diverse topography, creates a wealth of habitats between the foredunes and the shallow subtidal zone of a beach. This supports a high capacity for various species assemblages, explaining the high benthic variability along the across-shore gradient. This variability at the macrobenthic species richness, abundance and species assemblage level is decreasing towards the open sea (Van Hoey et al. 2004).

Treatment of the intertidal and (shallow) subtidal zones as distinctly separate habitats is frequently the result of convenience and economy of sampling, with MLW being traditionally regarded as the transition between intertidal and (shallow) subtidal communities (Dexter 1979; Knott et al. 1983). Although multidimensional scaling (MDS) hinted a continuum between the intertidal and shallow subtidal zone (Figure 3), the data from both zones were treated separately to avoid misinterpretation of trends (Van Hoey et al. 2004). Partitioning of macrobenthic variance indicated that larger scale spatial and temporal variability is less important for community differentiation than variability within Belgian sandy beaches, linked to elevation on the beach and median grain size of the sediment, in both the intertidal and shallow subtidal zone. Invertebrate macrobenthos often exhibit patchy alongshore distribution on exposed sandy beaches (Jaramillo & McLachlan 1993; Dugan & McLachlan 1999; Schlacher et al. 2008) resulting in unclear big spatial trends. Large year-to-year temporal variability in macrobenthic community structure is prominent in temperate regions, but community shifts have not been detected. As long as the main habitat characteristics do not change drastically, the basic composition and the distribution (including natural variability, such as seasonality) of the respective communities will remain stable over long periods of time (Govaere et al. 1980). Temporal variability should thus be considered as being subordinate to spatial variation, which seems to be the case on the Belgian beaches. Moreover, exposed marine beaches have been defined as physically stressful environments (McLachlan 1983; McLachlan & Jaramillo 1995) and thus the best way to understand macrobenthic population variability is by documenting response to abiotic factors (Jaramillo & McLachlan 1993). If information is available, the role of biotic intraspecific and interspecific interactions, e.g. competition and predation, will be incorporated as well.

4.2 Spatial and temporal variation

Over all studied beaches and years, minimum and maximum values could be detected between which the natural variation runs its course (table 2). Confirming previous research (Degraer et al. 2003a; Degraer et al. 2003b; De Moor 2006; Speybroeck et al. 2008a), the mean intertidal median grain size coarsens while the finer shallow subtidal sediment becomes even finer when going from west to east. Over the years, the intertidal median grain size is always higher than the shallow subtidal values. On all beaches and in all years, polychaetes, crustaceans and bivalves dominated the macrobenthos, as is the case on most beaches on a world-wide scale (McLachlan & Jaramillo 1995). The distribution limits of beach species can expand and contract over time, displaying considerable fluctuations in abundance coupled with episodic settlement events and/or mass mortalities (Coe 1956; Defeo & McLachlan 2005). Most sandy beach macrobenthic species are short-lived (2 to 8 years), have rapid growth to maturity and high recruitment (McLachlan et al. 1996a). This analysis showed that the mean intertidal species richness, abundance and biomass are generally lower than the shallow subtidal values. Looking at the overall temporal evolution of the abundance however, the mean intertidal values were higher than the shallow subtidal values in 2002, 2006 and 2008. In these years, very high numbers of *Bathyporeia pilosa* (2006 and 2008), *Eurydice pulchra* (2002 and 2006) and *Scolelepis squamata* (2002, 2006 and 2008) were detected in the intertidal zone. These species caused peak abundance values in the intertidal zone, up to 6 times higher than the shallow subtidal values.

4.3 Variation within beaches

Population variations in across-shore distribution may arise from abiotic, e.g. swash activity and tides, and biotic, e.g. interspecific competition and predation, factors acting on different spatio-temporal scales. By jointly analyzing all beaches and focusing on the different beach elevation zones (as depicted in figure 3), it was possible to describe this spatial variation perpendicular to the water line on a Belgian beach. As evidenced in figure 4, median grain size and elevation on the beach account for around 77 % of the variation in macrobenthic abundance on the Belgian sandy beaches. Both abiotic factors are negatively intercorrelated, with the median grain size decreasing when the beach elevation decreases (figure 8). On a typical sandy beach, the coarsest sediment particles do lie at the top of the beach and grade down to the finest sediments at the waterline, due to wave activity (Short 1991). Three very coarse intertidal outliers were found in Oostende-Center in 2006. Within the time frame of this study, Oostende-Center was nourished with coarse offshore sediment in June 2004 (600 000 m^3), spring 2005 (100 000 m^3), spring and autumn 2007 (each 75 000 m^3), autumn 2008 (75 000 m^3), spring 2010 (75 000 m^3) and autumn 2011 (75 000 m^3). Oostende-Center was monitored in April and May 2004 and in September 2006, so only the 2006 values were influenced by the nourishment projects. The coarse sediment found in Schipgatduinen (464.00 µm) however is unexpected since this area lies in the western part of the Belgian coast, which is characterized by finer sediment (De Moor 2006). Moreover, the beach of Schipgatduinen is part of a nature reserve founded in 1975 so the coarse sediment cannot be attributed to major impacts like beach nourishment. In the shallow subtidal zone, waves are important in distributing and affecting sediments, although the effect decreases exponentially with depth (Gray & Elliott 2009). The median grain size in the shallow subtidal zone was indeed lower than in the intertidal zone with some very low values on different beaches along the Belgian coastline. Overall, 95.85 % of all data points have a median grain size between 150 and 300 µm.

As sand particle size and beach elevation correlate with physical environmental factors affecting the beach, they also influence the distribution of the beach fauna. The Belgian ultra-dissipative beaches are gently sloped with fine to medium sands and they harbour a diverse, abundant macrofauna (Ricciardi & Bourget 1999; Brown & McLachlan 2002; Speybroeck et al. 2008a). Higher up on the beach, near MHW, the median grain size is the highest while the silt fraction, the total organic matter and the carbonate content are the lowest (table 3). The infaunal species richness, abundance and biomass usually increase with decreasing beach elevation and median grain size (Jaramillo & McLachlan 1993; Haynes & Quinn 1995). This trend can be observed for the species richness, except for the two deepest zones (6.73 and 7.47 species) and for the biomass, with peak values between 1 and -8 m in fine sediment (150 – 200 µm). The abundance is highest between 2 and 5 m where the median grain size varies between 200 and 250 µm and between -2 m and -5 m in sediment with a median grain size around 150 µm. Very high intertidal abundance values of species have been recorded before (Degraer et al. 2003b) and were only due to *Scolelepis squamata.* The contour plots however, suggest that these high values can be attributed to *Bathyporeia pilosa*, *Bathyporeai sarsi*, *Eurydice pulchra* and *Scolelepis squamata.* The relatively low abundance of benthic macrofauna near MLW may be related to unfavourable hydrodynamic and environmental conditions like mobile sediments and higher predation pressure of hyperbenthic and epibenthic organisms (Janssen & Mulder 2005). In the shallow subtidal zone, the high abundance values can be attributed to juvenile *Macoma balthica* and *Ensis* bivalves.

4.4 Niche properties of the dominant species: a synthesis

Zonation as studied by ecologists exists only during the low tide period. As the tide rises, populations move with some entering the water column while zones get compressed. The high mobility of most species, coupled to the disturbance effects of changing wave energy levels, results in species distributions being quite variable from day to day. Migrations of the fauna shuffle and recreate zones constantly and with each tide (McLachlan & Jaramillo 1995). Hence, faunal zones do not exhibit sharp boundaries, but rather tend to be distinguished by the presence of characteristic taxa. Dominant species typify the macrobenthic community. The five species that contribute the most to the total abundance in either the intertidal or shallow subtidal zone are considered to be dominant species of the Belgian sandy beaches. Knowledge of their observed niche together with their interpolated occurrence greatly improves our understanding of the Belgian sandy beach ecosystem (figure 12).

As stated by Van Hoey (2004), the *Eurydice pulchra – Scolelepis squamata* community is exclusively found at the upper intertidal zone of Belgian sandy beaches and is characterized by a low diversity (5 species per sample) and moderate abundances (983 individuals.m^{-2}), mainly of *Eurydice pulchra, Bathyporeia pilosa, Bathyporeai sarsi* and *Scolelepis squamata. Bathyporeia* species, *Eurydice pulchra* (Eleftheriou & McIntyre 1976; Degraer et al. 1999a) and *Scolelepis squamata* (Eleftheriou & McIntyre 1976; McDermott 1987; Souza & Gianuca 1995) are found to be abundant on many European beaches. Parallel communities, in which species might be replaced by (functionally) similar species, can be found within the upper intertidal zone of many sandy beaches worldwide (McLachlan & Jaramillo 1995). The lower intertidal and shallow subtidal zone cannot be distinguished from one another, based on dominant species, as juvenile *Ensis* and *Macoma balthica* bivalves (indicator species of *Macoma balthica* community, (Van Hoey et al. 2004), *Lanice conchilega* (indicator species of *Abra alba – Kurtiella bidentata* community, (Van Hoey et al. 2004), *Cirratulidae* and *Spio species* occupy both zones. *Spiophanes bombyx* is the only species that lingers in the narrow zone around MLW. None of the dominant species are indicator species of the *Ophelia borealis – Glycera lapidum* and *Nephtys cirrosa* communities (Van Hoey et al. 2004). The juveniles of *Ensis* and *Macoma balthica* bivalves (median grain size > 200 µm) seem to inhabit a different habitat than their adult stages. (median grain size < 200 µm) (Degraer et al. 2006). In the shallow subtidal zone, the polychaete *Lanice conchilega* can be found in fine sediment 150 – 200 µm between 0 and -6 m. This polychaete adds or alters physical, chemical and biological factors and is therefore often referred to as an ecosystem engineer (Rabaut et al. 2007). Reefs of this tube-building and habitat-forming polychaete are known to represent hotspots of biodiversity within intertidal and (shallow) subtidal soft sediments of the North Sea (Degraer et al. 2008a). Whereas individuals can be found in a wide range of fine to medium muddy sands to throughout the North Sea (Degraer et al. 2006; Van Hoey 2006; Willems et al. 2008), dense aggregations or reefs do not (Degraer et al. 2008a). Nonetheless, the habitat created by *Lanice conchilega* seems to be important for higher trophic levels such as juvenile flatfish (Rabaut et al. 2010) and birds (Godet et al. 2008).

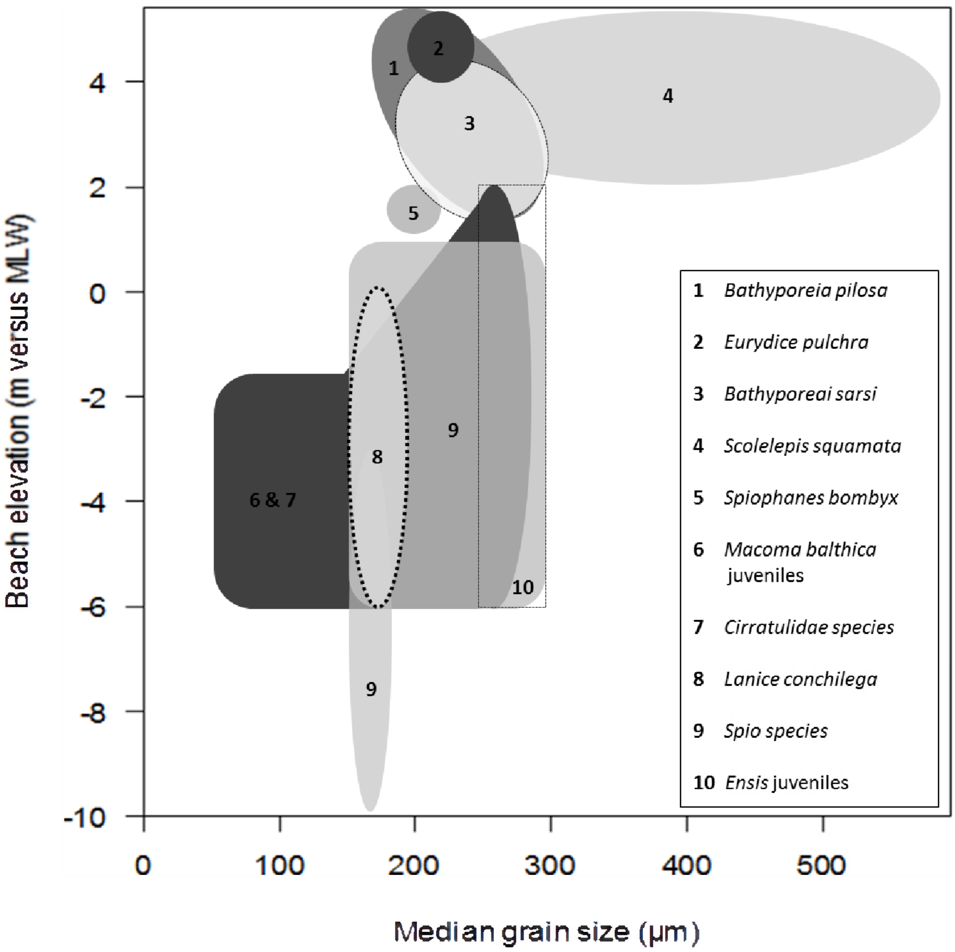


Figure 12: Observed niche and interpolated occurrence of the dominant Belgian macrobenthic species along an 'average' Belgian beach transect

5. Conclusion

We assessed the ecological characteristics of the Belgian beaches prior to the implementation of the Belgian Master Plan for Coastal Safety. The results of the present study show (1) a distinct difference in overall community structure between the intertidal and shallow subtidal zone; (2) the higher importance of variability within Belgian sandy beaches, linked to elevation on the beach and median grain size of the sediment, compared to big scale spatial and temporal variability; (3) the confirmation of some generally accepted spatial and temporal trends in abiotic factors, e.g. an overall median grain size between 150 and 300 µm, and in macrobenthic species richness, abundance and biomass; and (4) the realized niches of the dominant macrobenthic species of the Belgian beaches. Since all sampling locations were considered to be outside the influence of major impacting activities, these findings can be used as a preconceived basis (t0 situation) of inherent, natural and normal abiotic and macrobenthic variability on the Belgian beaches. This study will also aid in the detection of possible impact effects of past, current and future beach nourishment projects, as stipulated in the Belgian Master Plan for Coastal Safety.

Chapter 3: The monitoring of ‘ecological’ beach nourishment impacts on macrobenthos, within a Special Area of Conservation (SAC) along the Belgian coast

Sarah Vanden Eede, Dries Bonte, Magda Vincx (in prep.) The monitoring of ‘ecological’ beach nourishment impacts on macrobenthos, within a Special Area of Conservation (SAC) along the Belgian coast.

In preparation for submission

Abstract

The expected increase in intensity and frequency of storm events combined with the undeniable rise in sea water level, both connected to climate change, poses a threat to coastal low lands. As it safeguards the natural dynamics of the coast, beach nourishment has rapidly become the preferred protective and restoring measure in coastal zones worldwide. We tested whether optimizing the main technical aspects of a nourishment according to ecological recommendations, e.g. a gentle sloped beach with nourished sand resembling the original sediment very closely (average median grain size of 224 µm), leads to an 'ecological' nourishment with minor to no impact effects on the macrobenthos.

Monitoring of nourishment impact effects on the macrobenthic community structure can be regarded as a 'natural experiment' and is based on a BACI (Before/After, Control/Impact) design. A putatively impacted area and a matching control site are monitored in a time series (2000 – 2012) straddling the impact event (2009). Impact effects should then show up as interactions between the temporal and spatial factors. A wider, higher and flatter intertidal beach with coarser sediment (from 215.89 ± 3.63 µm in 2008 to 280.23 ± 8.94 µm in spring 2010) was created and no return to the pre-nourishment conditions in sedimentology was visible three years after nourishment. The sediment grain size distribution had changed as well, showing slow recovery in the three post-nourishment years. The analysis of the macrobenthos community structure showed that the nourishment under ecological optimal conditions on the beach of Lombardsijde yielded no significant effects on both the intertidal and shallow subtidal beach ecosystem 6 months after the nourishment. Within this time frame, the macrobenthos community had seemingly recovered from the impact of the ecological nourishment. Ecological nourishment thus proves to be the least ecologically damaging way of combating erosion, compared to all other coastal engineering activities.

Keywords: beach nourishment, macrobenthos, monitoring

1. Introduction

During the last decade, climate change has become a much debated topic. The expected increase in intensity and frequency of storm events combined with the undeniable rise in sea water level, both connected to climate change, poses a threat to coastal low lands. Beach erosion is likely to accelerate, driven by predicted consequences of climate change and coastal development (Brown & McLachlan 2002; Feagin et al. 2005; Slott et al. 2006; Schlacher et al. 2012). This diminishes the natural buffer function of beaches, leaving the hinterland less protected from scour, inundation and wave erosion (Young & Bryant 1992; Defeo et al. 2009).

Traditionally, coastal defence focused on beach stabilization by building hard structures of stone, concrete, wood, steel or geotextiles such as groynes, breakwaters, seawalls and dykes (Charlier et al. 2005). However, these structures alter the natural hydrodynamic system of waves and currents, thereby affecting sand transport rates, which in turn control the erosion dynamics of beaches (Defeo et al. 2009). There has been a gradual change from hard to soft coastal defence techniques, like beach nourishment. Beach nourishment is in essence the process of mechanically or hydraulically placing an amount of sand directly on an eroding shore to restore or form, and subsequently maintain, an adequate protective or desired recreational beach. As it safeguards the natural dynamics of the coast, beach nourishment has rapidly become a widely applied protective and restoring measure in coastal zones worldwide (Finkl & Walker 2002; Greene 2002; Hamm et al. 2002; Hanson et al. 2002; Speybroeck et al. 2006a; Cooke et al. 2012) for short-term urgencies (i.e. storm-induced erosion) as well as long-term issues (i.e. structural erosion and relative sea-level rise).

Even though beach nourishment is considered as the more ecologically sound option, an impact of such magnitude can be expected to affect the entire beach ecosystem (Speybroeck et al. 2008a). Coastal engineers prefer coarser grain sizes, obtained from marine extraction zones or nearby channel dredging, to produce a steeper, more stable and longer lived nourishment (Finkl & Walker 2002). Due to the forces of winds and waves, nourished sand will move between the dunes, the intertidal beach, the shallow subtidal and offshore zones until a stable equilibrium profile is achieved (Zeidler 1987), thus affecting the entire beach ecosystem and not only the nourished site. Unfortunately, there are still many uncertainties concerning the effects of beach nourishment. Natural perturbances, like storms, natural variability and spatial patchiness of organisms on sandy beaches may even obscure any but the largest effects. Long term data and large-scale datasets, field experiments and monitoring programmes can reveal insights into natural dynamics, or at least dynamics when unimpacted (Stauble & Nelson 1985; Defeo et al. 2009; Gray & Elliott 2009). Most studies documenting qualitative changes in the beach community have attributed these differences to natural variation (Culter & Mahadevan 1982; Saloman et al. 1982; Turbeville & Marsh 1982; Grober 1992), urging to interpret impact study results with caution.

The biotic composition of macrobenthos is an important indicator of the environmental quality and health of the benthic ecosystem (Goldberg 1988), playing a key role in the wider beach ecosystem and food web. Macrobenthos is generally defined as the organisms measuring over 1 mm long and living buried in the sediment. This group of bivalves, polychaetes, crustaceans and echinoderms feed primarily on faunal detritus and to a lesser extent on algal benthos and detritus (Sundbäck & Persson 1981; Josefson et al. 2002) and they are a major food source for birds and epibenthos. Owing to the highly dynamic nature of their environment, mainly determined by waves, tides and winds, macrobenthos species have a high, but not limitless tolerance towards several forms of environmental stress (Jaramillo & McLachlan 1993; Moffett et al. 1998). Therefore, according to many authors, nourishment should cause only minor and/or temporary damage to the ecosystem (Löffler & Coosen 1995; Miller et al. 2002). However, questions of recovery are less studied than those of short-term impacts (Schlacher et al. 2012). The currently accepted time frame for recovery of a single year may thus not reflect the return to a climax stage or the return to pre-nourishment conditions (Jones et al. 2008; Leewis et al. 2012; Schlacher et al. 2012).

Optimizing the technical aspects of future nourishment projects is indispensable to maintain an ecologically healthy beach ecosystem. For ecologically good practice of beach nourishment, it is advised (1) to choose nourishment sands with a sediment composition comparable to that of the natural sediment, (2) to avoid short-term compaction by ploughing immediately after construction, (3) to execute the nourishment in a period of low beach use by birds and other mobile organisms, (4) to choose a number of smaller projects rather than a single large nourishment project and (5) to select the nourishment technique with respect to the local natural values. The preferred time of nourishment entirely depends on the nature and location of the beach and the species inhabiting or exploiting it, but in temperate regions, the ideal period is the winter season (Speybroeck et al. 2006a).

This study aims at unravelling the *in situ* ecological effects on the soft-sediment macrobenthos of the beach nourishment on the Belgian beach of Lombardsijde. This nourishment was performed from March until September 2009, under optimal ecological conditions, e.g. phased nourishment project with nourished sand closely matching the original sediment and only moderate beach profile changes. The soft substrates of this beach have been extensively monitored prior to (Welvaert 2005; Van Ginderdeuren et al. 2007; Vanden Eede et al. 2008), during (Vanden Eede & Vincx 2010) and after the nourishment (Vanden Eede et al. 2013). To account for natural variation, the parallel monitoring of a control site, e.g. Nieuwpoort, was included in the research and several seasonal studies were performed between 2004 and 2012. This paper has three aims: (1) to quantify *in situ* ecological nourishment effects on macrobenthos, (2) to determine whether or not the macrobenthos has recovered within a period of 3 years and (3) to ascertain and confirm that ecological nourishment is the way forward within the Belgian coastal defence policy.

2. Material and methods

2.1 Study Area

According to the subdivision of the shallow Belgian coastal zone, Belgian beaches have a clearly defined intertidal and shallow subtidal zone (figure 1). The landward boundary follows the high water mark obtained by LIDAR observations of the Belgian coast in 2011 (data provided by the Agency for Maritime and Coastal Services: Coastal division – MDK) and the seaward boundary for the shallow subtidal foreshore is the 1 nautical mile from the zero depth (0 m) bathymetric line (figure 1).

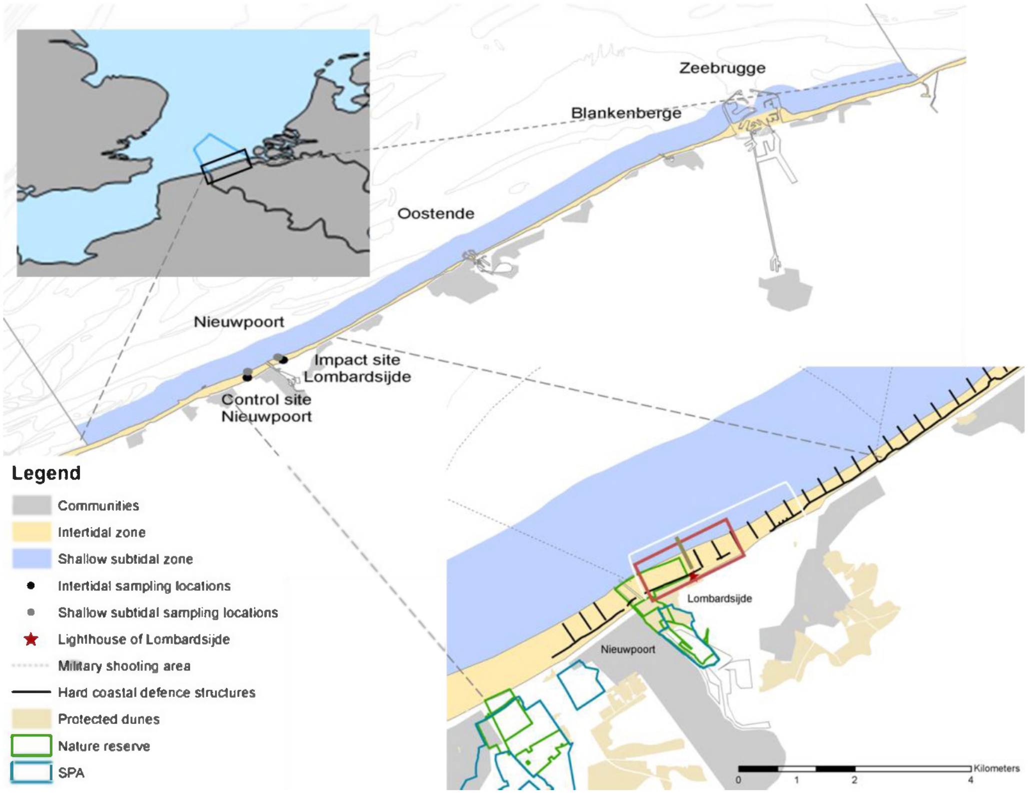


Figure 1: The Belgian coastal zone, with a distinction between the intertidal (light brown) and shallow subtidal zone (blue) at impact site Lombardsijde and control site Nieuwpoort, showing the location of the nourishment site (red box) within the nature reserve 'Ijzermonding' (green shape) and the Special Area of Conservation (SAC, white borders)

The beach of Lombardsijde is situated at the eastern side of the fairway to Nieuwpoort (figure 1). The whole beach is part of a Special Area of Conservation (SAC). The western part falls within the nature reserve 'IJzermonding' and is located in front of a Special Protection Area (SPA) and protected dunes while the eastern part falls within a military domain. The intertidal zone of Lombardsijde beach was nourished from March until September 2009, under optimal conditions. Approximately 650 000 m³ of sand, dredged from the fairway to Oostende, was deposited on top of the beach over a distance of around 1 200 m (15 m beach width at + 5.5 m followed by a slope of 1/70). The accumulated sand was moved with a bulldozer to profile the beach after deposition works finished. The sand resembled the original sediment, with a grain size between 200 and 250 µm. Hence, 250 m wide wet beach and 50 m wide dry beach were created. The aim of the Flemish government is to enclose part of Lombardsijde beach as a nature reserve for both birds and seals. The control site of any impact site should be located out of reach, preferably updrift, of the actual impact, e.g. the beach nourishment. The climate of flood currents, waves and winds at the Belgian coast is dominated by a southwest to northeast direction, narrowing the selection to all beaches located southwest from Lombardsijde. The beach of Nieuwpoort was chosen as a control site for Lombardsijde. It is situated in front of the municipality of Nieuwpoort at the western side of the fairway to Nieuwpoort (figure 1).

2.2 Data

Since 1997, the Marine Biology Research Group of Ghent University coordinated numerous macrobenthic studies along the Belgian coastline. The integrated database for the impact site Lombardsijde and control site Nieuwpoort, comprises all available relevant data gathered during the period 2008 – 2012 in the intertidal and shallow subtidal zones (table 1). Sampling was conducted once before, two times during and six times after nourishment (3 years, spring and autumn season).

Table 1: Sampling locations used for the integrated macrobenthic database. Only data collected in intertidal and shallow subtidal zones of the Belgian coast (key: S=spring (grey zones); A=autumn; a (Vanden Eede et al. 2008); b (Vanden Eede & Vincx 2010); c (Vanden Eede & Vincx 2011b); d (Vanden Eede et al. 2013); red = nourishment phase)

INTERTIDAL ZONE	**2008**	**2009**	**2009**	**2010**	**2010**	**2011**	**2011**	**2012**	**2012**
	S	**S**	**A**	**S**	**A**	**S**	**A**	**S**	**A**
Nourishment time	t0	t	t	t1	t2	t3	t4	t5	t6
Study	a	b	b	c	c	d	d	d	d
Nieuwpoort (control)	26	26	26	15	15	15	15	15	15
Lombardsijde (impact)	25	39	26	15	15	15	15	15	15
Total amount of samples	**51**	**65**	**52**	**30**	**30**	**30**	**30**	**30**	**30**

SHALLOW SUBTIDAL ZONE	**2008**	**2009**	**2009**	**2010**	**2010**	**2011**	**2011**	**2012**	**2012**
	S	**S**	**A**	**S**	**A**	**S**	**A**	**S**	**A**
Nourishment time	t0	t	t	t1	t2	t3	t4	t5	t6
Study	a	b	b	c	c	d	d	d	d
Nieuwpoort (control)	25	25	25	15	15	15	15	15	15
Lombardsijde (impact)	25	25	24	15	15	15	15	15	15
Total amount of samples	**50**	**50**	**49**	**30**	**30**	**30**	**30**	**30**	**30**

2.3 Sampling method

Intertidal sampling always started at high tidea and followed the receding water down the beach, as the organisms tend to cluster here very close to the water line, ending at low tide. Samples were taken by excavating a quadrat frame (surface area, 0.1026 m²) to a depth of 0.15 m. The samples were immediately sieved (ø 1 mm) and preserved in an 8 % formaldehyde-seawater solution. Next to each biotic sample, one core sample (ø 3.6 cm) for sediment analysis was collected. At every sampling location, a beach profile (Vanden Eede & Vincx 2011b) and the geographic position (geographic wgs84) were noted. As we always sample at the water line, we can deduce the real elevation of the sample locations to the water surface (MLW, mean low water level) using the M2 reduction model (Van Cauwenberghe et al. 1993).

The shallow subtidal zone was sampled with a small vessel at high tide. At every sample location, a Van Veen grab was lowered to take a sediment sample (0.1026 m²). Simultaneously, the coordinates (UTMwgs84), time and depth of each sample were determined. The depth was afterwards corrected (compared to MLW) using the M2 reduction model. Before opening the Van Veen grab, a core sample (ø 3.6 cm) for sediment analysis was taken. The samples were immediately sieved (ø 1 mm) and preserved in an 8 % formaldehyde-seawater solution.

2.4 Lab procedures

The sieved samples for faunal analysis were stained with Rose Bengal and elutriated ten times to collect the macrobenthos. The remaining material was examined to collect the larger organisms that were too heavy to be floated off by elutriation. All macrobenthic organisms, except for Oligochaeta, Actiniaria and Nematoda, were identified to species level, where possible, and counted. Faunal abundance was extrapolated to the number of individuals per m². Biomass (gram Ash Free Dry Weight or g AFDW) estimates were obtained by loss of mass on ignition (480 °C for 2 h) of oven-dried samples (110 °C for 24 h) (Van Ginderdeuren et al. 2007; Vanden Eede et al. 2008; Vanden Eede & Vincx 2010, 2011b; Vanden Eede et al. 2013). The biomass was calculated on higher taxon level except for the dominant species, being *Bathyporeia pilosa and Bathyporeia sarsi*, *Ensis* juveniles, *Eurydice* species, *Lanice conchilega*, *Macoma balthica* juveniles, *Nephtys cirrosa*, *Owenia fusiformis*, *Scolelepis squamata* and *Spisula subtruncata*.

After drying the sediment samples, the grain size distribution was determined with a Malvern Mastersizer 2000G laser with diffraction module (measuring range: 0.02 – 2000 µm). In this paper, sediment was characterized by median grain size (µm) and silt fraction (< 63 µm). The percentage of shell fragments was determined by means of the percent volume of sediment remaining on the 1 mm sieve (carbonate content). The values for Total Organic Carbon (TOC) were calculated with an automatic element analyzer 1500 Carlo Elba. The percentage of Total Organic Matter (TOM) was obtained by loss of mass on ignition, using the biomass analysis strategy.

2.5 Data analysis

In total, 348 intertidal and 329 shallow subtidal samples were gathered between 2008 and 2012. 171 species were identified from which 81 species were taken into account for further macrobenthic analysis (50 species in the intertidal zone and 80 species in the shallow subtidal zone). The species excluded from the analyses clearly attributed to other benthic groups, e.g. meiobenthos, hyperbenthos or epibenthos, and/or were gathered in a non-representative way, e.g. a frequency of occurrence of less than 0.5 % and a maximum of 5 individuals per sample (Appendices – Chapter 3 – table A).

Monitoring of the human-induced impact effects on the macrobenthic community structure can be regarded as a "natural experiment". It is however still an observational study and not an experiment, in the strict statistical sense of randomly allocating treatments to experimental units. This study is based on a BACI (Before/After, Control/Impact) design (Underwood 1991, 1992; Green 1993; Underwood 1994; Smith 2002). A putatively impacted area and a matching control site are monitored in a time series straddling the impact event. Impact effects should then show up as interactions between the temporal and spatial factors.

All tables and basic graphs were made in Microsoft Excel 2010. The sampling location map was created with ESRI ArcMap Version 9.3. Wilcoxon signed-rank tests were performed in R 2.14.2 to observe statistical differences between the mean values of the abiotic and biotic variables of impact and control site.

Multivariate analyses in Primer v6 were used to detect patterns in the distribution of macrobenthic assemblages, without transforming data. Multidimensional scaling was based on Bray-Curtis similarity, a statistic used to quantify the compositional dissimilarity (abundance data per species) between different sampling locations. Samples within a cluster are more similar than samples of different clusters (Clarke et al. 2008).

Regression analysis was used to estimate the relationship between median grain size (µm; dependent variable) and three independent variables, e.g. Elevation (m versus MLW), Time (t0, t, t1, t2, t3, t4, t5, t6) and Treatment (impact or control site). The term 'Treatment' was used in the analyses rather than 'Location' since the key comparison is between the experimental treatment (i.e. nourishment) and the control.This is the full model: Grain size = Elevation + Treatment + Time + Elevation*Treatment +

Elevation*Time + Treatment*Time + Elevation* Treatment*Time. For each pairwise comparison (t0-t; t0-t1;t0-t2 etc.), a significant interaction term Elevation*Treatment shows an average impact of the nourishment on the sediment grain size distribution. The relation between grain size and elevation has changed between the two treatments over time when the p-value of the interaction term Elevation*Treatment*Time is significant ($p < 0.05$). This means the sediment grain size distribution changed between the two beaches over time, and this could be due to changes at different times after the nourishment. If these changes become less clear over time, the sediment grain size distribution might have evolved back to original levels, e.g. recovery of the sediment grain size distribution has taken place.

The variance in the macrobenthic community structure was analysed with the adonis function of the Vegan package in R 2.14.2. This technique fits permutative linear models (e.g., factors, polynomial regression) to Bray-Curtis distance matrices and allows for nonparametric analysis of variance using these distance matrices. We follow the same approach as described by Schlacher et al. (2012). Under this approach, an impact is indicated by statistically significant Treatment*Time interactions ($p < 0.05$). All factors (i.e. Treatment and Time) were fixed. This is the full model: Community Structure = Elevation + Treatment + Time + Elevation*Treatment + Elevation*Time + Treatment*Time + Elevation*Treatment*Time. The final model excluded the interaction terms Treatment*Time and Elevation*Treatment*Time.

3. Results

Multidimensional scaling (MDS) of the macrobenthic community composition of all sampling data did not indicate a clear separation of the macrobenthic data based on spatial (different treatments) or temporal (different years, seasons, nourishment phases) variables, except for samples from the intertidal or shallow subtidal zone (Figure 2). Since these two habitats are substantially different to harbor other macrobenthic communities (Van Hoey et al. 2004) and only the intertidal zone was nourished, we treated the intertidal and shallow subtidal data separately. Appendices – Chapter 3 – table B1 and B2 give an overview of the abiotic and biotic mean values of the impact site per year and season for the intertidal and shallow subtidal zones separately while Appendices – Chapter 3 – table C1 and C2 provide the same overview for the control site. The Wilcoxon-values between impact and control site for all these abiotic and biotic factors per season in the intertidal and shallow subtidal zone can be found in Appendices – Chapter 3 – table D.

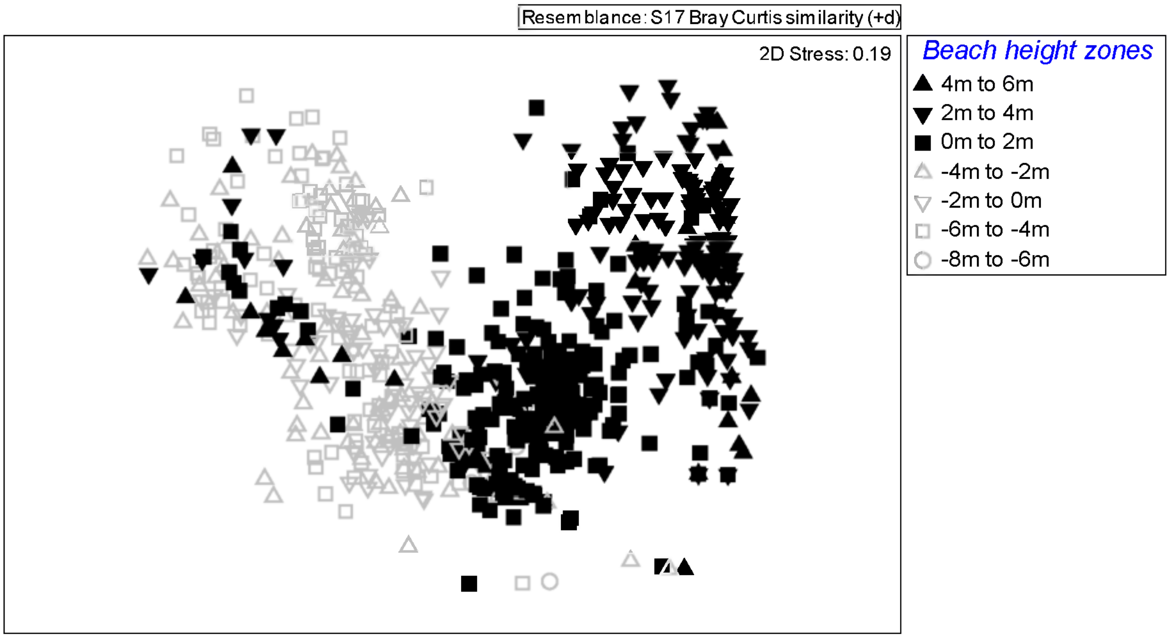


Figure 2: MDS plot of all data, showing a partitioning into two groups, being intertidal (black) and shallow subtidal (grey) samples with overlap around mean low water level, and to a lesser extent into beach elevation zones

3.1 Habitat changes

The nourishment activities created a wider and flatter intertidal beach (red line in figure 3 left), heightened over its entire width by the added sand (red dotted line in figure 3 left). Natural variation in accretion and sedimentation along the Belgian coastline can be detected by the similar evolution in the beach profiles of the control and impact site over the studied years and seasons. The final nourished beach slope (red dotted line in figure 3 left) resembled the original slope, although the entire beach was heightened. Moreover, it resembled the beach profile of the control site (red dotted line in figure 3 right) almost perfectly. Three years later, the impacted beach has still not returned to its original lower pre-nourishment profile.

Prior to the nourishment, the median grain size of the impact and control site did not differ significantly (table 2 and figure 6). Even though the nourished sand resembled the original sediment very closely with an average median grain size of 224 µm, the intertidal mean median grain size became coarser (from 215.89 ± 3.63 µm in 2008 to 280.23 ± 8.94 µm in spring 2010) than the value of the control site (stable mean around 200 µm) during and following the nourishment (table 2; figure 4 and 6), except in autumn 2010 when the control site had the coarsest sediment (impact site: 168.12 ± 4.13 µm and control site: 259.69 ± 4.35 µm). In the shallow subtidal zone, finer sediments were found at the impact site prior to nourishment. During the nourishment, the impact sediments became slightly coarser than at the control site and this was also found in spring 2011. In autumn 2010, a coarse peak value could be detected in the control site (244.79 ± 4.31 µm), followed by the lowest registered value (spring 2011: 137.51 ± 14.47 µm).

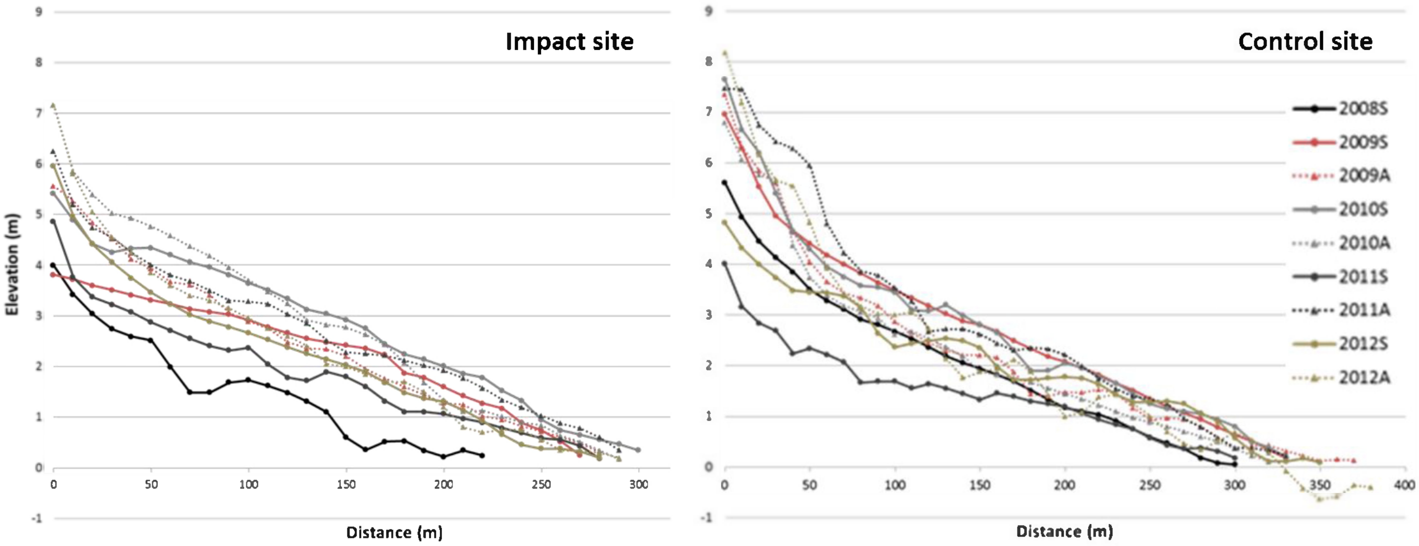


Figure 3: Intertidal beach profiles of the impact site (left) and control site (right) from spring 2008 (2008S) onwards

Table 2: Comparison of mean median grain size ± SE (µm) between treatments (control and impact), time (years) and levels of the beach (intertidal and shallow subtidal)

	INTERTIDAL			SHALLOW SUBTIDAL		
Year	Impact Site	Control Site	Impact-Control	Impact Site	Control Site	Impact-Control
2008	215.89 ± 3.63	206.35 ± 3.55	9.53	182.30 ± 2.35	186.48 ± 2.01	-4.18
2009	212.98 ± 3.94	197.53 ± 3.02	**10.38**	188.42 ± 2.69	182.33 ± 2.38	6.09
	236.84 ± 8.53	196.24 ± 2.83	**40.60**	195.89 ± 6.62	181.50 ± 2.03	14.39
2010	280.23 ± 8.94	201.75 ± 5.32	**78.48**	158.35 ± 12.17	182.32 ± 2.57	-23.97
	168.12 ± 4.13	259.69 ± 4.35	**-91.57**	190.12 ± 2.20	244.79 ± 4.31	**-54.68**
2011	262.80 ± 6.84	202.29 ± 5.63	**60.51**	177.04 ± 2.36	137.51 ± 14.47	**39.53**
	252.39 ± 9.54	206.81 ± 8.25	**45.58**	168.94 ± 5.46	178.84 ± 4.30	-9.89
2012	267.89 ± 5.69	208.63 ± 6.97	**59.26**	172.98 ± 1.39	184.06 ± 2.81	**-11.08**
	247.86 ± 2.87	201.97 ± 1.76	**45.89**	174.73 ± 0.35	178.92 ± 0.78	-4.19

The coarsest median grain size can thus be detected high on the beach (figure 4). Since the same trend can be observed in both the impact and control site, this decrease can't be caused by the beach nourishment. The dispersion of the data during and following the nourishment is however higher in the impact than in the control site, especially in the intertidal zone, with more coarse samples having a median grain size higher than 300 µm. The regression analysis between median grain size (µm; dependent variable) and Elevation (m versus MLW), Time (t0, t, t1, t2, t3, t4, t5, t6) and Treatment (impact or control site) was done with all possible interaction terms (table 4; Appendices – Chapter 3 – table E). The highly significant interaction term Elevation*Treatment showed an average impact of the nourishment on the sediment grain size distribution. The interaction term Elevation*Treatment*Time was highly significant in all post-nourishment times, except t3 (spring 2011), indicating that the sediment grain size distribution has changed between the two treatments over time. Moreover, these changes become less clear over time (e.g. p-value increase over time), hinting that the sediment grain size distribution might have evolved back to original levels.

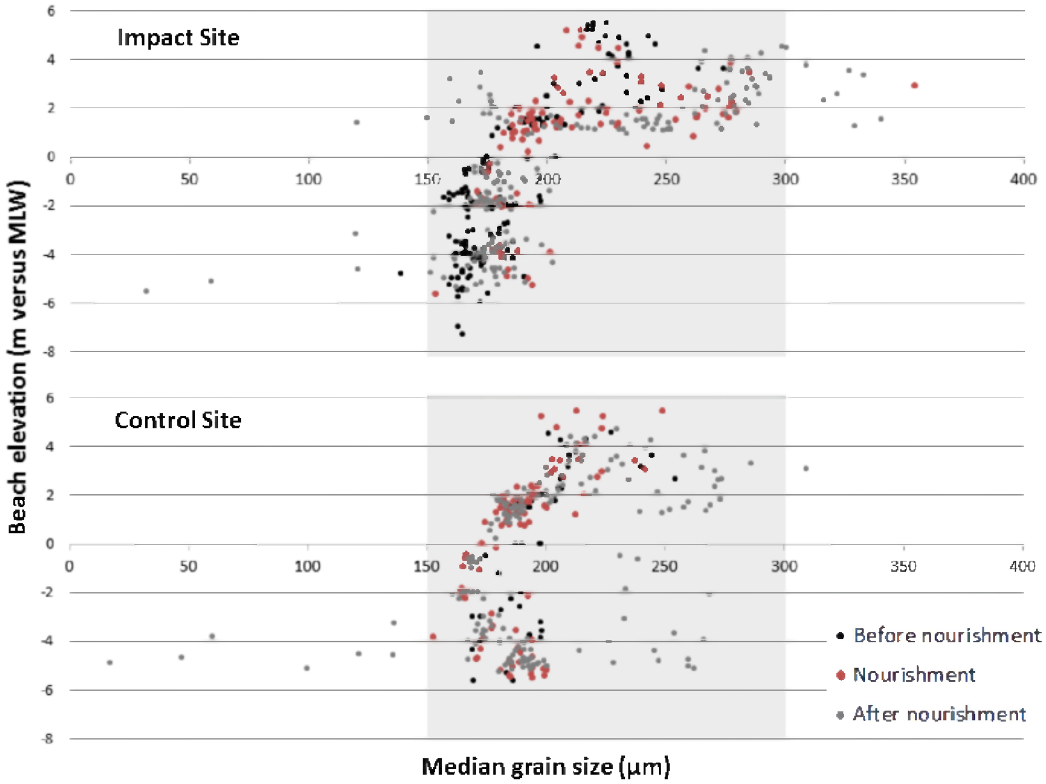


Figure 4: Beach elevation (m versus MLW) versus median grain size (µm) of the impact site (top) and the control site (bottom) before (2008; black), during (2009; red) and after the nourishment (2010, 2011 and 2012: grey); the grey area indicates the normal range of median grain size found on Belgian beaches

Table 3: The interaction terms 'Elevation*Treatment impact' and 'Elevation*Treatment impact*Time' of the regression analysis between median grain size (µm; dependent variable), Elevation (m versus MLW), Time (t0, t, t1, t2, t3, t4, t5, t6) and Treatment (impact or control site) with p-values (significance: $p < 0.05$) (Appendices – Chapter 3 – table E)

Interaction terms	Estimate	SE	t value	Pr(>|t|)
Elevation*Treatment impact	6.0904	1.2027	5.064	**5.41E-07**
Elevation*Treatment impact*Time t0	-3.7051	1.9643	-1.886	0.05972
Elevation*Treatment impact*Time t1	11.8099	2.2811	5.177	**3.04E-07**
Elevation*Treatment impact*Time t2	-11.5074	2.3882	-4.818	**1.82E-06**
Elevation*Treatment impact*Time t3	-3.4478	2.2877	-1.507	0.13229
Elevation*Treatment impact*Time t4	7.4492	2.4568	3.032	**0.00253**
Elevation*Treatment impact*Time t5	4.8311	2.2836	2.116	**0.03478**
Elevation*Treatment impact*Time t6	4.9633	2.3879	2.079	**0.03807**

3.2 Impacts on macrobenthos and recovery

When looking at the intertidal and shallow subtidal abundance data separately, the data did not separate into groups according to treatment (impact and control clusters) or nourishment time (t0 to tx) or a combination of both (figure 5). The interaction terms Treatment*Time and Elevation*Treatment*Time were not included in the final model (table 4). There is however a difference in the zonation patterns of the community structure between the impact and control site (Elevation x Treatment), as well as a difference in time (Elevation* Time).

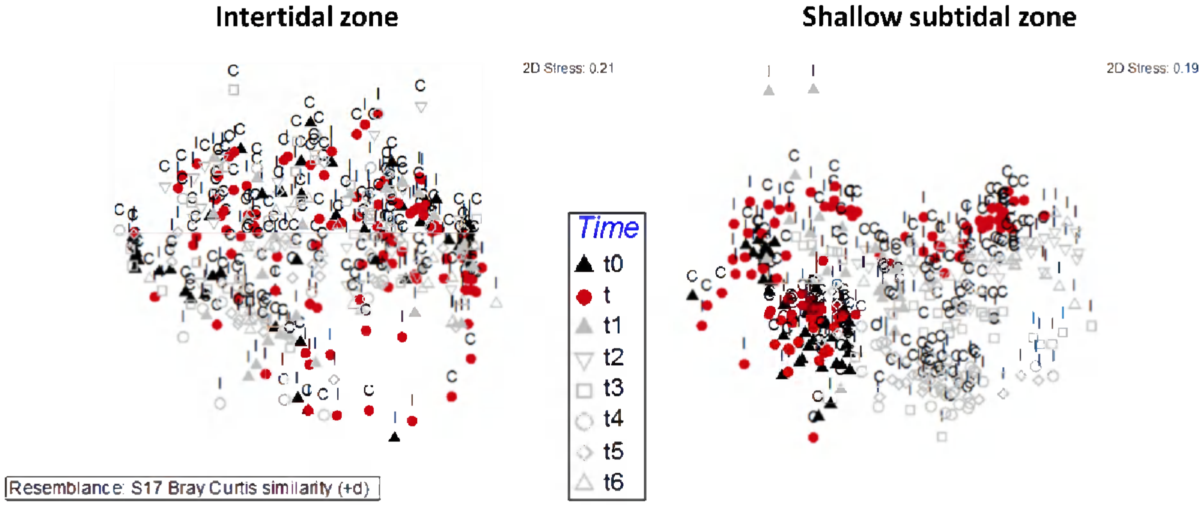


Figure 5: MDS plot of the data per treatment (impact=I and control=C) and per nourishment time (t0=black, t=red and t1=grey; t2 to t6=open grey symbols) in the intertidal and shallow subtidal zone

Table 4: Summary of a permutative linear model based on a Bray-Curtis distance matrix, partitioning multivariate variation in macrobenthic community structure in the intertidal and shallow subtidal zone zone

Main Model	Intertidal zone					Shallow subtidal zone				
	df	SS	Pseudo-F	R^2	P(perm)	df	SS	Pseudo-F	R^2	P(perm)
Elevation	204	85.601	2.442	0.709	**0.001**	253	99.427	2.324	0.808	**0.001**
Treatment	1	0.937	5.455	0.008	**0.001**	1	0.632	3.735	0.005	**0.001**
Time	7	4.846	4.028	0.040	**0.001**	7	9.095	7.682	0.074	**0.001**
Elevation x Treatment	45	12.744	1.648	0.106	**0.001**	28	6.018	1.271	0.049	**0.019**
Elevation x Time	46	12.156	1.538	0.101	**0.001**	27	6.070	1.329	0.049	**0.007**
Residuals	26	4.468		0.037		11	1.861		0.015	
Total	329	120.753		1.000		327	123.103		1.000	

The macrobenthos community structure will be further characterized by its species richness, abundance (number of individuals.m^{-2}) and biomass (g AFDW.m^{-2}) (figure 6). The evolution of the intertidal and shallow subtidal species richness and the shallow subtidal biomass in the impact site is almost identical to the evolution in the control site.

Figure 6 shows lower intertidal mean abundance, species richness and biomass values in the impact site compared to the control site. In autumn 2010 a peak value for abundance was observed in the impact site (121.92 ± 33.68 individuals.m^{-2}) while in both sites, a peak value for species richness was observed (impact site: 6 ± 0.50 species and control site: 7 ± 0.36 species). The intertidal mean biomass on the other hand only exceeded 1 g AFDW.m^{-2} in autumn 2012 (impact site: 1.10 ± 0.10 g AFDW.m^{-2}). The impact site showed another, though smaller, peak value in autumn 2010 (impact: 0.67 ± 0.51 g AFDW.m^{-2}). In autumn 2010, the high mean intertidal abundance, species richness and biomass in the impact zone coincide with the sudden drop in mean median grain size (168.12 ± 4.13 µm).

In the shallow subtidal zone, the mean abundance values before the nourishment (2008) were almost identical in both sites. Since then, there appears to be a difference between the impact and control values. When comparing the two seasons, peak abundance values could generally be observed in autumn, leading to a larger contrast between impact and control values in this season in the years 2009 (control site: 560.78 ± 90.76 individuals.m^{-2}), 2010 (impact site: 839.54 ± 201.25 individuals.m^{-2}) and 2012 (impact site: 844.70 ± 37.65 individuals.m^{-2}), with 2011 as the only exception (218.60 ± 42.55 individuals.m^{-2}). Consistently higher numbers of species were found in the control site, compared to the impact site. Moreover, the species richness increased over time in both sites and this increasing trend did not seem to level off by autumn 2012 (impact site: 16.20 ± 0.39 species and control site: 19.13 ± 0.43 species). An even higher increase in species richness is to be expected. Two mean biomass peak values could be detected in autumn 2010 (impact site: 85.23 ± 16.95 g AFDW.m^{-2} and control site: 59.56 ± 15.24 g AFDW.m^{-2}) and autumn 2012 (impact site: 74.23 ± 2.57 g AFDW.m^{-2} and control site: 54.18 g ± 2.33 AFDW.m^{-2}).

The dominant species in the intertidal and shallow subtidal zone of the impact and control site were determined by taking the five species that contributed the most to the total abundance before, during and after the nourishment (Table 5). The juvenile status of the bivalves *Ensis* species and *Macoma balthica* is clearly stated as they were significantly smaller than the adult individuals. In general, the intertidal zone of the impact and control site was dominated by two amphipod species (*Bathyporeia pilosa* and *Bathyporeia sarsi*), one isopod (*Eurydice pulchra*) and two polychaetes (*Scolelepis squamata* and *Nephtys cirrosa*). The dominance of either one of them shifted over the nourishment periods with the two amphipods taking turns in being the most dominant species. Before and after the nourishment *Bathyporeia pilosa* is clearly the most dominant species while during the nourishment *Bathyporeia sarsi* takes its place, in both the impact and control site. In the shallow subtidal zone of the impact site, *Donax vittatus* and *Nephtys cirrosa* dominate the pre-nourishment situation but their dominance decreased during the nourishment and they were eventually replaced by other dominant species, being *Ensis* juveniles and *Macoma balthica* juveniles. The *Ensis* juveniles also dominate the shallow subtidal zone of the control site during the nourishment so this dominance might be due to natural variation while *Macoma balthica* juveniles only apper dominant in the control site after the nourishment. *Magelona* species was the only species found in the impact site before and after the nourishment and in the control site in all three nourishment periods. *Cirratulidae* species became dominant after the nourishment in both the impact and control site.

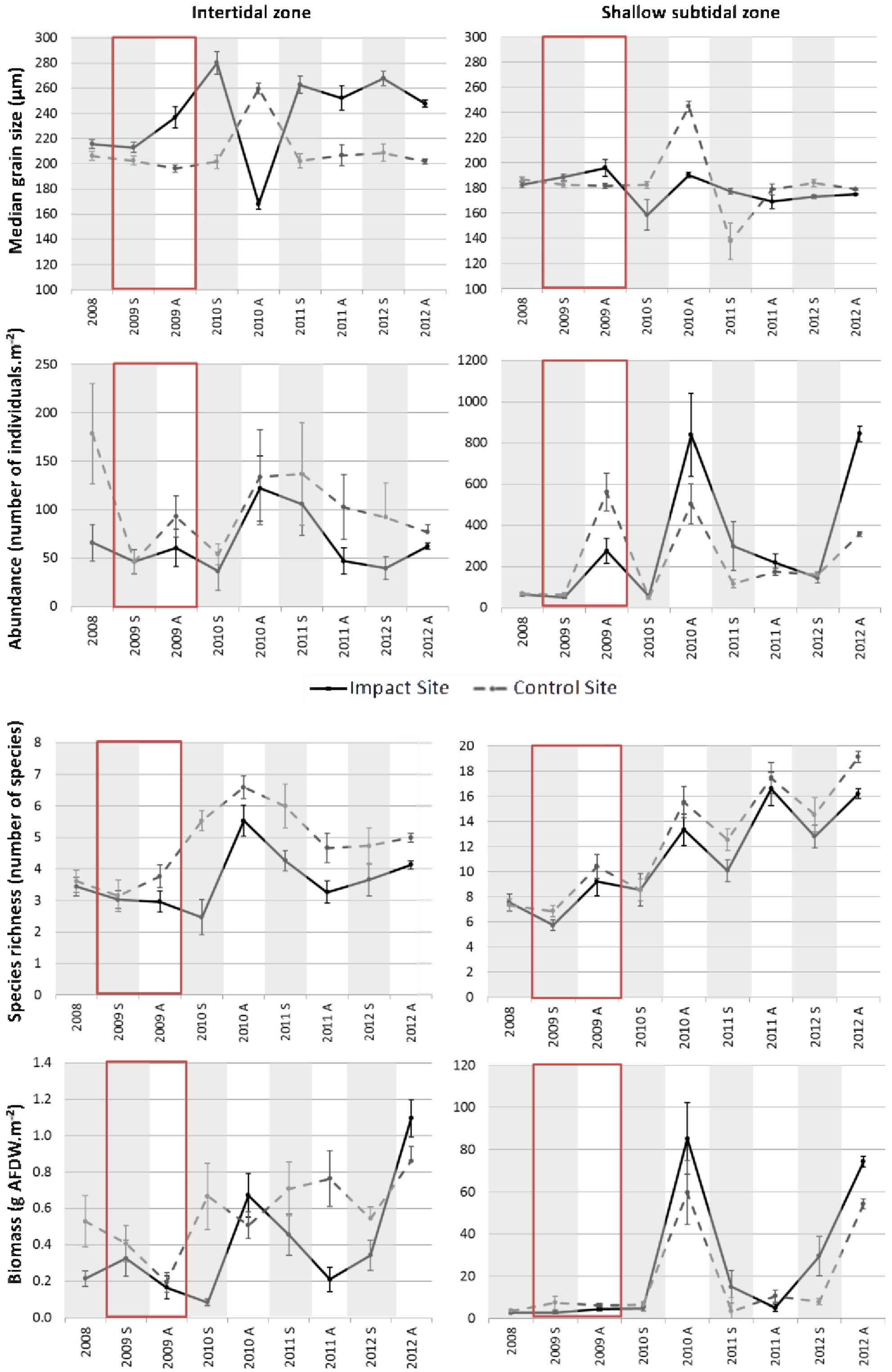


Figure 6: Mean median grain size (µm), species richness (number of species), abundance (number of individuals. m^{-2}) and biomass (g AFDW.m^{-2}) of the impact (black line) and control site (grey dotted line) per year with standard error, for the intertidal (left) and shallow subtidal zone (right) separately

Table 5: The dominant species in the intertidal and shallow subtidal zone of the control and impact site before, during and after the nourishment; the two most dominant species have been highlighted

Impact site	Before	%	During	%	After	%
Intertidal	*Bathyporeia pilosa*	**38.69**	*Bathyporeia sarsi*	**33.33**	*Bathyporeia pilosa*	**34.06**
	Bathyporeia sarsi	**18.35**	*Bathyporeia pilosa*	**28.28**	*Scolelepis squamata*	**23.15**
	Scolelepis squamata	17.64	*Scolelepis squamata*	14.48	*Bathyporeia sarsi*	15.36
	Nephtys cirrosa	7.40	*Nephtys cirrosa*	12.02	*Eurydice pulchra*	9.11
	Eurydice pulchra	7.25	*Eurydice pulchra*	4.51	*Nephtys cirrosa*	5.81
Shallow subtidal	*Donax vittatus*	**56.38**	*Ensis juveniles*	**60.71**	*Macoma balthica juveniles*	**37.48**
	Nephtys cirrosa	**11.60**	*Macoma balthica juveniles*	**20.17**	*Ensis juveniles*	**19.51**
	Magelona species	8.97	*Donax vittatus*	3.51	*Cirratulidae species*	13.81
	Polydora species	7.00	*Magelona species*	2.79	*Owenia fusiformis*	9.67
	Nephtys hombergii	4.67	*Nephtys cirrosa*	2.20	*Macoma balthica*	3.41
Control site	Before	%	During	%	After	%
Intertidal	*Bathyporeia pilosa*	**46.22**	*Bathyporeia sarsi*	**31.67**	*Bathyporeia pilosa*	**31.87**
	Scolelepis squamata	**31.45**	*Bathyporeia pilosa*	**24.78**	*Bathyporeia sarsi*	**26.49**
	Bathyporeia sarsi	10.95	*Scolelepis squamata*	19.34	*Scolelepis squamata*	12.37
	Nephtys cirrosa	3.51	*Ensis juveniles*	5.79	*Eurydice pulchra*	6.62
	Eurydice pulchra	3.45	*Nephtys cirrosa*	4.57	*Nephtys cirrosa*	4.79
Shallow subtidal	*Donax vittatus*	**45.17**	*Ensis juveniles*	**85.89**	*Ensis juveniles*	**41.46**
	Magelona species	**22.44**	*Magelona species*	**2.88**	*Magelona species*	**13.03**
	Polydora species	8.93	*Donax vittatus*	2.46	*Macoma balthica juveniles*	10.17
	Nephtys cirrosa	7.53	*Lanice conchilega*	2.15	*Spio species*	8.80
	Nephtys hombergii	5.02	*Polydora species*	1.27	*Cirratulidae species*	5.33

4. Discussion

4.1 Impacts on habitat characteristics

The nourishment resulted in measurable abiotic changes at the nourishment site. A wider and flatter intertidal beach was created, heightened over its entire width by the added sand (figure 3). No return to the originally lower elevated pre-nourishment profile was visible three years after nourishment. Even though the nourished sand resembled the original sediment very closely with an average median grain size of 224 µm, the intertidal sediment became significantly coarser during and after the nourishment. More intertidal samples had a median grain size higher than 300 µm, during and directly following the nourishment (table 2 and 3; figure 4 and 5).

The intertidal beach is not only closely linked to adjacent beaches that are down-drift but also to the shallow subtidal and offshore zone at the one side and to the dry beach, coastal dunes and hinterland on the other side of the beach ecosystem continuum, through the storage, transport and exchange of sand.

Therefore impacts on beaches have consequences for these adjacent habitats (Speybroeck et al. 2006a; Defeo et al. 2009; Komar 2009). Hence, the shallow subtidal sediments at the impact site did become slightly coarser during the nourishment and t3 (spring 2011). The coarsest median grain sizes were always detected high on the beach (table 3 and figure 4), regardless of the nourishment activities. However, the sediment grain size distribution had changed between the two treatments over time, starting directly after the nourishment (t1) and showing slow recovery in the three following years (tx). The highly significant interaction term Elevation*Treatment in the regression analysis assigns the nourishment as a cause for this impact effect.

Ignoring the effects of latitude and exposure, habitat characteristics define the immediate environment experiences by macrobenthos. An increase in sand particle size on a beach where tide range and wave energy have remained constant results in an irrevocable change in beach state and a decrease in species richness and abundance, at least in the medium term (McLachlan et al. 1996a). Even ecological nourishment introduces slightly coarser sediments (Veloso & Cardoso 2001; Defeo & McLachlan 2005) which might thus lead to a decrease in macrobenthos abundance and species composition. However, the nourished beach had a flatter beach slope over a wider beach and the median grain size stayed within the normal limits found on Belgian beaches (150 – 300 µm) so the effects of the 'ecological' nourishment on the macrobenthos should be small to neglible compared to usual nourishment practice.

4.2 *In situ* impacts on macrobenthos and recovery

The MDS analysis (figure 5) did not show groups according to treatment (impact and control clusters) or time (t0 to t6) nor to a combination of both. Conclusively, no significant effect of nourishment on the macrobenthos community structure could be detected in both the intertidal and shallow subtidal zone (table 4). There was however a difference in the zonation patterns of the community structure between the impact and control site (Elevation*Treatment), as well as a difference in time (Elevation*Time).

Figure 6 gives yet another affirmation of the lack of nourishment effect as it does not reveal any patterns that are clearly caused by the nourishment. The evolution of intertidal mean abundance, species richness and biomass values was almost identical in the impact and control site before, during and after the nourishment, with the values of the nourished site being lower (figure 6). In autumn 2010 peak values were recorded in mean abundance and mean species richness and to a lesser extent in mean biomass, coinciding with a sudden drop and rise in mean median grain size in the impact and control site. High numbers of both *Bathyporeia* amphipods (*B. pilosa*: < 250 µm and *B. sarsi*: 250 – 350 µm) and the polychaete *Scolelepis squamata* (125 – 500 µm) were detected, explaining the peak values in this period (Speybroeck et al. 2007; Van Tomme et al. 2012b).

In the shallow subtidal zone, the impact and control site were almost identical prior to nourishment (figure 5). The mean abundance values always peak in autumn since 2009, leading to a larger contrast between impact and control values in this season, except in 2011. In autumn 2010, peak values at both sites could be detected in mean abundance and mean biomass, coinciding with a sudden rise in mean median grain size in the control site and to a lesser extent in the impact site. The mean species richness was always higher in the control site and increased over time in both sites with no signs of leveling off by autumn 2012 (impact site: 16.20 ± 0.39 species and control site: 19.13 ± 0.43 species). *Donax vittatus* and *Nephtys cirrosa,* both prefering medium coarse sediment, dominated the pre-nourishment shallow subtidal zone but their dominance decreased during the nourishment and they were eventually replaced by other dominant species, being *Ensis* juveniles (natural variation since same pattern can be found in control site) and *Macoma balthica* juveniles (appeared later in the control site) (table 5). These two juvenile bivalves showed peak abundance values in autumn 2010, explaining the overall abundance and biomass peaks in that period. *Cirratulidae* species became dominant after the nourishment in both the impact and control site. Both bivalves and *Cirratulidae* species prefer finer, almost muddy sediments and they appeared when the silt fraction increased in both the impact and control site.

We observed short-term declines in species richness and biomass in the intertidal zone between the start of the nourishment (2009S) and 6 months after its completion (2010S). Literature reviews of usual beach nourishment practice (Nelson 1993; Hackney 1996; Peterson et al. 2000; Speybroeck et al. 2006a; Defeo et al. 2009; Schlacher et al. 2012) report such short-term declines in macrobenthic abundance, biomass and species richness following beach nourishment. Local benthos may be affected by burial and by changing sediment properties (McLachlan 1996; Essink 1999; Van Tomme et al. 2012b) as their limited mobility makes an active escape unlikely, often leading to large mortality (Schlacher et al. 2012). The recovery of the macrobenthic community then relies on the dispersal of benthic macroinvertebrate species of nearby beaches entering the water column and their planktonic larvae (Günther 1992; Cummings et al. 1995), both conditioned by the rate and direction of currents along the coastline (Hill 1991; Defeo 1996). Settlement will depend on the amount of suitable habitat and available space on adequate substratum within the nourished site (Defeo 1996). Connectivity patterns between sandy beach populations linked by larval dispersal are an unexplored field of research, and the mechanisms influencing larval distribution are poorly understood (McLachlan et al. 1996b; Defeo & McLachlan 2005; Schlacher et al. 2008). Long living organisms that don't reproduce successfully each year, e.g. Bivalvia and Echinodermata, need sufficient time to recover from the impact of ususal beach nourishment practice. If no successive nourishment projects are planned, complete recovery can be attained within maximal 4 to 5 years. If the nourishment altered the habitat characteristics too drastically, especially median grain size and beach profile, complete recovery might never take place and a shift in the macrobenthos beach community can be visible.

These results show that at least in some cases nourishment under ecological optimal conditions can show no significant effects in the macrobenthos community structure 6 months after the nourishment (2010S). Within this short-term time frame, the macrobenthos community recovers from the impact of the ecological nourishment, showing no dispersal or recruitment limitations.

Designation of recovery time should as such reflect the long-term biological, ecological and physical processes controlling recolonization and succession (Oliver & Slattery 1976). Complete recovery will only be achieved when the recovered communities resemble those found prior to nourishment. One might consider a benthic community to be recovered when at least 80 % of the species diversity or biomass has been restored (Newell et al. 1998; Essink 1999), bearing in mind that biomass eligibly recovers at a more rapid rate than the species diversity (Adriaanse & Coosen 1991). Caution is warranted as focus on a single parameter to measure recovery can lead to a biased view of the actual recovery. Polychaetes, abundant and cosmopolitan benthic invertebrates, recolonize quickly because of pelagic larval availability, variable food gathering behaviour and the ability to confine activities to the surface layer of the sediment (Hayward 1994; Hartmann-Schröder 1996). Pioneer populations of polychaetes may actually exceed the original populations in numbers of individuals and diversity (Grober 1992). High abundance, species richness and biomass values may as such be attributed to one or a few opportunistic species, misleadingly suggesting recovery.

4.3 Guidelines for beach management and coastal defence policy

A growing awareness of environmentally-sensitive approaches to beach management (Peterson et al. 2000; Speybroeck et al. 2006a; Schlacher et al. 2008; Defeo et al. 2009; Dugan et al. 2010) requires a good knowledge of the ecological effects of beach nourishment, especially of the factors influencing the size of the impact effects (figure 7, based on (Speybroeck et al. 2006a)). In general, the following ecological recommendations for beach nourishment have been formulated: (1) sediment sand should be non-contaminated (Essink 1999) and possess comparable hydraulic properties and characteristics as the original sediment, including clay/silt portion, shell content (Peterson & Manning 2001) and sediment grain size (McLachlan 1996; Hamm et al. 2002), (2) increase in beach slope should be kept as small as possible and the nourished beach profile should resemble the original profile (Short & Wright 1983; Defeo & McLachlan 2005; McLachlan & Dorvlo 2005), (3) nourishment should be done slowly in a sheeting spray of sand and water (rainbow spraying) to allow organisms to keep up with the sediment overburdens as they are applied (Grober 1992), (4) the total amount of nourished sediment should be kept as small as possible (Speybroeck et al. 2006a), limiting the thickness of deposits to 10 cm or less per single application (Schlacher et al. 2012) (5) nourishment should be done during winter season to disturb the reproduction and recruiting cycle as little as possible (Speybroeck et al. 2006a) and (6) several short nourishment projects in time (minimum one week in between) and space (leaving beach strips unnourished) are preferred over broad-scale, long lasting ones, especially in areas where short term morphological changes are unpredictable (Hillen & Roelse 1995).

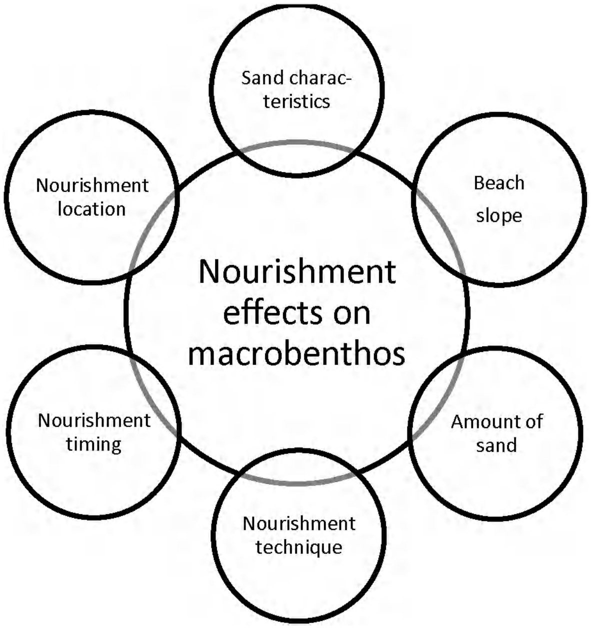


Figure 7: Factors causing ecological impact effects of beach nourishment (Speybroeck et al. 2007)

The nourishment on the intertidal part of the Belgian beach of Lombardsijde was an attempt to perform best-practice ecological beach nourishment. The optimal ecological conditions consisted of (1) creating a gentle sloped beach with at least a 250 m wide wet beach and a dry beach of 50 m wide, (2) using nourished sand with sediment characteristics resembling the original sand as close as possible, (3) establishing a natural transition from dunes to beach, (4) sand strengthening measures in the form of a row of wooden poles, some dune vegetation and fencing, and (5) heightening the harbour dam at the east side of the eastern palisade of Nieuwpoort to prevent the nourished sediment from relocating into the alongside harbour inlet. Unfortunately, the nourishment was postponed to spring and summer 2009 since winter 2008 – 2009 was a heavy storm season (October 15 – April 15) inducing large sand losses.

Nourishment under these conditions has no detectable impact effects on both the intertidal and shallow subtidal beach ecosystem, in particular the macrobenthos. Ecological nourishment thus proves to be an effective way of combating erosion while being less ecologically damaging than any other coastal engineering activity (Peterson et al. 2000; Speybroeck et al. 2006a; Schlacher et al. 2008; Defeo et al. 2009). Alternative locations for nourishment have been suggested though. For instance, nourishing in the shallow subtidal zone would result in a slow distribution of the nourished sand across the intertidal beach by hydrodynamic transport. It is likely that .msubtidal macrobenthos have a greater and more mobile pool of animals to supply recruits from which recovery can occur. However, the environmental effects of this strategy are unknown (Schlacher et al. 2012) and caution should be taken as the shallow subtidal is a known nursery area for juvenile fish and epibenthos (Beyst et al. 1999a; Beyst et al. 2001b).

Other approaches to battle inundation, scour and erosion include restoring and supporting the natural development of sand dunes or creating natural buffer zones between the sea and the area at risk, e.g. coastal communities. A combination of soft and hard coastal defence structures, where appropriate, has been proposed as well. In some coastal areas, this seems to be the most sensible approach, especially when vast efficient hard structures are already in place. Looking at the Belgian coastal zone, it would be feasible to keep nourishing the touristic, (semi-)urbanized, top priority coastal defence beaches up to a high protective level as long as enough beach zones are protected in an adequate manner. These 'pristine' beaches could then evolve into healthy beach ecosystems and provide a stock of animals for recolonisation when those sacrificed beaches do become viable again. Retreat and relocation are still the most ecologically favorable option. The shoreline is then left to erode, while buildings and infrastructure are relocated further inland. Resistance to regulation and the economic value of coastal communities currently prohibits the establishment of sound long-range retreat policies (Schlacher et al. 2008; McLachlan et al. 2013).

5. Conclusion

The nourishment on the intertidal part of the Belgian beach of Lombardsijde was an attempt to perform best-practice ecological beach nourishment. A wider, higher and flatter intertidal beach with coarser sediment was created and no return to the pre-nourishment conditions was visible three years after nourishment. The sediment grain size distribution had changed as well, showing slow recovery in the three post-nourishment years. The analysis of the macrobenthos community structure shows that at least in some cases nourishment under ecological optimal conditions can show no significant effects on both the intertidal and shallow subtidal beach ecosystem 6 months after the nourishment. Within this time frame, the macrobenthos community recovered from the impact of the ecological nourishment. Ecological nourishment thus proves to be the least ecologically damaging way of combating erosion, compared to all other coastal engineering activities.

Chapter 4: Macrofaunal sediment selectivity considerations for beach nourishment programmes

Joke Van Tomme, Sarah Vanden Eede, Jeroen Speybroeck, Steven Degraer, Magda Vincx (2012) Macrofaunal sediment selectivity considerations for beach nourishment programmes.

Published in: Marine Environmental Research 84:10-16

Joke Van Tomme is the first author of this article. Sarah Vanden Eede helped with the experimental work, the analysis of the results and the finalisation of the manuscript.

Abstract

Nowadays, beach nourishment is widely considered as a better alternative compared to the construction of hard structures to protect a sandy coast against detrimental erosive effects, both from an ecological and an engineering perspective. The rare studies conducted on the ecological impact of beach nourishment are short-term, *post hoc* monitoring investigations of the benthic macrofauna. Little is known of the biological processes during and after nourishment. To allow swift recolonization after nourishment, the characteristics of the nourished beach have to match the habitat demands of the benthic macrofauna. The sediment preference of the key intertidal species *Scolelepis squamata, Eurydice pulchra, Bathyporeia pilosa* and *Bathyporeia sarsi*, which dominate many West European sandy beaches, was investigated through laboratory experiments, both in single-species as well as combined-species treatments. While the former aimed at developing guidelines for impact mitigation of beach nourishment, the latter aimed at elucidating the role of biotic interactions in sediment preference. Results of the experiments indicated that *Bathyporeia pilosa* and *Eurydice pulchra* prefer the finest sediment, while *Bathyporeia sarsi* had a broader preference and also occurred in medium-coarse sediments. However, the sediment preference of *Eurydice pulchra* for fine sediments was not confirmed by other field and experimental studies. The polychaete *Scolelelpis squamata* had the broadest preference and even showed a high occurrence in coarse sediments that are not naturally occurring on the sandy beaches where the animals were caught for this experiment. However, this polychaete is a cosmopolitan species, not only occurring on fine-grained beaches, but also on coarse-grained beaches worldwide. The preferences imply that beach nourishment with coarse sediment will have a major effect on *Bathyporeia pilosa* while effects of coarse sediments on *Scolelepis squamata* will be minor. Finally, interspecific competition with the sympatrically occurring amphipod *Bathyporeia sarsi* was found to change the sediment selection of the amphipod *Bathyporeia pilosa* towards the coarser sediments where *Bathyporeia sarsi* occurred in lower frequencies.

Keywords: beaches, benthos, beach nourishment, environmental impact, sediment, macrobenthos, ecosystem management

1. Introduction

Beach nourishment is an episodic, dramatic event for the sandy beach ecosystem with diverse impacts (Speybroeck et al. 2006a) both on organisms inhabiting the beach (Jones et al. 2008; Leewis et al. 2012; Schlacher et al. 2012) as well as on adjacent ecosystems (Jordan et al. 2010). However, effects depend on a variety of diverse characteristics of the specific nourishment programme. The choice between high-shore, foreshore or profile nourishment greatly determines what species communities on the beach are influenced while the frequency between different nourishment projects is essential for the recovery of the system. In addition, the timing of the nourishment deserves careful consideration to maximally avoid periods of breeding or recruitment of different sandy beach organisms (Melvin et al. 1991; Peterson et al. 2000; Peterson & Manning 2001). As beach nourishment, supplying several tons of sediment on the intertidal beach, does not allow any survival of macrobenthic infauna (Schlacher et al. 2012), attempts to bring the post-nourishment beach back to pre-nourishment ecosystem conditions, have to address post-impact restoration. Two major process-related elements seem to be of importance for swift recolonization: (1) dispersal capacities and (2) habitat demands of the species. The first aspect is related to species-specific characteristics, albeit local geography and hydrodynamics of the area surrounding the nourished beach will play an important role. Large anthropogenic structures like harbor walls may hamper long-shore drift of pelagic larvae and water column dispersal of subadult and adult organisms. Once the nourished beach has been reached, animals will have to be able to settle, burrow and survive. All this will depend on their specific tolerances and preferences, in relation to the encountered habitat. Although peer-reviewed studies on the effect of beach nourishment are scarce (Jones et al. 2008; Leewis et al. 2012; Schlacher et al. 2012), several studies have investigated the effects after dredging (Somerfield et al. 1995; Radenac et al. 1997; Savage et al. 2001; Byrnes et al. 2004; Diaz et al. 2004; Witt et al. 2004; Powilleit et al. 2006), thereby demonstrating that benthic macrofauna frequently show changes in abundance, species richness and community structure. The negative effects may either be small, with a short period of recovery in some regions (Van Dolah et al. 1984; Radenac et al. 1997; Roberts & Forrest 1999), since macrofauna of dynamic coastal zones is tolerant to disturbances (Newell et al. 1998), or may be highly important, permanently altering the macrobenthic association (Harvey et al. 1998). Structural damages on the macrofauna may occur due to changes in the granulometric characteristics, since the macrofauna composition is closely related to the sediment characteristics (McLachlan 1996; Brazeiro 2001; McLachlan & Brown 2006). Indeed, sediment composition is a major controlling factor for changes in benthic associations within the constraints of the adjacent species pool as it is directly linked to the organic matter content (food availability) which is one of the important factors in determining trophic complexity and species abundances (Knox 2001; Incera et al. 2006; Rodil et al. 2012). However, sediment organic matter is not the only structuring factor and other factors such as the beach morphodynamics also have an important role in structuring sandy beach communities (McLachlan & Jaramillo 1995; McLachlan 1996; Defeo & McLachlan 2005). Recent studies show that both physical as well as nutritional variables are important for the sandy beach community structure (Incera et al. 2006; Cisneros et al. 2011). Therefore, information on the responses of macrobenthic species on changing sediment characteristics is one of the crucial elements to assess the impact of beach nourishment on the macrobenthic community. Unfortunately, experimental studies on sediment preferences of sandy beach species are scarce and existing studies only examine sediment selection of higher trophic species such as flatfish (Gibson & Robb 2000; Nasir & Poxton 2001; Carl et al. 2008) while studies on the preferences of macrobenthos are rare (Speybroeck 2007).

Since profile beach nourishment mostly affects the high-intertidal beach as large amounts of sediment are first placed on the high shore and are than divided by bulldozers over the entire beach (Hanson et al. 2002), we examined the sediment preferences of the key macrobenthic species of the high-intertidal *Scolelepis squamata* – *Eurydice pulchra* community of the Belgian beach ecosystem (Van Hoey et al. 2004). Belgian beaches are characterized by gentle slopes and fine sediments and are thus generally considered to be dissipative (Degraer et al. 2003b). The selected species of the high-intertidal community of these dissipative beaches were the polychaete *Scolelepis squamata*, the isopod *Eurydice pulchra* and the two amphipods *Bathyporeia pilosa* and *Bathyporeia sarsi*. *Scolelepis squamata* is a suspension feeding polychaete (Dauer 1983) while the amphipods feed on epipsammic diatoms attached to the sand grains (Nicolaisen & Kanneworff 1969). The isopod *Eurydice pulchra* is an aggressive and very mobile predator, feeding on polychaetes and crustaceans such as *Bathyporeia pilosa*, *Bathyporeia sarsi* and *Scolelepis squamata* (Jones 1968).

The aims of this study, investigating the sediment selection of sandy beach macrobenthos of dissipative sandy beaches, were (1) to examine the sediment preference of the four dominant macrobenthic species (*Scolelepis squamata, Eurydice pulchra, Bathyporeia pilosa* and *Bathyporeia sarsi*) of these beaches to formulate valuable recommendations for the used sediment in beach nourishment projects and (2) to study the effect of interspecific interactions in influencing this choice.

2. Material and methods

2.1 Experimental design

Sediment preference was examined for the four species, both in single-species and combined-species conditions. Besides the single-species treatments, several two-, three- and four-species combinations were experimentally studied (table 1) during the summer of 2011. Due to the high number of two- and three-species combinations possible, only the two-species combinations between species with different trophic positions or between possible competitors were tested. As the polychaete and the amphipods are known to feed on different food sources (Nicolaisen & Kanneworff 1969; Dauer 1983), polychaete-amphipod combinations were thus not tested. Furthermore, only a limited number of three-species combinations were tested as the results of these treatments could not unequivocally indicate what species was the most influencing for possible preference changes. The experiment was conducted in a climate room at 19° C, the summer temperature on Belgian sandy beaches, in a natural summer dark/night regime (16:8 h light/dark).

Table 1: Sediment preference treatments. Single-species (column 1) and combined-species treatments (column 2-4) where sediment preferences were tested for

Single-species treatment	2-species treatment	3-species treatment	4-species treatment
Bathyporeia pilosa	*Bathyporeia pilosa – Bathyporeia sarsi*	*Bathyporeia pilosa – Bathyporeia sarsi – Scolelepis squamata*	*Bathyporeia pilosa – Bathyporeia sarsi – Scolelepis squamata – Eurydice pulchra*
Bathyporeia sarsi	*Eurydice pulchra – Bathyporeia pilosa*		
Scolelepis squamata	*Eurydice pulchra – Bathyporeia sarsi*		
Eurydice pulchra	*Eurydice pulchra – Scolelepis squamata*		

The experimental organisms were released into round-shaped aquaria (cross-section = 30 cm; h = 10 cm), subdivided into four quarters by metal partitions which prevented movement between sections via the sediment. Each section was covered with a layer of one of the four different sediment types, either naturally occurring on sandy beaches or used in current and future beach nourishment projects (fine: 125 – 180 µm; medium-fine: 180 – 250 µm; medium-coarse: 250 – 355 µm; coarse (outside the range of sediments naturally occurring on the beaches considered in this study): 355 – 500 µm). Each species treatment was replicated five times. Sediment depth was 4 cm and the seawater depth on top of the sediment was 5 cm. Sediments remained submerged throughout the experiments, ruling out desiccation of experimental specimens. During the 48 hour experiment, the aquaria were constantly aerated but no food was added since experimental time was limited. Experiments were started at low tide and animals were released at random into the aquarium by pouring the organisms (submerged in a small amount of sea water) in a circular movement over the four subdivisions. As the experiment started at low tide when most species stay buried, a time lag of 15 minutes was respected after addition of the first species before adding the next species to allow every species to bury in the sediment. After 48 hours (ensuring several swimming cycles of the species at high tide), the experiment was terminated and all living individuals were extracted from each section and counted. During several subsequent weeks from May to July 2011, all species combinations were examined each time using new experimental organisms.

2.2 Collection of organisms, sand and sea water

Beach sediment was collected at the beach of De Panne (Belgium; 2°33'24" E 51°05'42" N) and after removal of organic matter by heating the sediment up to 450°C, the sand was sieved over a sequence of sieves with mesh width of 125 µm, 180 µm, 250 µm, 355 µm and 500 µm. The sea water, originating from the same Belgian beach, was filtered over a 45 µm filter to remove all fauna from the water.

All organisms were collected by sieving the beach sediment on the high-intertidal beach in De Panne. In the experimental treatments, natural densities of the macrobenthic species were used that ensured enough encounters to force active selection between sediment types (Speybroeck 2007): 150 individuals/treatment (=2125 ind.m^{-2}) for *Bathyporeia pilosa*; 70 individuals/treatment (=991 ind.m^{-2}) for *Bathyporeia sarsi;* 20 individuals/treatment (=284 ind.m^{-2}) for *Scolelepis squamata* and 10 individuals/treatment (=143 ind.m^{-2}) for *Eurydice pulchra.*

In the multi-species treatments, total species densities were higher than in the single-species treatments, but as this actually reflects the field situation, this was expected to give valuable results. Indeed, the zonation patterns of the high-intertidal macrobenthos species show overlap (Degraer et al. 2003a; Degraer et al. 2003b), resulting in a higher overall species abundance on the beach. Before the start of the experiment, species stocks were left overnight to allow acclimatization of the experimental organisms.

2.3 Data analysis

The distribution of species was tested with a replicated G-test of goodness-of-fit (Sokal & Rohlf 1995; Stoner & Ottmar 2003). This test was used to examine whether the species showed a random distribution over the four sediment types offered. The null hypothesis states that the number of observations in each sediment is equal to the expected distribution, i.e. as a random distribution is hypothesized, the number of observations in each sediment type should be equal. The replicated G-test of goodness-of-fit has the advantage that the null hypothesis can be tested for each individual experiment (partial G's) but also for the pooled data set (Sokal & Rohlf 1995). Heterogeneity G(Gh) (with [no. of replicates – 1] × [no.of sediment types– 1] degrees of freedom) was calculated to assess heterogeneity among replicate treatments. Pooled G (Gp) (with no. of sediment types –1 degrees of freedom) tested the goodness of fit for the pooled data over all experimental replicates, and Gt, the sum of Gh and Gp (with [df Gh] + [df Gp] degrees of freedom) tested whether the data as a whole fitted the expected distribution. In the combined-species treatments, the same G- test was used against the null hypothesis that species distribution was similar to the species distribution in the single-species experiments.

The sediment selectivity was estimated by the Electivity index, E'. E' is calculated per sediment type as: E'= $(c_i\text{-}o_i)/(c_i + o_i)$ where c_i is the species abundance in one sediment type and o_i the expected abundance, in case of random distribution, for that sediment (Ivlev 1961). Positive E' values indicate a preference, negative ones a rejection (Hiddink et al. 2002).

3. Results

3.1 Single species treatments

The results of the G-test showed a significant sediment preference for all tested species (table 2 and 3). In detail, *Bathyporeia pilosa* clearly preferred the finer sediments since 87 % of the experimental population of this amphipod was found in the sediments with a grain size smaller than 250 µm (figure 1A and table 2). As 42 % of the experimental population of the related amphipod *Bathyporeia sarsi* was found in the sediment types with a grain size larger than 250 µm, *Bathyporeia sarsi* had a broader preference (figure 1B and table 2). *Scolelepis squamata* was more divided over finer and coarser sediments, 30 % of these polychaetes was even found in sediment with a grain size larger than 355 µm (figure 1C) and table 2), whereas for *Eurydice pulchra* the sediment preference resembled the preference of *Bathyporeia pilosa* (figure 1D and table 2). The results of the G-tests for goodness of fit showed that replicates were heterogeneous for *Bathyporeia pilosa* and *Eurydice pulchra*. Nevertheless, the partial G's were highly significant ($p < 0.001$).

3.2 Combined species treatments

Sediment preferences of all tested macrobenthic species differed significantly between single-species and combined species conditions (table 2). Although replicates were heterogeneous for all tested species, the partial G's were highly significant ($p < 0.001$). In the presence of *Eurydice pulchra*, the *Bathyporeia pilosa* frequency of occurrence in the fine and medium-fine sediment decreased, while the frequency of occurrence in the medium-coarse sediment increased from 11 ± 1 % to 22 ± 5 % (figure 1A). In the presence of the related amphipod *Bathyporeia sarsi*, the *Bathyporeia pilosa* frequency of occurrence in the medium-fine sediment decreased from 45 ± 3 % to 25 ± 4 %, while the frequency of occurrence of *Bathyporeia pilosa* in the two coarsest sediments increased (figure 1A). In the 3-species treatment, the *Bathyporeia pilosa* frequency of occurrence in the medium-fine sediment decreased to 28 ± 3 %, while the frequency of occurrence in the medium-coarse sediment increased (figure 1A) and in the 4-species treatment, there was a decrease of *Bathyporeia pilosa* frequency of occurrence in the fine sediment, while there was an increase in the medium-coarse and coarse sediments (figure 1A).

In the presence of *Eurydice pulchra*, a strong increase of *Bathyporeia sarsi* from 18.94 ± 1.93 % to 42.05 ± 13.36 % was observed in the medium-fine sediment, while a decrease was found in the fine and coarse sediments (figure 1B). In the presence of *Bathyporeia pilosa*, the sediment preference of *Bathyporeia sarsi* changed only slightly (figure 1B). The polychaete *Scolelepis squamata* showed a significant increase in the fine sediment from 13.33 ± 4.16 % in the single-species treatment to 29.23 ± 4.10 % in the 3-species treatment and even 34.16 ± 8.48 % in the 4-species treatment (figure 1C). Isopod frequency of occurrence increased in the coarse sediment from 13.11 ± 4.19 % to 27.56 ± 7.58 % and to 23.00 ± 10.20 % in the 3- and 4-species treatments respectively (figure 1D).

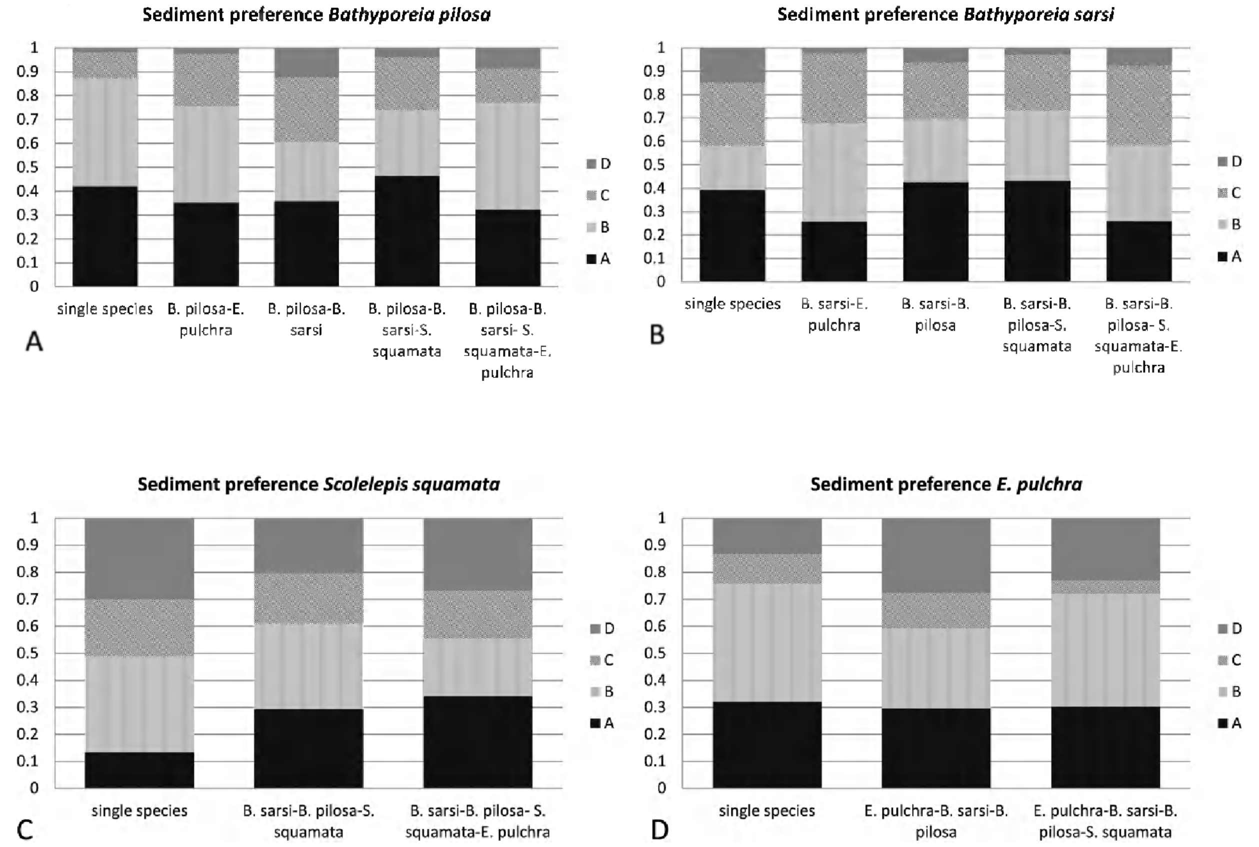


Figure 1: Sediment preference of Bathyporeia pilosa (A), Bathyporeia sarsi (B), Scolelepis squamata (C) & Eurydice pulchra (D) in single-species- and combined-species conditions. X-axis: species treatments; Y-axis: average proportion of the experimental population in sediment types: A: 125 – 180 µm; B: 180 – 250 µm; C: 250 – 355 µm; D: 355 – 500 µm

Table 2: G-test results of the single-species and combined-species treatments of *Bathyporeia pilosa, Bathyporeia sarsi, Scolelepis squamata* and *Eurydice pulchra*

Bathyporeia pilosa	**Gt *(df: 15)***	**p(Gt)**	**Gh *(df: 12)***	**p(Gh)**	**Gp *(df: 3)***	**p(Gp)**
B. pilosa (single species treatment)	435.31	< 0.001	56.85	< 0.001	378.47	< 0.001
B. pilosa - B. sarsi - S. squamata - E. pulchra	290.79	< 0.001	183.71	< 0.001	107.08	< 0.001
B. pilosa - B. sarsi - S. squamata	1008.08	< 0.001	883.89	< 0.001	124.18	< 0.001
B. pilosa - B. sarsi	128.87	< 0.001	57.14	< 0.001	71.73	< 0.001
E. pulchra - B. pilosa	108.01	< 0.001	84.06	< 0.001	23.95	< 0.001
Bathyporeia sarsi	**Gt *(df: 15)***	**p(Gt)**	**Gh *(df: 12)***	**p(Gh)**	**Gp *(df: 3)***	**p(Gp)**
B. sarsi (single species treatment)	24.71	0.054	7.59	0.82	17.13	< 0.001
B. pilosa - B. sarsi - S. squamata - E. pulchra	225.62	< 0.001	171.58	< 0.001	54.04	< 0.001
B. pilosa - B. sarsi - S. squamata	100.13	< 0.001	32.91	< 0.001	67.22	< 0.001
B. pilosa - B. sarsi	2331.39	< 0.001	59.79	< 0.001	2271.59	< 0.001
E. pulchra - B. sarsi	276.71	< 0.001	172.83	< 0.001	103.87	< 0.001
Scolelepis squamanta	**Gt *(df: 15)***	**p(Gt)**	**Gh *(df: 12)***	**p(Gh)**	**Gp *(df: 3)***	**p(Gp)**
S. squamata (single species treatment)	20.19	0.16	9.32	0.68	10.88	0.012
B. pilosa - B. sarsi - S. squamata - E. pulchra	107.22	< 0.001	77.93	< 0.001	29.29	< 0.001
B. pilosa - B. sarsi - S. squamata	92.04	< 0.001	39.83	< 0.001	52.21	< 0.001
Eurydice pulchra	**Gt *(df: 15)***	**p(Gt)**	**Gh *(df: 12)***	**p(Gh)**	**Gp *(df: 3)***	**p(Gp)**
E. pulchra (single species treatment)	61.23	< 0.001	47.26	< 0.001	13.97	0.0029
B. pilosa - B. sarsi - S. squamata - E. pulchra	88.11	< 0.001	64.13	< 0.001	23.98	< 0.001
B. pilosa - B. sarsi - E. pulchra	43.08	< 0.001	34.95	< 0.001	8.13	0.043

Table 3: Sediment selectivity based on the Electivity index

	125 – 180 µm	180 – 250 µm	250 – 355 µm	355 – 500 µm
Bathyporeia pilosa	+	+	-	-
Bathyporeia sarsi	+	-	+	-
Eurydice pulchra	+	+	-	-
Scolelepis squamata	-	+	+	+

4. Discussion

4.1 Species sediment preference

The preference of *Bathyporeia pilosa* for the two sediment types with a grain size smaller than 250 µm, is in line with observed field preferences of this amphipod for sediment with a median grain size smaller than 250 µm and even smaller than 210 µm (Vader 1965; Vader 1966; Khayrallah & Jones 1980; Persson 1982; Nicolaisen & Kanneworff 1983). The field sediment preference of *Bathyporeia sarsi* for somewhat coarser sediment (Vader 1965) was also confirmed in this experimental study. While a previous study by Jones (1969) found a preference for coarser sediments, the isopod *Eurydice pulchra* preferred fine sediment in the current study. Since the pattern was found both in combined-species as well as in single-species conditions, the presence of prey species in the finer sediments could not explain this behavior. As *Eurydice pulchra* is a highly energetic swimmer (Alheit & Naylor 1976), the preference for the finer sediment is likely to have been an active choice. The differences between studies are remarkable and differing experimental conditions can be an important cause. However, a former experimental study in the same laboratory and under similar experimental conditions as the current study showed a preference for coarse sediment (Vandewalle 2009). The only clear difference between these studies is the origin of the experimental organisms. While the species used in this study were gathered on the dissipative beach of De Panne, the used species in the study of Vandewalle (2009), were collected on the dissipative beach of Raversijde but sediment did not differ significantly between these two beaches. The statistical analysis of this study did however indicate that replicates were heterogeneous and this can hamper a clear interpretation of the sediment preference. Hence, the sediment preference of *Eurydice pulchra* might have been less specific than for other sandy beach species and a broad tolerance could be suggested for the isopod. This conclusion is supported by the cosmopolitan occurrence of *Eurydice pulchra*, both on fine-grained dissipative beaches (Degraer et al. 2003b) as well as on coarse-grained reflective beaches (Rodil et al. 2006).

The most striking result in this study was the preference of the polychaete *Scolelepis squamata* for both medium-fine as well as coarse sediment, also found by Speybroeck (2007). While this spionid polychaete inhabits fine to medium sediments on West European dissipative beaches (Degraer et al. 2003b; Janssen & Mulder 2005), it is a rather cosmopolitan species inhabiting both fine-grained as well as coarse-grained sediments (Dahl 1971; Hartmann-Schröder 1996; Van Hoey et al. 2004), which is in accordance with the results of our experiments.

4.2 Recommendations for beach nourishment of West European sandy beaches

Although differences were found between preferences in single-species and combined-species conditions, general recommendations for nourishment could be made based on the results of this study. All studied species preferred sediment with a median grain size smaller than 250 µm (figure 2). Sediment with a median grain size between 250 µm and 355 µm negatively influenced the presence of the amphipod *Bathyporeia pilosa* and the isopod *Eurydice pulchra*, while coarse sediment (355 – 500 µm) negatively influenced all species except the polychaete *Scolelepis squamata* (figure 2).

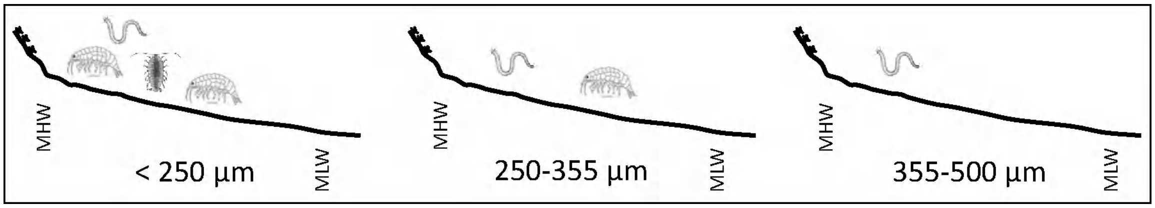


Figure 2: Hypothetical high-intertidal macrobenthos community after beach nourishment using three sediment types

The results of this experimental study on sediment preferences of the most dominant species of dissipative sandy beaches do not immediately imply field mortality or a decrease in field recruitment when the habitat is altered due to nourishment projects. However, observations and monitoring after nourishment are showing that when the habitat of sandy beaches is altered towards less favorable conditions, some species do not recolonize the nourished beach or only recolonize the beach in lower abundances after several months (Schlacher et al. 2012). As the intertidal sandy beach environment is a dynamic habitat and sandy beach animals are very mobile, they are likely to avoid those habitats that do not satisfy their preferences.

While other factors like beach profile, inundation time and organic matter are also important in determining the outcome of a nourishment, repeated beach nourishment projects with coarse sediments will inevitably lead to habitat loss for macrobenthos on dissipative beaches, especially for those species preferring fine sediments like *Bathyporeia pilosa*. As a result, the macrobenthos diversity and abundance will decrease and beaches will in essence be inhabited by extremely opportunistic species like the polychaete *Scolelepis squamata* (figure 2) as was also found after dredging events (Rosa & Bemvenuti 2006). This polychaete will probably suffer least from nourishment events as it can quickly recolonize nourished beaches due to their pelagic larvae, and will not suffer from the presence of coarse sediment. In addition, when nourishment projects are characterized by coarse sediment and steep slopes, there is a risk of not only decreasing biodiversity but also of causing entire community shifts. Indeed, macrobenthos communities in flat, fine-grained dissipative beaches differ greatly from communities in coarse-grained, steeper reflective beaches (McLachlan 1990; Defeo et al. 1992; Defeo & McLachlan 2011) and the alternation of the morphodynamics of a beach may thus lead to community shifts. For the West-European dissipative beaches this evolution would cause an important loss of biodiversity since dissipative beaches are known to be richer than reflective ones (McLachlan et al. 1996a).

4.3 Sediment preferences and species interactions on sandy beaches

Examining biotic interactions by sediment selection experiments is an indirect approach (Dugan et al. 2004), but previous research has shown its merit (Defeo et al. 1997). Hence, the results of this sediment selection experiment can give insights in the role of biotic interactions on dissipative sandy beaches. *Bathyporeia pilosa* significantly changed its sediment preference towards the coarser sediments, where densities of the related amphipod *Bathyporeia sarsi* were lower in combined-species conditions. These changes seemed to be steered by interspecific competition with *Bathyporeia sarsi*. Adversely however, *Bathyporeia sarsi* did not seem to actively avoid *Bathyporeia pilosa* and was thus probably not affected by competition of *Bathyporeia pilosa*. Since former experiments on competition between the co-occurring amphipods *Bathyporeia pilosa* and *Bathyporeia sarsi* were not decisive on the role of interspecific competition (Van Tomme et al. 2012a), this sediment selection experiment could gain a better insight into their segregated zonation pattern on the intertidal beach (Speybroeck et al. 2008b). Interspecific competition usually has asymmetric effects (Connell 1983; Schoener 1983), especially in the marine intertidal zone, with larger species being competitively dominant (Paine 1980; Schoener 1983; Brown & Maurer 1986; van Riel et al. 2007). In this study, the competitive superiority of the largest amphipod, *Bathyporeia sarsi* (Speybroeck et al. 2008b), was indicated, suggesting that asymmetric interspecific competition can play a structuring role on dissipative sandy beaches.

Predation by the predator *Eurydice pulchra* could also be hypothesized to be an important factor in influencing species distribution on sandy beaches. In combined-species treatments where the predator *Eurydice pulchra* was present, a clear avoiding behavior could be inferred from the data since the amphipods and especially *Bathyporeia pilosa* moved to sediments with the lowest density of *Eurydice pulchra*.

Finally, it was clear that the amphipod *Bathyporeia pilosa* was suffering most from biotic interactions and this could explain its small realized niche on the high-intertidal sandy beach. Although the morphology of the co-occurring *Bathyporeia sarsi* is not highly different at first sight, competition and predation did not seem to have a clear effect on the behavior of this larger amphipod (Speybroeck et al. 2008b), as could be reflected in its occupancy of a wider zone on the beach compared to *Bathyporeia pilosa* (Speybroeck et al. 2008b).

5. Conclusions

The results of this sediment selection experiment show that while the amphipods *Bathyporeia pilosa* and *Bathyporeia sarsi* were preferring fine to medium-fine sediment, the opportunistic polychaete *Scolelepis squamata* preferred coarse sediment. The isopod *Eurydice pulchra* preferred fine sediment but these results were not in accordance with former field and experimental studies. Additionally, interspecific competition with the sympatrically occurring amphipod *Bathyporeia sarsi* was found to change the sediment selection of the amphipod *Bathyporeia pilosa* towards the coarser sediments where *Bathyporeia sarsi* occurred in lower frequencies.

To mitigate the impact of beach nourishment projects on intertidal sandy beaches and to assure a swift recolonization of the nourished beach by the original sandy beach community, the use of sediment that resembles the initial beach sediment, is therefore strongly encouraged. The use of coarse sediments is likely to have a negative effect on some of the dominant macrobenthic species of the high-intertidal on fine-grained beaches. Therefore, both technical as well as ecological aspects of the sandy beach ecosystem should be considered in beach nourishment programmes to assure its highly valuable ecosystem role.

Acknowledgments

The first author acknowledges an aspirant grant provided by FWO-Vlaanderen (Belgium). Additional funding was provided by the Special Research Fund of the Ghent University (BOF-GOA 01GA1911W). We also want to thank all the helping hands in the field for the collection of the test organisms.

Chapter 5: Assessing the impact of beach nourishment on the intertidal food web through the development of a mechanistic-envelope model

Sarah Vanden Eede (*), Joke Van Tomme (*), Charlotte De Busschere, Martijn Vandegehuchte, Koen Sabbe, Eric Stienen, Steven Degraer, Magda Vincx & Dries Bonte (in prep.) Assessing the impact of beach nourishment on the intertidal food web through the development of a mechanistic-envelope model.

(*) = equal contribution

In preparation for submission to Journal of Applied Ecology

Sarah Vanden Eede and Joke Van Tomme worked together on this chapter in equal measure from the starting idea to the first complete draft (as included in the PhD of Joke Van Tomme). Sarah Vanden Eede was the leading author of the current final draft (more and other analyses, restructuring and rewriting the manuscript) and she will be the contact person once the manuscript has been submitted.

Abstract

Beach nourishment is widely applied as a coastal protection measure because of its reduced ecological impact relative to hard coastal protection. In order to predict expected ecological impact on the sandy beach ecosystem, we developed a simulation model that integrates species envelope-based projections for the dominant macrobenthos species and mechanistic food web modules for higher trophic levels.

Species envelopes are estimated by using Bayesian inference of species' biomass relationships according to the three main determining abiotic variables, i.e. median grain size, total organic matter and the intertidal elevation, obtained from multiple independent sampling campaigns along the Belgian coast. Maximal possible abundances of higher trophic levels, being birds, shrimp and flatfish, are estimated based on their functional and energetic relationships with macrobenthos as an important food item.

After validation, we demonstrate that sediment grain size is the most important factor determining beach-level diversity and production, with strong deterioration after nourishment with too coarse sediment (e.g. >> than 300 µm). Nourishment slope had a smaller impact on the species zonation patterns compared to the sediment grain size. Patterns for higher trophic levels do not follow the decreasing patterns in macrobenthos abundance and biomass.

The advised gradient in sediment grain size for nourishment of fine-grained beaches is defined as 200 – 300 µm. This modelling approach shows that the assessment of ecosystem health needs to include the evaluation of different species richness and biomass variables. Focusing solely on for instance the potential abundance of species from higher trophic levels might lead to deceptive conclusions due to the dominance of opportunistic prey species.

Key words: beach nourishment, species envelope modeling, macrobenthos, fish, birds

1. Introduction

Coastal ecosystems are strongly threatened by climate change due to changes in sea level rise, erosion, changes in storm and wave regimes, flooding, altered sediment budgets and the loss of coastal habitat (Harley et al. 2006; Jones et al. 2007). In the last hundred years global average sea level has risen by 0.1 – 0.2 m (Houghton et al. 2001). In the past, the adaptive management response for the soft sediment ecosystems such as sandy beaches has been the enhancement of existing sea defences and retreat in areas of low-value land. Furthermore, coastal erosion has become an important issue on sandy beaches over the last decades as globally 70 % of beaches are receding, while 20 – 30 % remain stable and 10 % or less are accreting (Bird 2000).

Sandy beaches are the single largest coastal ecosystem on earth and they are covering 70 % of all continental margins (McLachlan & Brown 2006). They have a multitude of ecosystem functions as they are an important habitat for a variety of flora and fauna, and are concurrently of immense social and cultural importance to humans as prime recreational assets. More people interact directly with beaches than with any other type of shoreline worldwide (Schlacher et al. 2008). Sandy beaches also play an important role in coastal defence by functioning as a natural buffer between sea and land (Brampton 1992; Riddell & Young 1992), thus protecting landward sea defences from scour and wave erosion (Brampton 1992; Riddell & Young 1992).

In the past, the construction of hard structures as a management strategy for coastal defence enhanced beach erosion and destroyed important ecosystem functions (Defeo et al. 2009). Current widely applied defence approaches use beach nourishment to counteract coastal erosion and protect the land from flooding. This is particularly the case on the West European beaches of Belgium and The Netherlands, as these countries are vulnerable to sea level rise and storms due to their low height. Different types of beach nourishment can be executed. The most common types are high-shore, low-shore and profile nourishment (Speybroeck et al. 2005a). Up till now, technical aspects (e.g. easily available sand with coarse grain size and a rather steep and thus more stable beach slope (Finkl & Walker 2002) were dominant in taking management decisions for beach nourishment projects. In the light of international and European legislation, urging towards Integrated Coastal Zone Management (ICZM), attention for the functionality of the sandy beach ecosystem has increased (Schlacher et al. 2008). Although beach nourishment is generally considered as the least harmful beach management option (Hamm et al. 2002; Hanson et al. 2002), it does put a severe pressure on the biota living on, in and around sandy beaches (Speybroeck et al. 2006a). However, well-conceived impact studies are scarce (Jones et al. 2008; Leewis et al. 2012; Schlacher et al. 2012) and especially adequate information to predict the impact of nourishment on the beach ecosystem and to ecologically adjust nourishment projects, is lacking.

Optimizing technical aspects (e.g. characteristics of the sediment used, slope of the nourished beach, nourishment timing) of the nourishment projects is indispensable to maintain an ecologically healthy ecosystem on the beach. Indeed, the ecological characteristics of the beach fauna and flora are very much determined by morphodynamic beach characteristics such as grain size and beach slope (McLachlan & Jaramillo 1995; McLachlan 1996; Defeo & McLachlan 2005). As management of beaches is a multi-faceted and complex endeavour, where the interests of several stakeholders need to be combined, coastal managers need to interact with ecologists to integrate ecological aspects in beach management. Hence, clear and user-friendly management tools are essential in taking interdisciplinary management decisions (Schlacher et al. 2008).

As a good knowledge on the morphodynamics of Belgian sandy beaches is available (Degraer et al. 2003b; Speybroeck et al. 2008a), this beach ecosystem was used to develop a combined mechanistic-niche envelope model to predict the impact of beach nourishment on species richness at different trophic levels. The model builds further on well-established insights that the realised niche of lower trophic levels can be predicted based on three beach parameters, i.e. slope, grain size and organic matter, that are correlated under equilibrium conditions (Degraer et al. 2003b; Speybroeck et al. 2006a). Microphytobenthos and macrobenthos species composition are well documented along the Belgian coast. The importance of macrobenthos as food for birds and fish is also illustrated and quantified (Beyst et al. 1999a; Beyst et al. 1999b; Vanermen et al. 2009) and here mechanistically modelled by applying energy-based trophic interaction rules.

The general objective of this study was to develop a nourishment simulation model for the Belgian beach ecosystem. The simulation model had the aim (1) to predict short-term changes in beach macrobenthos species richness in response to changes in beach profile and grain size following beach nourishment and (2) to elucidate how these changes in community composition potentially feedback on the abundance of dominant species of higher trophic levels (birds, fish and shrimp).

2. Material & methods

2.1 Model description

For the model description, the 'Overview, Design concepts and Details' protocol (ODD) (Grimm et al. 2010) was followed. This protocol standardizes published model descriptions, making them less subject to criticism for being irreproducible.

2.2 Species envelopes

A species envelope is defined as the set of environmental conditions at locations where a species is known to occur, thereby assuming that on other locations with similar conditions, the species will also be present. This approach has been hugely successful, also in marine systems (Snickars et al. 2013) in other applications like climate change research, despite the caveats of such an approach for predicting large scale species ranges (Guisan & Zimmermann 2000; Bahn & McGill 2012). Earlier research elucidated either linear or quadratic relationships among the abundance or biomass of the main macrobenthos species with abiotic parameters (Beyst et al. 2002; Degraer et al. 2003b; Speybroeck et al. 2006a). Species envelopes for these taxa were derived from data collected on 23 intensively sampled beaches along the Belgian coast, during different seasons within the period 1997 – 2011 (Degraer et al. 2003b; Speybroeck et al. 2003; Speybroeck et al. 2005b; Welvaert 2005; Van Ginderdeuren et al. 2007; Vanaverbeke et al. 2008; Vanden Eede et al. 2008; Vanden Eede & Vincx 2010, 2011b) (see also Appendices – Chapter 5 – appendix A).

2.3 Entities, state variables, and scales

Model entities

The model consists of three major modules, one determining the abiotic conditions of the beach, a second module modelling predicted (changes in) abundance and biomass of macrobenthos according to their envelopes and a third one predicting the maximum abundance of the most important species from higher trophic levels. For the macrobenthos, the abundance and biomass of the eleven most dominant species were taken into account. These dominant species comprised the polychaetes *Nephtys cirrosa* and *N. hombergii*, *Capitella capitata*, *Spio filicornis*, *Pygospio elegans*, *Eteone longa* & *Scolelepis squamata*, the amphipods *Bathyporeia pilosa* and *Bathyporeia sarsi* and the isopods *Eurydice pulchra* and *E. affinis.* The most important species from higher trophic levels included the gulls *Larus canus* and *L. ridibundus*, the waders *Calidris alba* and *C. alpina*, the shrimp *Crangon crangon* and juvenile flatfish (mainly *Pleuronectes platessa*).

State variables

Three state variables determine the species richness and abundance of macrobenthos along sandy beaches: median grain size (MGS), total organic matter (TOM) and elevation (h) relative to the lowest tide, being 0 m TAW, i.e. the vertical level of reference in Belgium (McLachlan & Jaramillo 1995; Degraer et al. 2003b; Veloso et al. 2003). The slope of the beach (α) determines the submergence area and availability of prey for higher trophic levels (figure 1). The available biomass of species belonging to the lower trophic levels are input variables for estimating abundance of species from higher trophic levels and species richness (*S*).

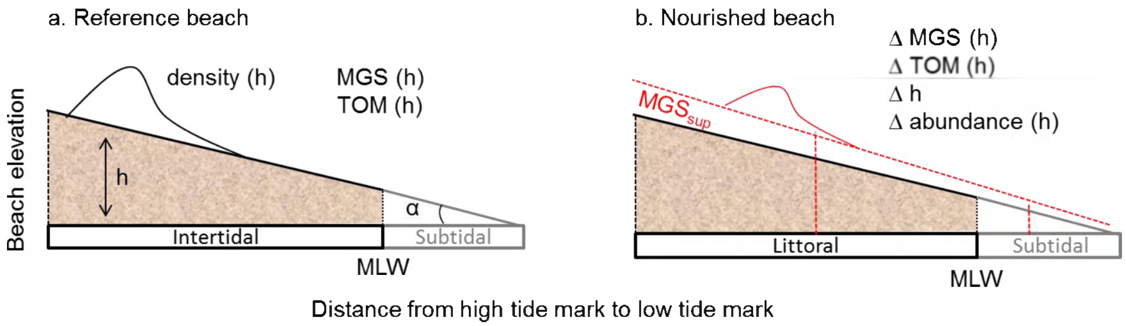


Figure 1: Schematic overview of the principal state variables. MLW: mean low water level

Scales

The model predicts the biomass, abundance and species richness of the macrobenthos and maximum carrying capacity for foraging predators at the scale of 1 m^2, according to the local conditions of the beach state variables. These estimates are interpreted along the intertidal transect and, when summed, at the beach level.

2.4 Process overview and scheduling

According to the input data on the beach profile after beach nourishment (changes in height according to the distance from the low water mark and grain size of the nourished sand), the model first estimates the local *TOM.* Based on the beach state variables and input data on macrobenthos niche properties, local abundance (number of individuals.m^{-2}) of macrobenthos is estimated and subsequently converted to biomass (g AFDW.m^{-2}) (appendix 1). Total availability of chlorophyll a (mg.m^{-2}) is estimated without conversion. More detailed information on the source of the input data and sampling strategies can be found in appendix 1.

This basic envelope modelling is used for both the reference situation, with *MGS* estimated according to *h* since earlier work has demonstrated the prevalence of such grain sorting mechanisms (Short 1991) and for nourished beaches with a predefined *MGS.* These envelope models are subsequently projected on the supplemented beach profile (changes in *h*, *α* and *MGS*), with *TOM* inferred from its previously derived relationship with *MGS.* In a second phase, estimated macrobenthos biomass is integrated into functions to determine the maximum local abundance of higher trophic levels, according to available biomass of prey species and availability according to tidal frequency. The model is stochastic with parameters for species envelopes and beach characteristics estimated from prior statistical distributions.

For each beach condition, 10 000 simulations are performed to estimate mean values and variance of species and predation pressure from higher trophic levels.

The model has been successfully evaluated and validated by three data sets from two beaches (of which data were not used to create the envelope model) for the dominant macrobenthos species that were sampled in an identical way as described in Degraer *et al.* (2003). Abundance of the dominant species, biomass (g total ash free dry weight (AFDW)) and species richness from the samples were subsequently compared with simulated data (average values and 95 % confidence intervals for 10 000 simulations) according to the sample location *h* and grain size *MGS* (see Appendices – Chapter 3 – appendix A).

2.5 Design concept

Basic principles

The model integrates envelope modelling approaches to estimate abundance and biomass of lower trophic levels into mechanistic modules to quantify maximum available biomass for higher trophic levels. The model allows a biotic evaluation of local beach nourishment impact on species richness by comparison of pre-nourishment and post-nourishment states. As currently, recolonisation processes are not well-known, no lag effects are incorporated in the model. The predicted state of the beach subsequently assumes equilibrium in species dynamics according to the envelope.

Interactions

Sand grains are sorted according to their elevation on the beach (Short 1991) with coarse grains deposited at higher locations. Total organic matter is positively correlated with the median grain size. Beach nourishment alters the height and the profile of the beach and it initially induces an unsorted, often coarser, sand grain distribution. Emerging abundances and biomass of prey items will eventually impact higher trophic levels, but no implicit interactions due to predation and interspecific competition are modelled. The input data for the prey items in this model are derived from non-disturbed beaches, so niche properties are assumed to reflect realized niche dimensions.

2.6 Submodel structure

Macrobenthos envelopes

We estimated niche dimensions for eleven dominant macrobenthos species (Appendices – Chapter 3 – appendix B) in relation to three abiotic input variables. As earlier work clearly indicated Gaussian niche dimensions, niche envelopes were modelled by second-order polynomial Poisson regression models because prior information evidenced both linear and quadratic responses according to the abiotic input variables (Degraer et al. 2003b). Parameter estimates were obtained by Bayesian estimation using a Monte-Carlo Markov chain (MCMC) procedure in WinBugs v. 1.4. (Spiegelhalter et al. 2003).

Macrobenthos biomass

The obtained regression coefficients (Appendices – Chapter 3 – appendix B) are used to estimate species abundances according to implemented beach characteristics in the main simulation model. Parameters were sampled from the obtained regression distributions kernels N(,SD) but constrained within the 95 % credibility interval. The eventual abundance estimates are subsequently transformed towards biomass (gram ash free dry weight, g AFDW) by earlier determined conversion factors ((Speybroeck et al. 2006b; Vanden Eede et al. 2013, in prep.); Appendices – Chapter 3 – appendix C).

Modelling trophic relationships

Macrobenthos – epibenthos

Previous work has demonstrated the importance of intertidal habitat for residing epibenthos foraging on macrobenthos (Kuipers & Dapper 1984; del Norte-Campos & Temming 1994; Beyst et al. 1999a; Koot 2009). Along Belgian beaches, epibenthos is dominated by *Crangon crangon* (95 %), while the other 5 % constitutes of juvenile flatfish, mainly *Pleuronectes platessa* (Beyst et al. 2001b). We experimentally quantified the maximum proportion of prey consumed by either *C. crangon* or juvenile flatfish. These values are used to estimate the maximum local predation pressure by epibenthos based on the available macrobenthos abundance. Because predation pressure is additionally time constrained, and only possible under submergence, the total available biomass at a certain elevation along the beach (*h)* for higher trophic levels is described by the following function:

$$\text{Biomass}_{\text{available}}\,(h) = {}_{x\text{->}i}\sum\,[\text{biomass}(h)*(1\text{-}h/h_{max})*\text{Predation pressure}_{x}] \qquad \text{(eq.1)}$$

with x = preyed species from the macrobenthos, h_{max} *the height of the beach at high tide* and [biomass] = g AFDW.m^{-2}

The caloric value of macrobenthos equals 23 kJ/g AFDW (Beukema 1997), so the available energy for higher tropic levels is

$E_{\text{available}}\ (h) = \text{biomass}_{\text{available}}\ (h) * 23$ with [energy] = kJ (eq.2)

From this available source of energy, the maximum number of *C. crangon* (constituting 95 % of the epibenthos), able to feed on this biomass at height *h* is based on their daily energetic needs (NEI_{crangon}), being 16 % of their total body mass (del Norte-Campos & Temming 1994). Based on the average biomass of a single *C. crangon* of 0.0175 g AFDW (Beukema 1992) and a mean average caloric value of 4.768cal/mg AFDW ≈ 20kJ/g AFDW (Szaniawska 1983; Zwarts et al. 1996), $NEI_{\text{crangon}} = 0.056$ kJ; the maximum abundance of *C. crangon* at location *h* can be calculated as follows:

$\text{Crangon}_{\max}\ (h) = (E_{\text{available}}\ (h)*0.95)/NEI_{\text{crangon}}$ (eq. 3)

Similarly, the *NEI* for juvenile flatfish is estimated to be 10 % of the body mass (Lockwood 1984), so the maximal abundance of flatfish at location *h* can be calculated as follows:

$\text{Flatfish}_{\max}\ (h) = (E_{\text{available}}\ (h)*0.05)/NEI_{\text{flatfish}}$ (eq. 4)

with $NEI_{\text{flatfish}}=0.188$ kJ

These estimates should be regarded as ceiling values for higher trophic levels, since it assumes immediate consumption under laboratory conditions, though mimicking natural prey abundances.

Waders

Two wader species, *Calidris alba* and *C. alpina*, feed predominantly on macrobenthos along Belgian beaches. Although both species show some differences in foraging behaviour, they both feed on the same prey and therefore we treat them as being one functional group. Both species were found to forage about 25 % of their total residential time on all macrobenthos species (Speybroeck 2007; Vanermen et al. 2009). According to Vanermen et al. (2009), waders along soft sanded beaches only forage from two hours before till four hours after low tide. This implicates that foraging is not possible at the high littoral (upper quartile of the beach). At the low littoral, foraging is possible twice a day for six hours, resulting in a daily foraging time of 12 hours in the low littoral and thus a foraging time of 0.50 (eq. 5).

This leads to maximal foraging time at a certain height (*h*) as follows:

$\text{Foraging Time}\ (h) = -0.25+0.75*h_{rel}$ (eq. 5)

with h_{rel} the relative proximity to the low water level (being 1 when at 0 m TAW, being 0 at high water) and Foraging Time (*h*)=0 when eq.6 yields values <0.

The availability of prey is additionally dependent on the slope of the beach since this affects the depth of the prey burrowing into the sediment, with prey unavailable for waders when the water table exceeds 40 cm beneath the surface (Stienen, personal communication). Foraging possibilities are theoretically maximal on flat beaches and minimal when beach slopes exceed 21° (a zone of less than 1 m available at the water line). Taking into account continuous changes in biomass availability (foraging time) for the central and lower littoral, a caloric value of macrobenthos of 23 kJ/g AFDW and a daily energy uptake for small waders (NEI_{waders}) of on average 224 kJ/day (Kersten & Piersma 1987; Castro et al. 1989; Speybroeck 2007), potential wader pressure can be calculated as follows:

$$\text{Wader pressure } (h) = \text{foraging time} * \text{ biomass } (h)*(1-\alpha/21°)*23/NEI_{\text{waders}} \qquad \text{(eq. 6)}$$

Small-sized gulls

Larus canus and *L. ridibundus* are the principle gull foragers on beaches. They feed on polychaetes and *C. crangon* (Speybroeck 2007). Prey availability within the littoral food web is maximal at low tide and concentrated in beach pools. Because the lack of any insights into this pool formation and temporal patterns in gull foraging behaviour, we assume polychaetes and *Crangon* biomass to be available after submergence.

$$\begin{aligned} \text{Biomass}_{\text{avialable}} (h) &= \text{biomass}_{\text{polychaeta}} + \text{biomass}_{\text{crangon}} \qquad \text{(eq. 7)} \\ &= {}_{x\text{->}i}\sum \text{biomass}_x\, (h)*(h/h_{max}) \end{aligned}$$

with x = species polychaetes and *C. Crangon*

Given caloric (cal) values for polychaetes and *C. crangon* of respectively 23 kJ/g AFDW and 20 kJ/g AFDW, and the average daily energy need of small *Larus* species ($NEI_{\text{gulls)}}$ of 607 kj (Ysebaert & Meire 1989), the potential maximum number of foraging gulls is:

$$\text{Gulls } (h)= {}_{x\text{->}i}\sum ((\text{Biomass}_{\text{available}})_x * \text{cal}_x)/ NEI_{\text{gulls}} \qquad \text{(eq. 8)}$$

with x respectively polychaetes and *C. crangon*

Although several bird species are also known to feed on stranded wrack material, this trophic link was not incorporated in this model due to the difficulties of quantifying stranded wrack on beaches.

2.7 Model application & research strategy

The nourishment model was applied on a typical Belgian beach. Two of the thee parameters predicting the realised niche of lower trophic levels, e.g. slope and grain size, are technical aspects of the nourishment projects and can be optimized accordingly. The relative importance of each parameter is tested under realistic conditions, by keeping the other parameter constant. Initially, the influence of altered beach profile and nourishment slope was simulated for the fauna on the beach. While maintaining the natural sediment grain size (ranging from 139 to 285 µm), three beach profiles were tested: the natural beach profile (t0) a nourishment profile of 18° (s1) and a nourishment profile of 33° (s2) (figure 2). After nourishment, average macrobenthos abundances can be higher, lower or equal to the abundances before the nourishment and this response differs between species. We predicted the effects on macrobenthos abundance per dominant species, total macrobenthos biomass and predation pressure of higher trophic levels including avifauna and epibenthos (fish and shrimp), present on the beach. Subsequently, the effect of varying sediment grain sizes used for nourishment was simulated for the afore-mentioned fauna: 200 – 500 µm with increments of 50 µm. In this case, the nourishment profile did not deviate from the natural beach profile (t0). Predictions of the effects on macrobenthos and on higher trophic levels were made, identical to the ones made for different beach profiles.

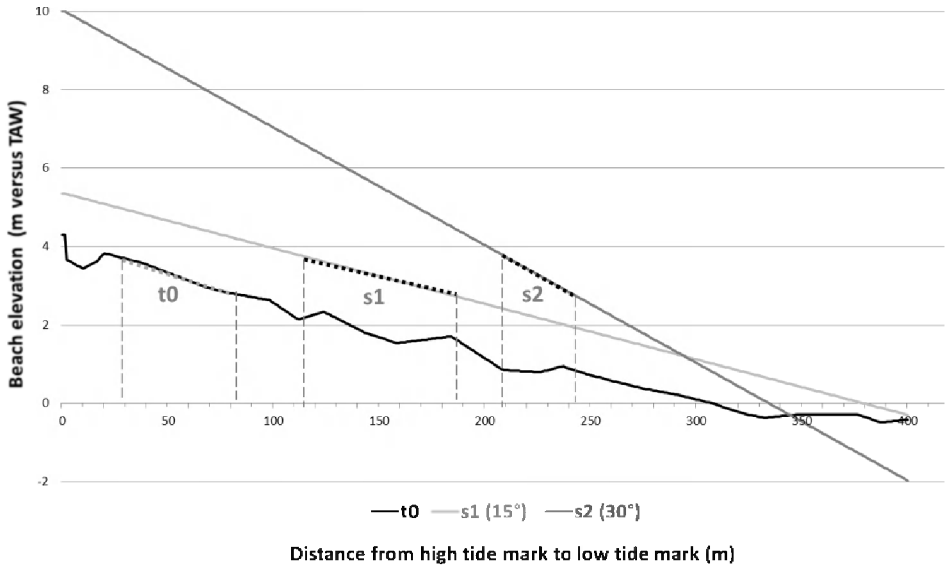


Figure 2: Different nourishment profiles and the exemplary shift of habitat on the beach due to nourishment. t0: natural beach profile; s1: nourishment profile of 18°; s2: nourishment profile of 33°. X-axis: across-shore beach distance from a fixed point above high tide mark (left) towards low tide mark (right). Y-axis: relative beach elevation (m versus TAW), calculated by the M2-reduction model (Van Cauwenberghe et al. 1993)

The t0 situation, frequently depicted in the following figures and tables, encompasses the macrobenthos situation on this typical Belgian beach prior to nourishment impact effects. This t0 simulation is based on the information of 23 intensively sampled beaches along the Belgian coast, during different seasons within the period 1997 – 2011. In these simulated pre-nourishment conditions, sediment is in equilibrium and well-sorted across the shore (with coarser sediment on the upper shore and finer sediment on the lower shore). Conversely, the simulated t1 situation is characterized by a uniform sediment grain size.

3. Results

3.1 Influence of altered beach profile and nourishment slope

An important consequence of beach nourishment, coinciding with the steeper beach slope, is the shift in intertidal beach area (figure 2) and consequently the decrease in species abundance (table 1), except for *Bathyporeia pilosa* and chlorophyll a, which is a measure for the microphytobenthos. The macrobenthos species abundance (table 1), the total macrobenthos biomass and the trends for the higher trophic beach levels (figure 3) did not differ between nourishment profile types s1 (15°) and s2 (30°) used on a single beach. The chlorophyll a levels and the abundance of *Bathyporeia pilosa*, *Nephtys cirrosa* and *Scolelepis squamata* increases slightly when a steeper slope is applied (s2: 30°) while the opposite is true for *Bathyporeia sarsi* and *Eurydice pulchra* (table 1).

Table 1: Simulated chlorophyll a (mg.m^{-2}) and abundance (number of individuals.m^{-2}) of the dominant species on a typical Belgian beach for the pre-nourishment (t0, median grain size: 218.31 µm and slope: 15°) and post-nourishment situation, using different slopes: s1 (15°) and s2 (30°); mean ± SE (based on 10 000 simulations)

Slope	chlorophyll a (mg.m^{-2})	*Bathyporeia pilosa*	*Bathyporeia sarsi*	*Eurydice Pulchra*	*Nephtys cirrosa*	*Scolelepis squamata*
t0	2.03 ± 0.01	111.5 ± 9.3	323.8 ± 7.7	7.4 ± 1.0	47.0 ± 1.4	178.9 ± 10.2
s1 (15°)	3.05 ± 0.02	242.9 ± 15.4	283.0 ± 7.0	3.7 ± 0.4	26.1 ± 1.0	59.5 ± 5.2
s2 (30°)	3.07 ± 0.02	287.8 ± 16.1	263.5 ± 6.8	2.6 ± 0.5	34.4 ± 1.2	60.1 ± 5.4

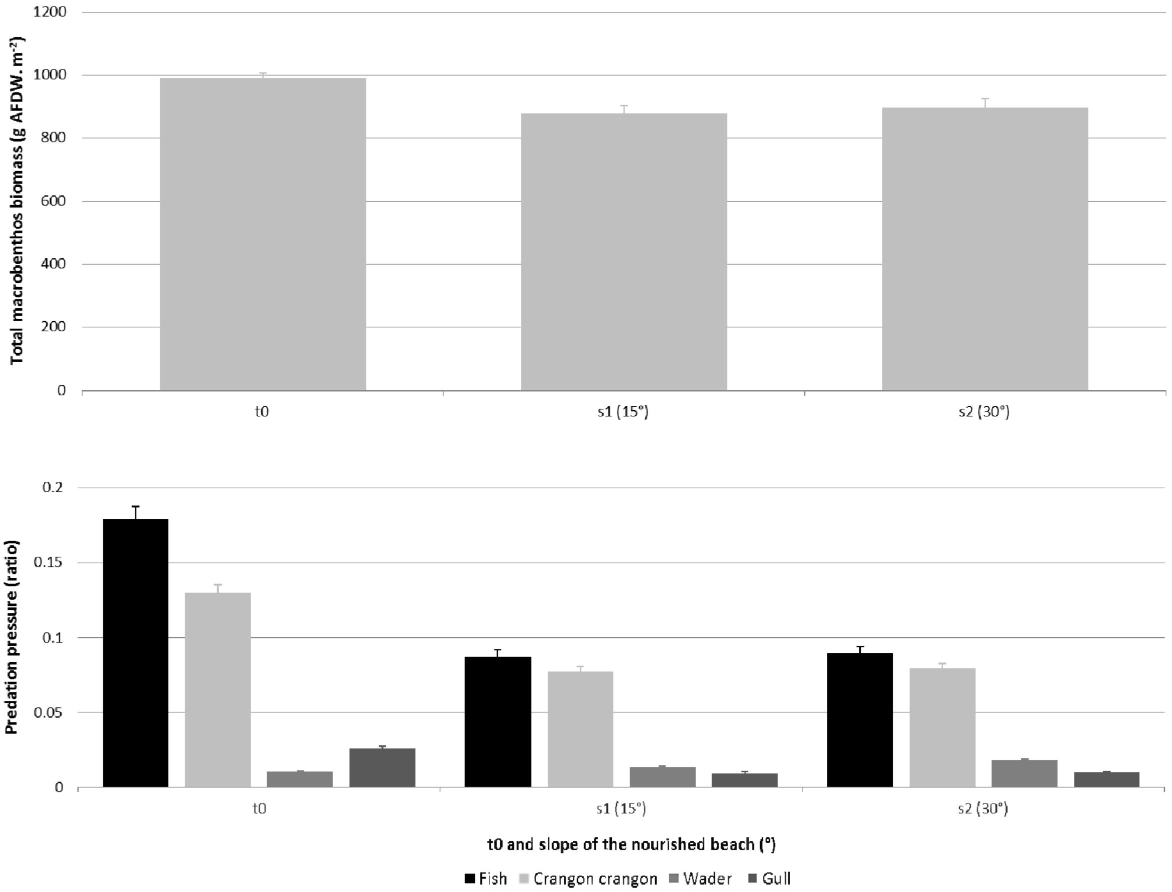


Figure 3: Simulated macrobenthos biomass and potential predation pressure of higher trophic levels on a typical Belgian beach. X axis: t0 and slope of the nourished beach (°); Y-axis: *above:* total macrobenthos biomass (g AFDW.m^{-2}) and *below:* potential predation pressure (ind.m^{-2}); mean ± SE (based on 10 000 simulations)

3.2 Influence of the used sediment (grain size)

The average simulated abundance of the species after nourishment is similar to pre-nourishment conditions when the grain size of the used sediment resembles natural conditions (200 – 250 µm). However, when nourishment sediment differs in grain size from natural beach sediment, species abundances will respond more profoundly to this habitat transformation. The chlorophyll a pattern (measure for the microphytobenthos) and macrobenthos patterns are given in table 2. The microphytobenthos, the amphipod *Bathyporeia pilosa* and the polychaete *N. cirrosa* show a clear negative trend when beaches are nourished using sediment with a median grain size of 300 µm or coarser (table 2). For the amphipod *Bathyporeia sarsi*, this negative trend starts from 350 µm onwards (table 2). *Eurydice pulchra* and *Scolelepis squamata* are not negatively but positively influenced by nourishment using sediments with a medium grain size of (more than) 300 µm (table 2). However, this coarse sediment is not naturally occurring along the Belgian coastline (Vanden Eede & Vincx 2011b; Vanden Eede & Vincx 2011a).

Table 2: Simulated chlorophyll a (mg.m^{-2}) and species abundance (number of individuals.m^{-2}) on a typical Belgian beach for the pre-nourishment (t0, median grain size: 218.31 µm) and post-nourishment situation, using different sediment grain sizes; mean ± SE (based on 10 000 simulations)

Grain size (µm)	chlorophyll a (mg.m^{-2})	*Bathyporeia pilosa*	*Bathyporeia sarsi*	*Eurydice pulchra*	*Nephtys cirrosa*	*Scolelepis squamata*
t0	2.03 ± 0.02	111.5 ± 9.3	323.8 ± 7.7	7.4 ± 1.0	47.0 ± 1.4	178.9 ± 10.2
200	3.05 ± 0.02	242.9 ± 15.4	283.0 ± 7.0	3.7 ± 0.4	26.1 ± 1.0	59.5 ± 5.2
250	2.32 ± 0.01	225.8 ± 14.8	311.2 ± 7.6	12.5 ± 1.4	12.7 ± 0.6	302.8 ± 13.7
300	1.59 ± 0.01	162.9 ± 12.1	309.4 ± 8.0	20.0 ± 2.5	4.7 ± 0.3	553.4 ± 19.1
350	0.98 ± 0.01	82.4 ± 7.3	125.4 ± 4.8	17.5 ± 2.8	1.7 ± 0.2	524.7 ± 19.0
400	0.82 ± 0.01	0.0 ± 0.0	117.0 ± 5.2	41.0 ± 4.8	0.7 ± 0.1	423.6 ± 17.6
450	0.55 ± 0.01	0.0 ± 0.1	30.9 ± 2.5	47.2 ± 5.5	1.4 ± 0.2	424.3 ± 18.8
500	0.42 ± 0.01	0.0 ± 0.2	79.8 ± 4.5	32.8 ± 4.5	1.8 ± 0.3	468.3 ± 20.4

Figure 4 shows that after nourishment using sediment with a median grain size from 300 µm onwards, macrobenthos species richness on the beach decreases. Nourishment with sediment characterized by a median grain size of 350 µm will cause a decrease in macrobenthos species richness by 30 % compared to the t0 situation. There seems to be no apparent species richness loss when fine sediment is used (200 and 250 µm) (figure 4).

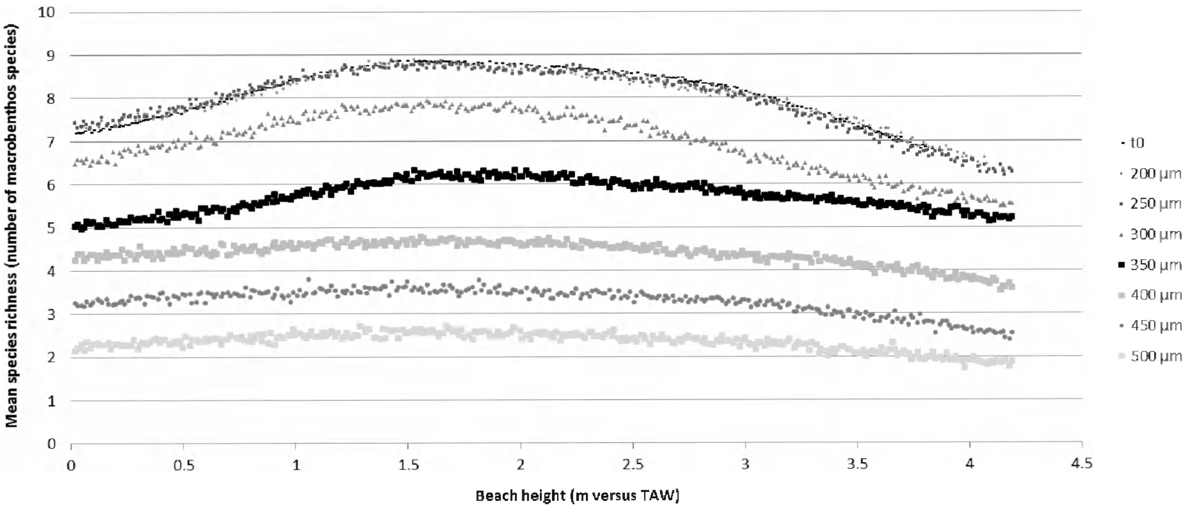


Figure 4: Expected species richness on a typical Belgian beach before (t0, median grain size: 218.31 µm) and after nourishment with different sediment grain sizes (µm). X-axis: beach height (m versus TAW); Y-axis: mean species richness (number of macrobenthos species)

The species richness results (figure 4) are contrasted by the biomass results (figure 5). While the maximum macrobenthos species richness was found at a median grain size of 200 – 250 µm, the maximum total macrobenthos biomass was found at 300 µm. The total biomass after nourishment with 350 µm was comparable to the biomass in the t0 situation, but from 350 µm onwards, there was a decrease in total macrobenthos biomass. At 400 µm, total biomass was similar to the situation at 200 µm and from 400 µm onwards, there was again an increase in total macrobenthos biomass towards conditions comparable to the t0 situation (figure 5). The effects of beach nourishment on higher trophic levels can be completely linked to the evolution of total macrobenthos biomass after nourishment with different types of sediment. For shrimp, juvenile flatfish and birds, there is a maximum potential predation pressure (number of individuals.m^{-2}) at 300 µm, followed by first a decrease and then again an increase in presence on the beach at increasing coarser grain sizes (figure 5).

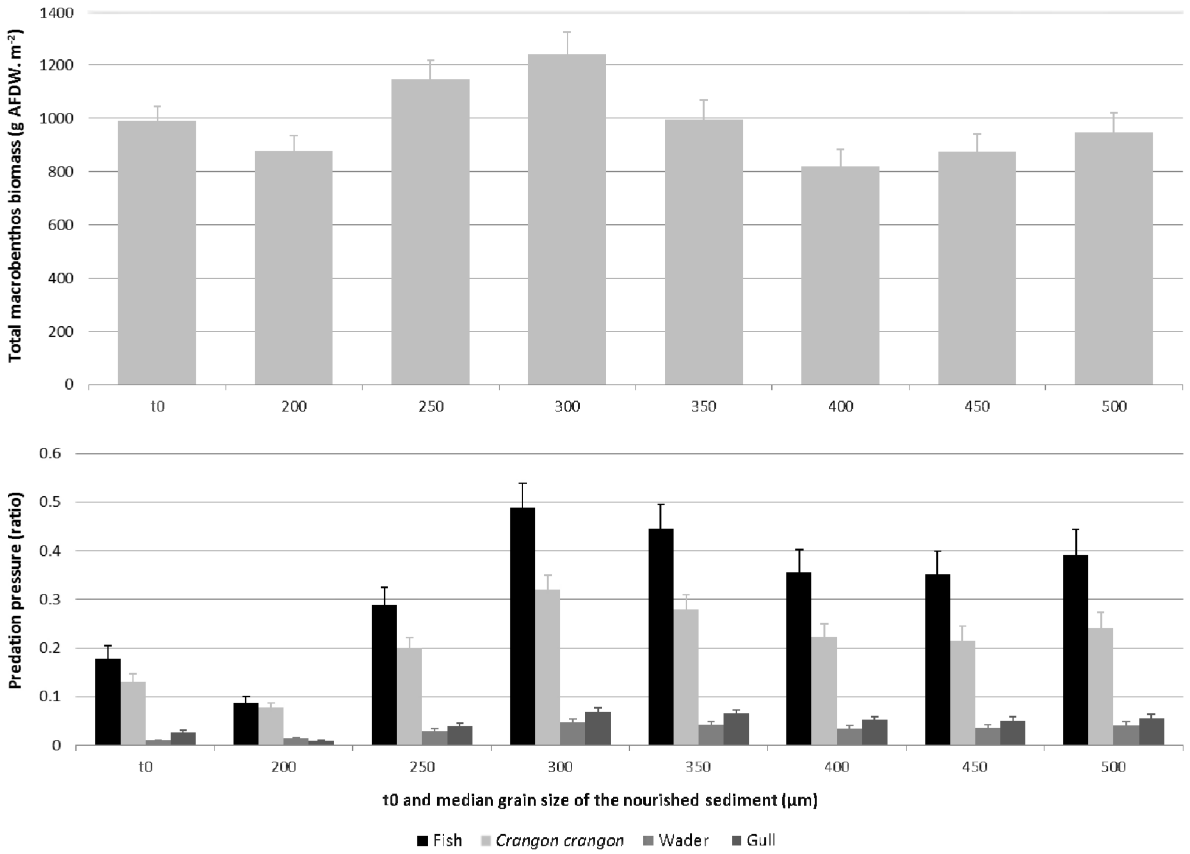


Figure 5: Simulated macrobenthos biomass and potential predation pressure of higher trophic levels on a typical Belgian beach. X axis: t0 (median grain size: 218.31 µm) and median grain size of the nourished sediment (µm); Y-axis: *above:* total macrobenthos biomass (g AFDW.m^{-2}) and *below:* potential predation pressure (ind.m^{-2}); mean ± SE (based on 10 000 simulations)

4. Discussion

Beach nourishment is known to alter the morphodynamic state of the beach due to the significant changes in beach slope and sediment. Our modelling approach indicates that the grain size of the sediment, used for beach nourishment, is the dominant factor in determining the effects on the ecosystem. The gradient for median grain size of nourishment sediment on dissipative West European beaches is advised to be 200 – 300 µm, in order to reduce the impact effects as much as possible.

The evaluation of the beach ecosystem health by total macrobenthos biomass or by the presence of socially or economically important species, like some birds and fish, can be deceptive as generalist species (cfr. *Scolelepis squamata*) can become very abundant on a beach impacted by nourishment. Hence, the combination of different variables (species richness, biomass) is advisable to estimate the effects of nourishment on the beach ecosystem. Finally, the development of the nourishment model in this study is only a first step and the refinement and enhancement of the model relationships should greatly contribute to a better ecosystem-based nourishment approach in the future.

4.1 Assessing ecological impact

Envelope models are widely used in ecological assessment studies. These models assume that species show no dispersal limitations and have constant niches. Moreover, biotic interactions are expected to be incorporated in the models as input data are gathered from field situations where these interactions determine the distribution patterns of species (Araújo & Peterson 2012). In the nourishment context, envelope models are thus highly usable and have advantages over models predicting processes on a larger spatial scale (such as climate models). The nourishment model developed in this study is based on a large set of input data, was validated and assumes realistic and highly likely short term responses.

Influence of the beach profile and slope

Beach erosion combined with increasing economic and human development along the coast, is resulting in coastal squeeze. To counteract this evolution, beach nourishment is aiming at coastal relaxation. This management measure however has ecological implications for the sandy beach ecosystem. The model simulations indicated a decrease in total macrobenthos biomass on the beach (figure 4) as a result of the steeper nourishment slope leading to a narrower intertidal beach (difference between t0 and s1 or s2).

On a smaller scale, when different nourishment slope types were taken into account on one specific beach, the nourishment model showed that particular nourishment slopes could positively influence the microphytobenthos and even favour specific species. This possibility to compare species responses to different slopes makes this model especially useful for management purposes, aiming at protecting species or habitats. Indeed, by positively influencing abundance of macrobenthos, through the use of specific beach slopes enlarging the habitat of dominant prey species (such as high-shore nourishment, positively influencing *Bathyporeia pilosa* in the high-intertidal), the presence and abundance of higher trophic levels such as birds or fish can also be positively influenced. In this context, the knowledge that intermediate beaches with steeper beach slopes are unfavourable for epi- and hyperbenthos (Beyst et al. 2001b), should be taken into account when developing nourishment slopes.

Influence of the used sediment (grain size)

The nourishment sediment is of vital importance to predict the effects of nourishment on the beach ecosystem (Speybroeck et al. 2006a). The dominant role of sediment grain size was supported through the results obtained by testing different beach slopes and sediment grain sizes. Indeed, most of the modelled macrobenthos species did not respond to the different slope type but only to the grain size of the sediment used. Coarse sediments (median grain size of 300 µm or coarser), not naturally occurring on Belgian beaches, negatively influenced the microphytobenthos, the amphipods *Bathyporeia pilosa* and *Bathyporeia sarsi* and the polychaete *N. cirrosa* (table 2). However, these coarse sediments positively influenced the polychaete *Scolelepis squamata*, resulting in an increase in total macrobenthos biomass and an increase of the trophically linked birds and fish present on the beach. The polychaete *Scolelepis squamata* is a generalist and cosmopolitan species both thriving on fine-grained as well as coarse-grained beaches (Dauer 1983; Hartmann-Schröder 1996). Although the polychaete is restricted to the upper-intertidal zone on Belgian beaches, its distribution on other beaches worldwide is more spread over the entire beach. Currently, the reason(s) for these different spatial distribution patterns remain unknown and no clear sediment preference for this species could be found. The high biomass of *Scolelepis squamata* after nourishment further confirms its broad sediment tolerance range.

Opposite to the calculated abundance and biomass patterns, the overall species richness was simulated to decline when coarse sediment was used for beach nourishment. The contrasting abundance, biomass and species richness' patterns clearly show that macrobenthos or avian biomass, as single descriptors to evaluate the health of an ecosystem, are insufficient and can lead to wrong conclusions concerning ecosystem health. We therefore advise to use a combination of species richness, abundance and biomass indices to monitor the ecological impact of nourishment on sandy beach ecosystems.

Furthermore, it is precarious to conclude that the effect of nourishment using coarse sediment is harmless for higher trophic levels in general, due to the afore-mentioned uncertainties in the model concerning these higher trophic levels.

4.2 Strengths and weaknesses of the model

The model predicts if the habitat after nourishment is considered suitable for the most dominant members of the sandy beach community, although the predicted species may in practice be absent because of other anthropogenic impacts, such as tourism or natural temporal variability (Brown & McLachlan 2002; Defeo & McLachlan 2005; Defeo et al. 2009; McLachlan et al. 2013). The model however gives a prediction without taking into account the nourishment period, the nourishment method and the techniques used. After a nourishment event, both the beach slope as well as the characteristics of the beach sediment will evolve towards pre-nourishment conditions (Speybroeck et al. 2006a). Although some data show that macrobenthos recovers completely within short time frames (Gmelig Meyling & De Bruyne 1994; Slim & Löffler 2007), these statements are not based on peer-reviewed information or well-developed monitoring. Furthermore, the post-nourishment evolution is depending on several nourishment-specific (e.g. high-shore versus profile nourishment) but also ecosystem-dependent features (erosion-susceptibility of the beach ecosystem, recolonising capabilities of the sandy beach species). Nevertheless, personal observations during post-nourishment monitoring indicate a rapid recolonisation of the infauna due to the rather small scale of the current nourishment, so no lag effects are currently incorporated in the model. A further examination of the temporal and spatial post-nourishment processes, based on a scientifically based monitoring, is however essential to extend and refine the predictions of the model for a longer time period and for more large-scaled nourishment projects. These predictions on the beach evolution after nourishment will also be highly important for the frequency of repeated nourishment and this repeated nourishment will in turn have an effect on the evolution of the beach after nourishment.

As the sandy beach food web is complex and all species interactions are not yet elucidated, the predictive effect of the nourishment impact on higher trophic levels should be regarded as an assessment of the potential rather than realistic predictions. One of the main predictions of the model showed a positive effect on both the total macrobenthos biomass as well as on the potential presence of predators after nourishment using coarse sediment. However, it should be kept in mind that the model simulated the potential presence of a restricted but relevant set of species belonging to the higher trophic levels solely based on macrobenthos productivity. The increase in total macrobenthos biomass, following nourishment with coarse sediment, can be exclusively attributed to the increase of the generalist polychaete *Scolelepis squamata*. Furthermore, the predators incorporated in the model were strongly linked to this polychaete as main food item and therefore their potential presence was simulated to be relatively high. Nevertheless, the latter result needs careful consideration, as firstly, these predators also feed on other sources that were currently not incorporated in the model (such as stranded wrack material (De Meulenaer 2006) and secondly, the potential presence of predators is not linked in the current model to abiotic variables such as beach morphodynamics or hydrological conditions although they may affect the presence of epibenthos and hyperbenthos (Beyst et al. 1999a; Beyst et al. 2001a; Beyst et al. 2002).

Although on other beaches worldwide other species occur, their taxonomic position and trophic relationships are very similar to those of the species considered here (McLachlan et al. 1996a; Defeo & McLachlan 2005) and therefore the model can be valuable for sandy beach ecosystems worldwide. In addition, when invasive species will recolonise the nourished and morphodynamically altered beach or when keystone species disappear, a complete ecosystem shift is possible (Schlacher et al. 2008; Mumby et al. 2012; Perry et al. 2013).

4.3 Synthesis and applications

Beach nourishment in practice occurs with a sediment grain size from 200 µm onwards. Based on the results of the model, we distinguish three groups of beach habitat, based on the most important variable according to the model, grain size of the sediment: (1) 200 – 250 µm; (2) 300 µm and (3) ≥ 350 µm. In figure 6, an overview of the beach nourishment effects for these different habitats is given.

The different ecosystem components included in the model are evaluated. As long as the used sediment resembled the sediment in pre-nourishment conditions, the ecosystem did not change. The use of coarse sediment (sediment grain size ≥ 300 µm) for nourishment had a negative effect on macrobenthos species richness. Due to the differences between simulated patterns of total biomass and species richness, the importance of these variables should however be carefully considered.

Furthermore, it could be stated that beach nourishment with a sediment grain size of 300 µm is most favourable for higher trophic levels based on the results of this model. Nevertheless, this result is largely depending on the strong correlation of the used predators with the generalist polychaete *Scolelepis squamata* and is likely to change when more predators and additional trophic and abiotic links are included in the model. Due to these uncertainties regarding the presence of higher trophic species, the gradient in sediment grain size that is advised to be used for nourishment of natural fine-grained beaches is established as 200 – 300 µm.

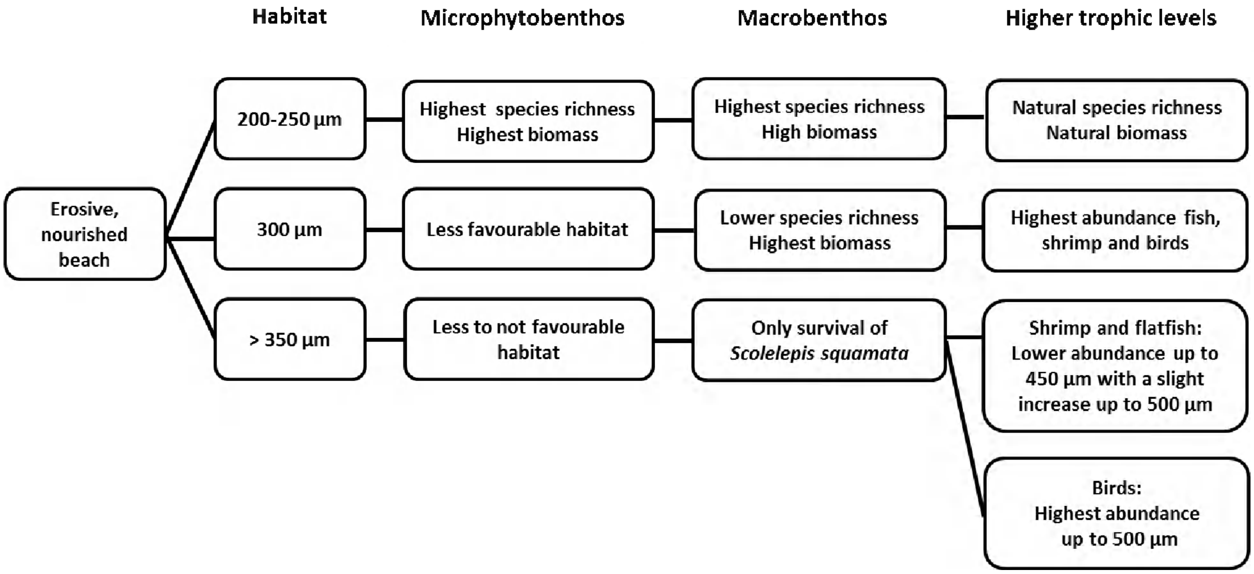


Figure 6: Schematic overview of beach nourishment impact on the beach ecosystem, based on the nourishment model simulations

5. Acknowledgements

The first authors are grateful to the funding authorities AMINAL-Nature (Flemish administration responsible for environment, nature, land and water management, department Nature – within the BEST project, file number AN.GKB/2002/nr.2), AWK (Coastal Waterways Division – within the projects with file numbers 202.165, 202.290, 04/A2084-3, 205.240, 207.230, 208.245, 209.090, 211.095), ANB (Agentschap Natuur en Bos) and Ghent University and they acknowledge an aspirant grant provided by the Flemish Fund for Scientific Research (FWO).

Chapter 6: Marine biological valuation of the shallow Belgian coastal zone: a space-use conflict example within the context of marine spatial planning

Sarah Vanden Eede, Lia Laporta, Klaas Deneudt, Eric Stienen, Sofie Derous, Steven Degraer, Magda Vincx (in prep.) Marine biological valuation of the shallow Belgian coastal zone: a space-use conflict example within the context of marine spatial planning.

In preparation for submission to Ocean and Coastal Management

Abstract

The Belgian coastal zone hosts a complex of space-use and resource-use activities with a myriad of pressures. Specifically at the beaches, predictions on sea-level rise, storms and flood risk from the North Sea have led to several big coastal defence projects. Management of sandy beaches is a multi-faceted and complex endeavor, where the interests of several stakeholders need to be combined.

In this paper, we used the marine biological valuation (BV) method in order to (1) analyse the spatial structure of the intertidal and shallow subtidal Belgian coastal zone; and (2) explore the applications of BV for an ecosystem-based approach to marine spatial planning of two space-use conflicts at the Belgian coast, being flood protection, by means of beach nourishment, and nature conservation.

The biological value was assessed with a focus on a detailed and integrated dataset (1995 – 2011), gathering all available ecological information on macrobenthos, epibenthos, hyperbenthos and birds. The 67 km Belgian coastline was divided into an across-shore intertidal and shallow subtidal subzone while the width of the along-shore subzones comprises 250 m for benthic components and wider distances of 3 km for the birds. The intrinsic biological value of each subzone was calculated using the BV method and the pertained score, ranging from *very low* to *very high,* was plotted accordingly in order to obtain a marine biological valuation map (BVM).

Following trends in BV along the Belgian coastline were detected: (1) a strong mosaic pattern of BV along the coastline; (2) a clear lack of (benthic) data at the eastern part of the Belgian coast; (3) a rather *high* biological value score for around 70 % of the shallow part of the subzones, compared with the intertidal part; (4) a *high/very high* biological values found in intertidal zones located immediately to the east of the harbours Nieuwpoort, Oostende and Zeebrugge.

A detailed analysis of protected areas and areas under coastal flood risk indicates that the use of BVMs is very promising in order to differentiate between several impact values. BV can therefore be used as a management tool by local decision makers and can allow for the integration of 'natural/ecological values' at an early stage of policy implementation.

Key words: biological valuation, shallow coastal zone, space-use conflict, marine spatial planning

1. Introduction

Marine and coastal waters are sensitive habitats that support high levels of biodiversity and provide many essential ecosystem goods and services (Costanza et al. 1998; de Groot et al. 2002; Beaumont et al. 2007; Beaumont et al. 2008). The escalating crisis in these ecosystems, from biodiversity losses and transformed food webs to marine pollution and warming waters, has been recognized to increasingly undermine the ocean's capacity of providing goods and services and maintaining resilience to stressors and changes (Worm et al. 2006). This crisis is in large part a failure of integrated governance (Crowder et al. 2006; Crowder & Norse 2008). Current governance of marine systems is not place-based (Pikitch et al. 2004) but developed for particular marine resources and within individual economic sectors (Laffoley et al. 2004; Douvere 2008). In Belgium for instance, legal jurisdiction concerning coastal management is shared between the Flemish Government (landwards from the mean low water level (MLW)) and the Belgian State (seawards from the MLW). Such 'multi-level government' structure (Cliquet 2001, De Ruyck et al. 2001, Cliquet et al. 2007) most often results in conflicting priorities and overall lack of clarity in the implementation of relevant policies at the coastal zone (Commission of the European Communities 2007). It fails to provide a comprehensive integrated management of human activities, leading to fragmentation and spatial/temporal mismatches in governance. However, ecosystems, natural resources and human activities affecting coastal zones have place-based characteristics thus increasing the need to look at the 'system' from a spatial and temporal perspective. This also implies that all policies and management strategies (e.g. fisheries management, marine transportation management and marine protected area management) directed towards influencing human use of ecosystems and their resources, will inherently have a spatial and temporal dimension (McLeod 2005; Crowder & Norse 2008).

During the last decade, marine spatial planning (MSP) has gained considerable importance in establishing ecosystem-based management in the marine environment. Ecosystem-based MSP seeks to attain not only consensus in sea-use management among distinct sectors, but also and most importantly to maintain the ecosystems' integrity and services through the conservation of marine biodiversity (Douvere 2008; Pomeroy & Douvere 2008; Douvere & Ehler 2009; Ehler & Douvere 2009; Commission 2013b). This approach has been implemented in a few countries on a preliminary basis (Ehler 2008; Gilliland & Laffoley 2008; Foley et al. 2010), including Belgium, although only in the marine offshore areas under federal jurisdiction. Biodiversity can be valued under several approaches and at several scales (Noss 1990; Oksanen 1997; Costanza 1999; Balvanera et al. 2006; Granek et al. 2009). In fact, the objectives behind each approach are directly linked with the respective definition of the term 'value' (Derous et al. 2007a). Most commonly, this is associated with the socio-economic value of ecosystems (Pearce & Moran 1994; Costanza 1999), reflecting vestiges of the anthropocentric perspective over natural resources (Collet 2002). Valuing ecosystems by estimating the benefits they provide to society, accruing to ecosystems' goods and services, is an increasingly common practice in literature (de Groot et al. 2012). Under an ecosystem-based management approach however, biodiversity should also be valued intrinsically, independently of its potential usefulness for human beings (Wilson 1986; Ghilarov 2000).

The present work focuses on marine biological valuation (BV), a spatial tool that provides an integrated view on nature's intrinsic value, within a certain time frame (Derous et al. 2007a; Derous et al. 2007b). In this valuation method, all levels of biodiversity are assessed through a hierarchical ecological framework (Zacharias & Roff 2000). By compiling all available biological and ecological information for a selected study area, and allocating an integrated intrinsic biological value to the subzones within the study area, biological valuation maps (BVMs) are produced. These maps facilitate the provision of a greater-than-usual degree of risk aversion in the management of activities as they are a tool for calling attention to areas which have particularly high ecological or biological significance (Derous et al. 2007a). Therefore, they can be used as reliable and meaningful baseline maps for spatial planning, marine policy and management approaches (Derous et al. 2007a; Derous et al. 2007b; Pascual et al. 2011). Hitherto, marine biological valuation has been performed in different European subtidal coastal waters (Derous et al. 2007d; Forero Parra 2007; Rego 2007; Vanden Eede 2007; Pascual et al. 2011) including the Belgian Part of the North Sea.

The goals of this paper are two-fold: (1) to analyse the ecological structure on a spatial scale of the intertidal and shallow subtidal Belgian coastal zone using the marine BV method; and (2) to explore the applications of BV for an ecosystem-based approach to MSP of two space-use conflicts at the Belgian coast, being flood protection, by means of beach nourishment, and nature conservation.

2. Material and methods

2.1 Study Area

The Belgian natural coastline (figure 1) is entirely composed of sandy beaches. The ecological continuum expected in this type of ecosystem, from the intertidal zone to the foredunes, is however disrupted by stone groynes and concrete dykes (De Ruyck et al. 2001), as a response to coastal flood risk (Speybroeck 2007; Roode et al. 2008). Previous research of the Belgian coastal ecosystem (Speybroeck et al. 2008a) suggested a zonation scheme, delimitating three main zones, along the tidal range: (i) the *supralittoral zone*, the area above the high water line influenced by sea water, represented by embryonic dunes, the dry beach area, and the drift line; (ii) the littoral or *intertidal zone*, the area comprised between high water and low water lines; and (iii) the *infralittoral or shallow subtidal zone*, represented by the subtidal foreshore as the seaward continuation of the beach profile until a depth of 4 m below the mean low water level (MLW). The subdivision of the shallow Belgian coastal zone follows this ecological zonation, focusing specifically on the intertidal and the shallow subtidal zones, and is defined by a landward boundary that follows the high water mark obtained by LIDAR observations of the Belgian coast in 2011 (data provided by the Agency for Maritime and Coastal Services: Coastal division – MDK) and a seaward boundary for the shallow subtidal foreshore of 1 nautical mile from the zero depth (0 m) bathymetric line (figure 1). The width of the subzones was chosen as fixed distances of 250 m for benthic components and wider distances (figure 1) of 3 km for birds, as these are highly mobile species (Derous et al. 2007c) .

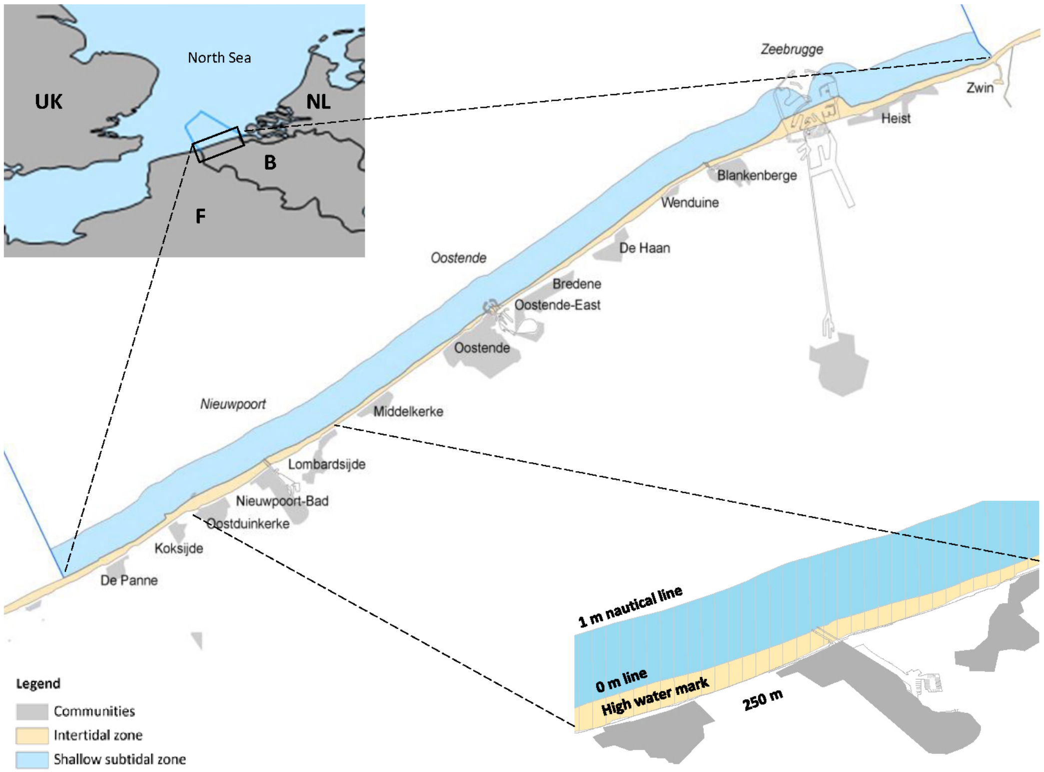


Figure 1: Study area of the Belgian coastal zone, with a distinction between the intertidal (light brown) and shallow subtidal zone (blue) and a detail showing the subdivisions performed for biological valuation

2.2 Databases

For the biological valuation of the shallow Belgian coastline, all available relevant data of benthos and birds in the intertidal and shallow subtidal zones during the period 1995-2011 were gathered (see Table 1 for references and sampling locations). The use of different sampling gears defines a differentiation among the benthic organisms: (i) macrobenthos – sampled with Van Veen grabs and/or quadrats and sieved over 1 mm; (ii) epibenthos – sampled with 5 mm mesh size trawl nets (or push nets) over the bottom; and (iii) hyperbenthos – sampled with 1 mm mesh size trawl nets (or push nets), approximately 1 m above the bottom. The birds data were gathered through morning countings on the beach. The sampling strategy used for each ecosystem component was always the same. All datasets include the geographical coordinates, the sampling gear used and the area sampled. Species richness data (number of individuals per species and per sample) were standardized into densities (number of individuals per m^2).

Table 1: References used for the integrated database per ecosystem component. Restricted to data collected in intertidal and shallow subtidal zones of the Belgian coast (mainly from unpublished data of Marine Biology, Ghent University)

Year of Collection	Sampling Locations	Reference
MACROBENTHOS		
1995	De Panne, Bray-Dunes and Koksijde	(De Neve 1996; Mouton 1996)
1997	De Panne ('De Westhoek'), Schipgatduinen, Koksijde, Paelsteenpanne, Ijzermonding, Lombardsijde, Raversijde, Spinoladijk, Vosseslag, Blankenberge, Fonteintjes, Zeebermduinen, Zeebrugge-bad, Baai van Heist, Heist, 'Zwin' and VNR 'Zwinduinen en -polders'	(Volckaert 1998; Speybroeck et al. 2005b)
2001	Knokke-Heist, Blankenberge, Wenduine, Oostende, Westende, Oostduinkerke, De Panne, Koksijde and Zeebrugge	(De Backer 2001; Boulez 2002)
2002, 2004, 2006, 2008, 2009, 2011	Lombardsijde, Nieuwpoort, Bredene, Koksijde-Oostduinkerke, Oostende (Centrum, Oosteroever, Vaargeul), Wenduine, Blankenberge, Mariakerke	Beach Nourishment Project[1]
EPIBENTHOS		
2001	Koksijde	(Buyle 2002)
2003	De Panne ('De Westhoek'), Ijzermonding, VNR 'Zwinduinen en -polders', Spinoladijk, Fonteintjes, Raversjide, Zeebermduinen, Schipgatduinen, Zeebrugge-bad, Baai van Heist, Paelsteenpanne	(Speybroeck et al. 2005b) (Speybroeck et al. 2005b)
HYPERBENTHOS		
1997	Lombardsijde	(D'Hondt 1999)
2001	Koksijde	(Buyle 2002)
2003	De Panne ('De Westhoek'), Ijzermonding, VNR 'Zwinduinen en -polders', Spinoladijk, Fonteintjes, Raversjide, Zeebermduinen, Schipgatduinen, Zeebrugge-bad, Baai van Heist, Paelsteenpanne	(Speybroeck et al. 2005b)
BIRDS		
2003 and 2004	De Panne ('De Westhoek'), Ijzermonding, VNR 'Zwinduinen en -polders', Fonteintjes, Raversjide, Zeebermduinen, Schipgatduinen, Zeebrugge-bad, Baai van Heist, Paelsteenpanne	(Speybroeck et al. 2005b)

---

[1] Beach nourishment project: Speybroeck et al. 2003, Welvaert 2005, Van Ginderdeuren et al. 2007, Vanden Eede et al. 2008, Vanden Eede & Vincx 2010, 2011, 2013

2.3 Biological Valuation Protocol

Method application

The purpose of marine biological valuation is to provide an integrated view on nature's intrinsic non-anthropogenic value of the subzones (but relative to each other), within a study area (Derous et al. 2007d). Unlike the previous applications of the protocol (Derous et al. 2007c; Forero Parra 2007; Rego 2007; Vanden Eede 2007; Weslawski et al. 2009; Pascual et al. 2011), the procedure used now is effectuated based on *R*, which is open-source software for statistical computing and graphics[2]. The *R* script for marine biological valuation has been recently developed by the Flanders Marine Institute (VLIZ), in Oostende, Belgium (Deneudt et al., submitted). Due to the fact that the protocol is flexible and subject to specific adaptations for each application, each of the steps, used for this valuation of the Belgian beaches, will be explained in the following subsections.

The *R* script for marine biological valuation guarantees general data quality control (geographical coordinates, dates, time and taxonomy, based on the World Register of Marine Species (WoRMS[23]). The set of assessment questions (table 2) relates the available biological data to the valuation criteria, being rarity and aggregation-fitness consequences, and to a specific organizational level of biodiversity. These valuation criteria were proposed by Derous (2007), after an extensive literature review and selection based in part on the framework for identification of Ecologically Significant and Biologically Significant Areas (DFO 2004) and expert judgment (Derous et al. 2007c). Biodiversity is not included as a separate valuation criterion, but linked to one or more of the selected valuation criteria using the 'marine ecological framework' created by Zacharias and Roff (2000).

Table 2: Set of assessment questions (Derous et al. 2007c)

Assessment Question	Categories of Species
Is the subzone characterized by high counts of many species?	all species
Is the abundance of a certain species very high in the subzone?	all species
Is the abundance of rare species high in the subzone?	rare species
Is the subzone characterized by the presence of many rare species?	rare species
Is the species richness in the subzone high?	all species
Is the abundance of ecologically significant species high in the subzone?	ecologically significant species
Is the abundance habitat-forming species high in the subzone?	habitat-forming species

The assessment questions are based on several 'categories of species', such as *all* species, *rare* species, *ecologically significant* species and *habitat-forming* species (table 2) giving differential value to some

---

[2] http://www.r-project.org/

[3] http://www.marinespecies.org/

species categories. For all species, species richness is calculated as the mean species richness per sample, location and subzone. Some sensibility to sampling effort bias cannot be excluded when using this calculation but it remains limited as the sampling method is uniform per ecosystem component and the species richness is calculated per sample. Derous et al. (2007) determined the criteria on rare species, by their percentage of occurrence in the samples: rare species were defined as those appearing in less than 5 % of the studied subzones. However, this threshold can be changed if properly justified as is the case when all species occur in more than 5 % of the subzones and as such no rare species can be determined. Since the protocol was designed to be flexible and aims at offsetting the relative differences between subzones as much as possible, the threshold was elevated to 10 %. Therefore, rare species were defined as those appearing in less than 10 % of the studied subzones. Habitat-forming species (HFS) were selected based on expert judgment, supported by the extensive literature existent on the role of such species dwelling the Belgian coast and continental shelf (Hiittel 1990; Rasmussen et al. 1998; Callaway 2006; Rabaut et al. 2007; Van Hoey et al. 2006; Rabaut 2011). Ecologically significant species (ESS) were selected based on expert judgment assessment and literature review (Van Hoey et al. 2005, 2007). It should be noted that subjectivity cannot be totally excluded in this BV method. A list of selected HFS and ESS, and the rationale behind this selection can be found in Appendices – Chapter 6 – Appendix A.

The assessment questions for each of the ecosystem components need to be translated into mathematical algorithms (see Appendices – Chapter 6 – Appendix B). Solving these algorithms yields a numeric answer to each assessment question, corresponding to a score translated into a semi-quantitative classification system of five value classes: *very low, low, medium, high* and *very high* BV. If there is no data to answer a specific question for a certain subzone, this is labeled 'NA'. An example of the scoring process described above can be seen in Appendices – Chapter 6 – Appendix C1. The scores for all assessment questions are added together per subzone, though separated for different ecosystem components and bearing in mind that each assessment question has been attributed an equal weight in the total score. These results are then illustrated in a BVM per ecosystem component.

The reliability of the assessed values for each subzone are noted with an attached label, perceptible in the final map (low, medium, high). Such label can either display the amount and quality of the data used to assess the value of a certain subzone (data availability) or it displays how many assessment questions could be answered per subzone given data availability (reliability of information). For example, when a certain question cannot be answered for one or more subzones, these subzones are scored on the basis of the remaining questions (the ones that could be answered), decreasing the completeness of the information and the reliability of the scoring. On a further level, when certain subzones lack data for one or more ecosystem components, these are valued based on the final score for the remaining available ecosystem components only, being less reliable than subzones valued based on all of the ecosystem components. An example of how data availability and reliability of information have been incorporated into the protocol can be seen in Appendices – Chapter 6 – Appendix C. These reliability labels and the BVMs should be consulted simultaneously as they allow us to identify knowledge gaps.

The total biological value of the subzones is determined by averaging the intermediate values for the different ecosystem components. An example of how to perform the final scoring can be seen in Appendices – Chapter 6 – Appendix C4. The results of the BV are then presented on a final BVM, where each subzone is assigned a color corresponding to its resulting biological value. Both reliability and availability labels of each subzone are displayed on the BVM by using different intensities of color or different fillings.

Using BV for solving space-use conflicts, e.g. flood risk and nature conservation

After a final BVM map of the Belgian coastal zone was obtained, the applications of this map were investigated. For the flood risk scenario, information regarding areas already identified as extremely vulnerable to coastal flood risk, and hence highly likely of undergoing coastal defence activities in the near future, has been collected and transformed into a spatial layer for analysis (see Appendices – Chapter 6 – Appendix D1). The final BVM was displayed along with this spatial layer. In order to analyse the results from a management perspective, spatial data joining was performed using the final BVM and the ten delimited Belgian coastal areas covered by Provincial Spatial Implementation Plans (PSIPs) (Appendices – Chapter 6 – Appendix D1).

For the nature conservation scenario, the final BVM was displayed together with the existing protected areas at the shallow Belgian coastal zone, under European (RAMSAR, Birds & Habitat Directive combined in the Natura 2000 Network – Special Areas of Conservation & Special Protection Areas) and National/Flemish legislation (marine/nature reserves, and protected dunes) (see Appendices – Chapter 6 – Appendix D2). Data were obtained from the interactive coastal atlas of the Flemish Region (Maelfait & Belpaeme 2009).

3. Results

3.1 BVM per ecosystem component

The BVMs for birds, macrobenthos, epibenthos, and hyperbenthos can be seen in Appendices – Chapter 6 – Appendix E1, E2, E3 and E4 respectively. The reliability indices, data availability and information reliability, per ecosystem component are depicted in the maps of Appendices – Chapter 6 – Appendix F1, F2, F3 and F4. Information reliability was maximal (*high*) for all subzones with data, meaning the chosen assessment questions for each ecosystem component could be answered in every subzone with data. Table 3 shows the number of subzones with data per ecosystem component. It is clear that the ecosystem component 'macrobenthos' delivers the highest amount of data for the total valuation. To check whether data availability is correlated with the valuation scores, a simple Pearson correlation was performed (table 3). No correlation could be detected.

Table 3: Number and percentage (%) of subzones with data, out of the total number of subzones per ecosystem component; Pearson correlation (*r*), with corresponding coefficient of determination (r^2) between data availability and BV scores per ecosystem component

Ecosystem component	**Total number of subzones**	**Number of subzones with data (%)**	***r***	**r^2 (%)**
Birds	42	10 (24 %)	0.30	0,09 (9 %)
Macrobenthos	463	124 (27 %)	-0.40	0,16 (16 %)
Epibenthos	463	11 (2 %)	0.73	0,53 (53 %)
Hyperbenthos	463	14 (3 %)	0.16	0,03 (3 %)
Total valuation	463	216 (47 %)	0.21	0,04 (4 %)

3.2 Integrated BVM

Figure 2 shows the final BVM for the Belgian coastal zone. The mosaic-like variability of scores is apparent and can also be seen in the BVM of macrobenthos (Appendices – Chapter 6 – Appendix E2). There is a clear difference in the amount of data collected to the west of Oostende if compared to the east and around 70 % of the shallow subtidal subzones with data scored *medium*, *high* or *very high.* Moreover, biologically *high* valued intertidal zones are not necessarily bordered by biologically *high* valued shallow subtidal zones and vice versa. Both final reliability indices, information reliability and data availability, are mapped together in figure 3. Most subzones displayed *medium* to *high* information reliability and have a *low* or *medium* data availability. *High/very high* biological values are consistently found in intertidal zones located immediately to the east of the three prominent Belgian harbours (figure 4).

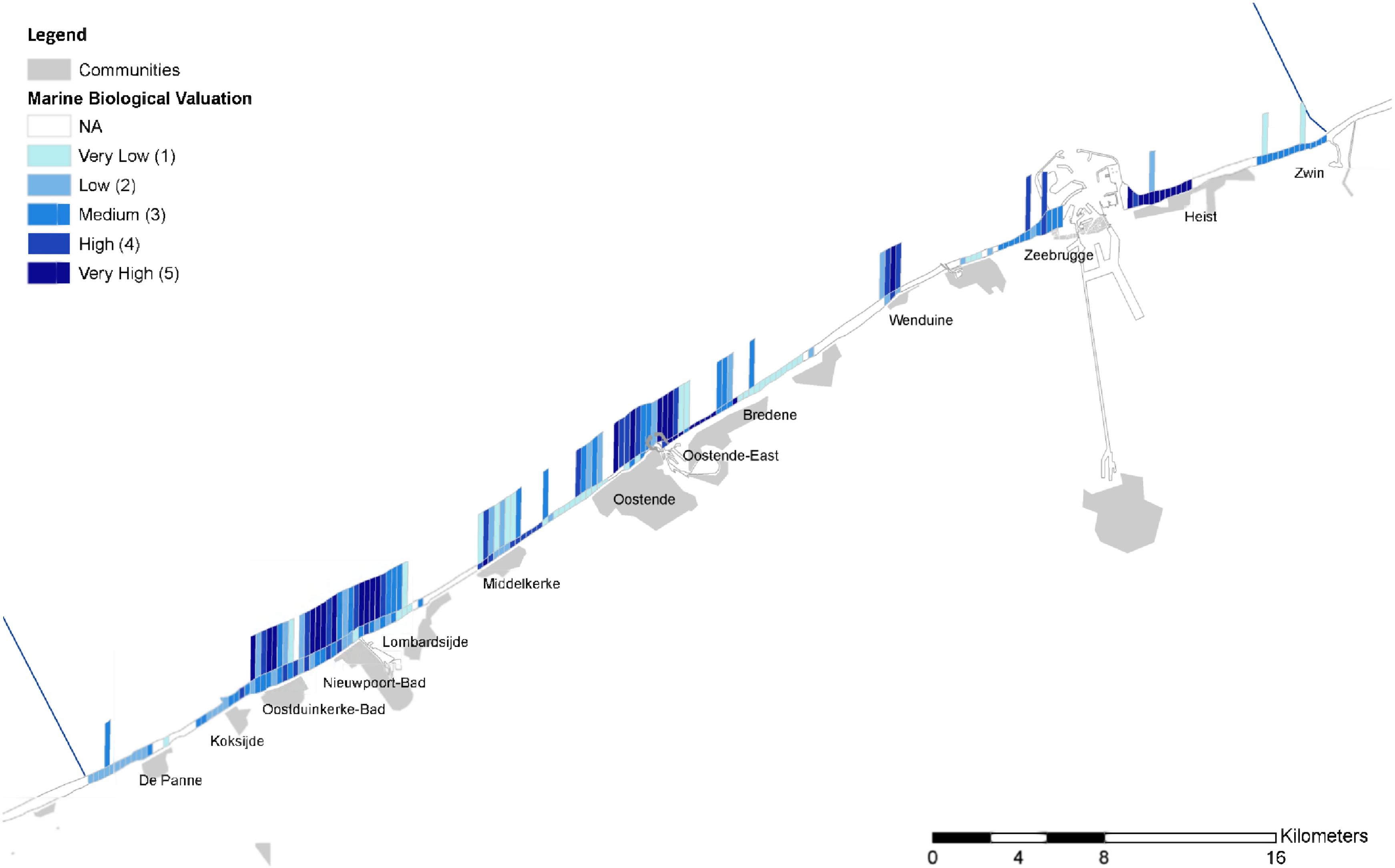


Figure 2: Final BVM for the Belgian coast

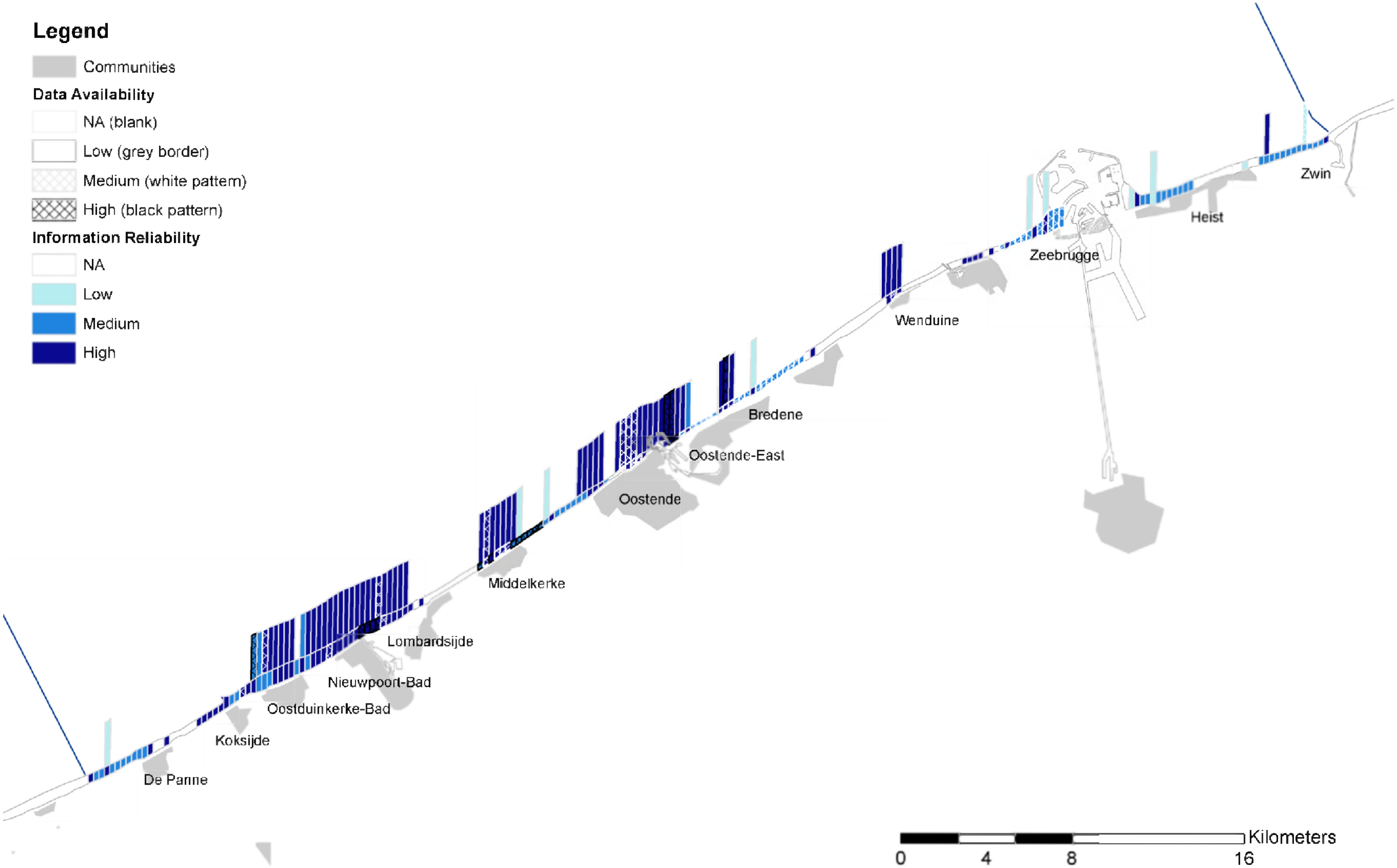


Figure 3: Final map depicting Information Reliability and Data Availability for the Belgian coast

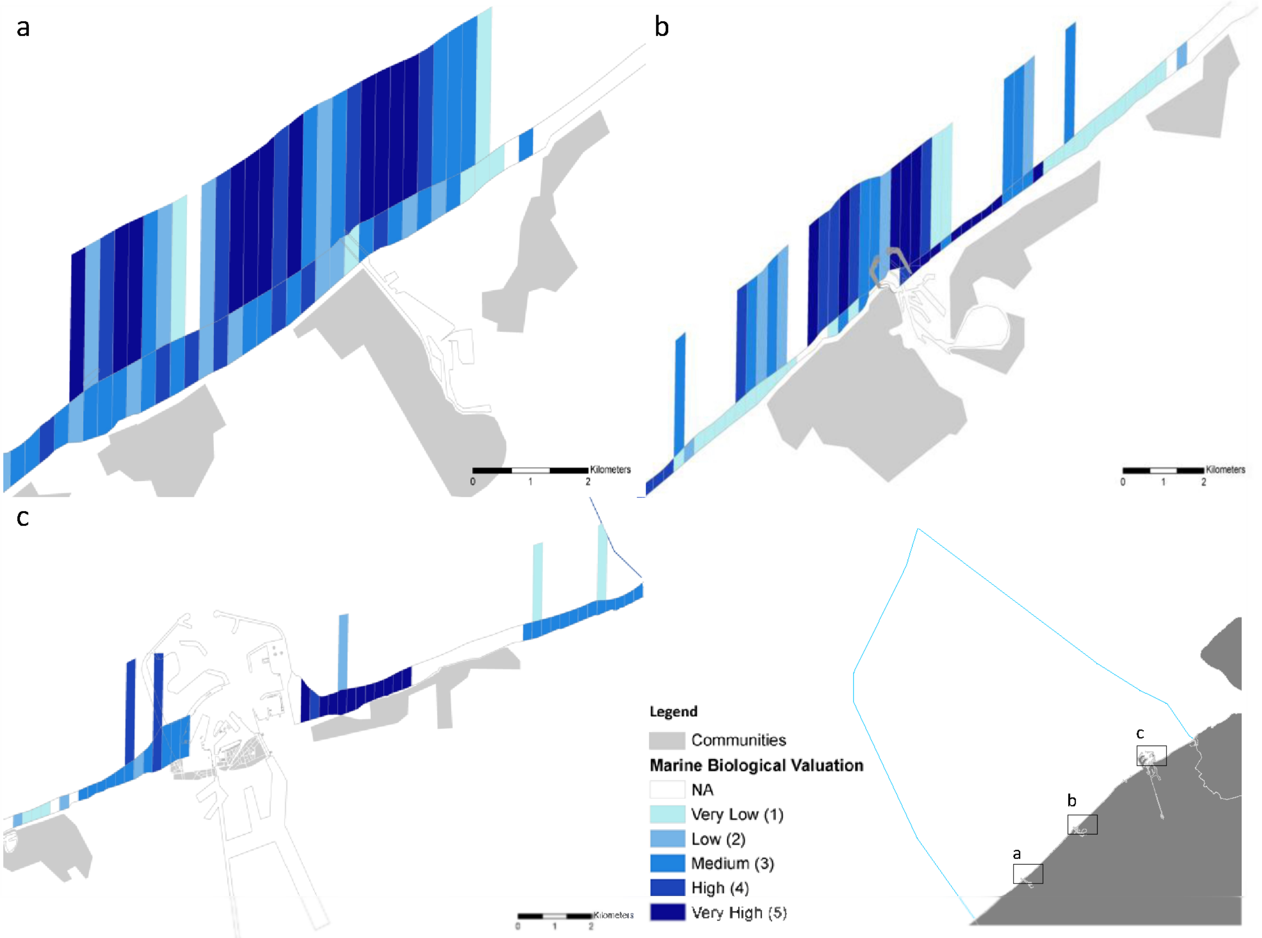


Figure 4: Detailed information on the BV of areas located at the east side of the main harbours at the Belgian coast: (a) Nieuwpoort (Lombardsijde); (b) Oostende (Oostende-East); (c) Zeebrugge (Baai van Heist).

3.3 Using BV for solving space-use conflicts

The final BVM was displayed along with areas under coastal flood risk (Appendices – Chapter 6 – Appendix D1) and along with the PSIPs. Since the PSIPs only cover the intertidal part of the Belgian beaches, the maps in figure 5 and Appendices – Chapter 6 – Appendix G only show the biological value of the intertidal area. Figure 5 focuses on the harbour areas as they have been given high coastal defence priority in the current Integrated Master Plan for the Flemish coast (Mertens et al. 2008) and the areas just east of the harbours seem to attain a *high/very high* biological value (figure 4). Areas for which no spatial plan exists, e.g. the beach of Lombardsijde, are commonly addressed as blank or undesignated areas (figure 5a) (Bogaert & Maes 2008). Areas sensible to coastal flood (in red) but lacking biological data (no color) were identified within almost all of the PSIPs (figure 5c). Areas sensible to coastal flood and displaying *high/very high* biological value were also identified (figure 5a and 5c and Appendices – Chapter 6 – Appendix G1 to G6).

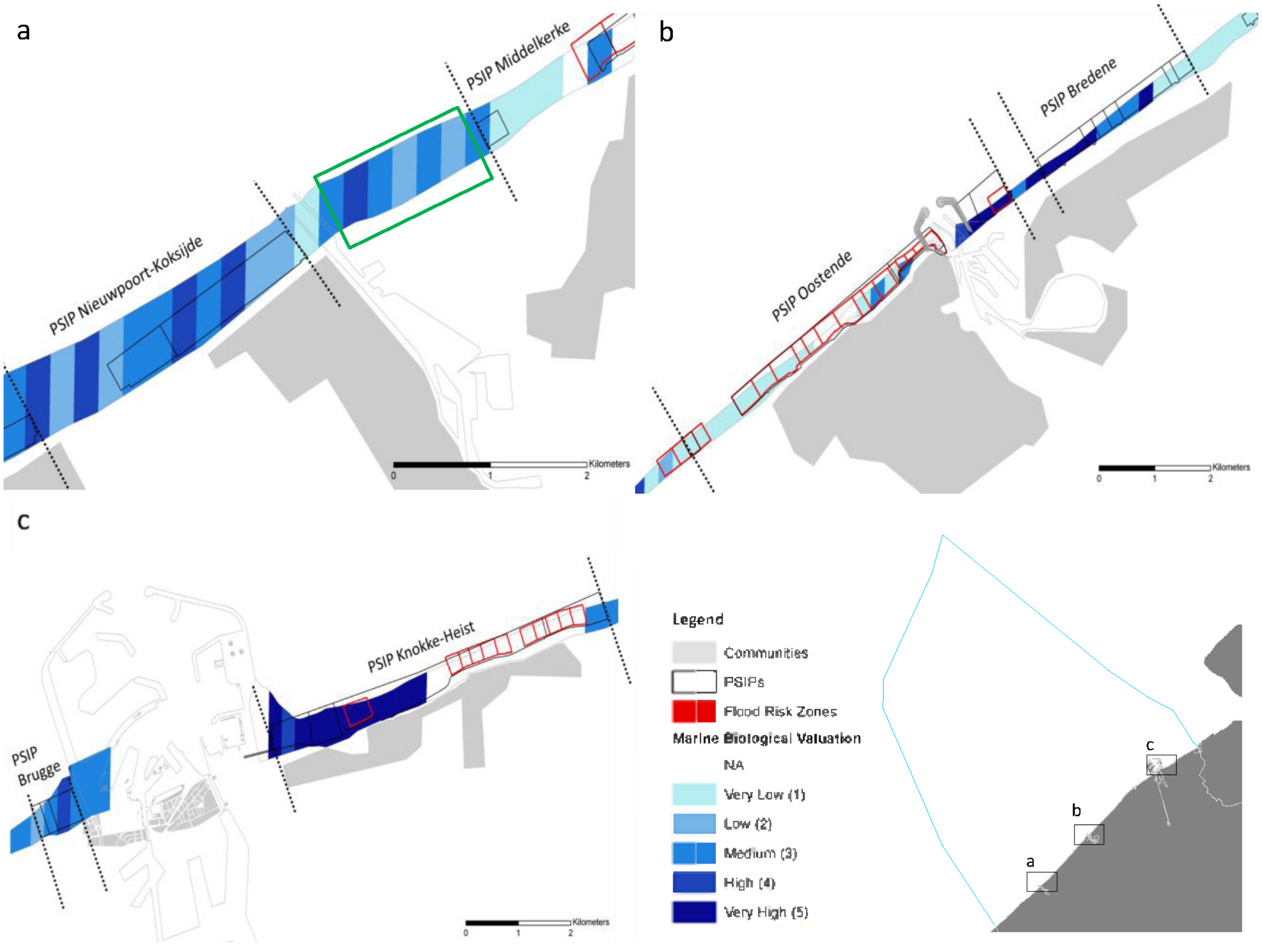


Figure 5: Detailed map with BV scores of intertidal areas located at the east side of the main harbours at the Belgian coast, inside PSIPs. Red indicates areas under coastal flood risk. The dashed lines mark the boundaries of each PSIP: (a) Nieuwpoort (Lombardsijde); the beach of Lombardsijde (green rectangle) falls inside an undesignated area as it is not covered by any PSIP (Maes & Bogaert 2008); (b) Oostende (Oostende-East); (c) Zeebrugge (Baai van Heist);

Considering the nature conservation scenario, all protected areas in the shallow Belgian coastal zone are displayed together with the final BVM (Appendices – Chapter 6 – Appendix D2). Detailed maps of the most important protected areas are shown in figure 6. Overall *low* BV scores for De Panne and 'De Westhoek' (figure 6a) and the *medium* intertidal value and *low* shallow subtidal value for 'Zwin' (Figure 6c) were certainly lower than expected. Lombardsijde beach area of the Flemish nature reserve 'IJzermonding' gets a *medium/high* intertidal score and a *very high* shallow subtidal score (figure 6b). The Flemish nature reserve 'Baai van Heist' (figure 6c) attained a *very high* BV.

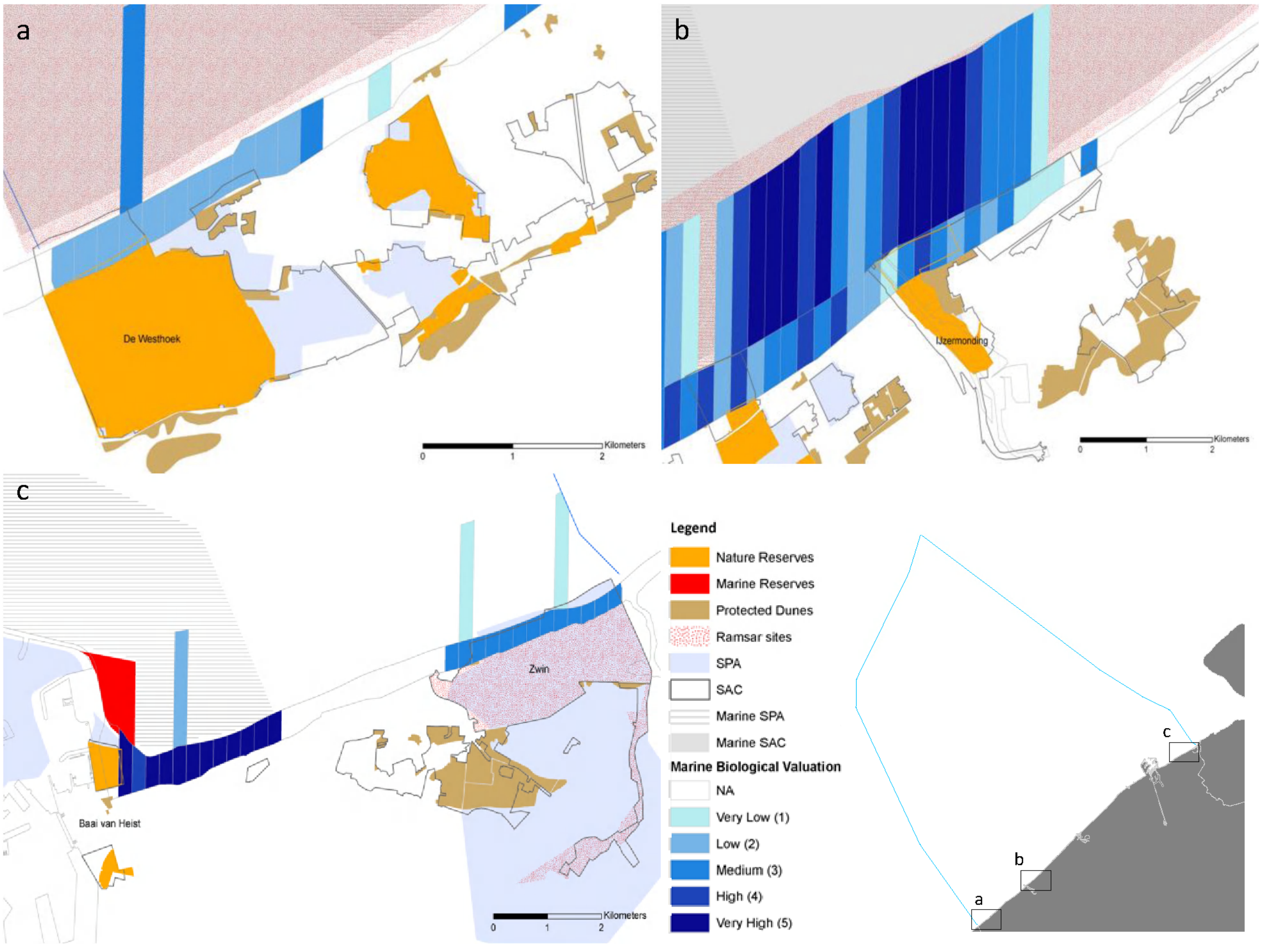


Figure 6: Detailed information on the BVM of protected areas located at the Belgian coast:

a) 'De Westhoek' (De Panne): only *low* intertidal scores were obtained despite its ecological importance;

b) Nature Reserve IJzermonding (Lombardsijde): *very high* valuation scores were obtained for the shallow subtidal waters adjacent to Lombardsijde beach, providing a visual support for the extension of the reserve seawards;

c) Zwin: an overall *medium* score, whereas intertidal subzones located near Baai van Heist have *high/very high* scores

4. Discussion

4.1 Integrated BVM of the Belgian coast

According to table 3, data used in this biological valuation covers almost half of the total study area (47 %), with the ecosystem component 'macrobenthos' delivering the highest amount of data for the total valuation. A simple correlation test was performed in order to check if the amount of data obtained in each subzone would be influencing the valuation score (table 3). Although a relatively higher r^2 was obtained for epibenthos (0.53), overall r^2 values were low and showed no strong correlation between the variables. The datasets used for epibenthos and hyperbenthos have been incorporated into the final valuation although they cover only around 3 % of the study area each (table 3), not allowing to deliver reliable results on these two ecosystem components as data availability and spatial coverage are just too far from satisfactory (Appendices – Chapter 6 – Appendix E3 and F3; E4 and F4 respectively). Table 3 also shows that the macrobenthos dataset is definitely the largest and as such contains data for the majority of the subzones. Most observed trends of the integrated BVM can be explained by taking a closer look at the BVM of macrobenthos (Appendices – Chapter 6 – Appendix E2).

Firstly, the mosaic-like variability of scores is apparent in both the final BVM (figure 2) as well as in the BVM of macrobenthos (Appendices – Chapter 6 – Appendix E2). This can be explained by the irregular and patchy distribution of sediments in the coastal zone due to minor across-shore and along-shore morphodynamic and morphological differences (Degraer et al. 2003b; Van Hoey et al. 2004; Vanden Eede et al. 2013, in prep.). Combined with the diverse topography of the Belgian coastal zone, this creates a wealth in habitats supporting a high capacity for varied benthic species assemblages (Van Hoey et al. 2004). Secondly, there is a clear difference in the amount of data collected to the west of Oostende if compared to the east. Furthermore, information at the eastern part of the Belgian coast is much scarcer, even for areas of great ecological importance such as 'Baai van Heist' or 'Zwin'. This is easily explained since the largest clusters of data (Lombardsijde, Nieuwpoort, Bredene, Koksijde-Oostduinkerke, and Oostende) were gathered during sampling campaigns in the framework of environmental assessments for beach nourishment projects, which are located mostly westwards of Oostende. Thirdly, around 70 % of the shallow subtidal subzones with data scored *medium*, *high* or *very high.* The breakdown of this result shows that these high values were obtained through questions related to Aggregation-Fitness consequences. Due to specific abiotic conditions, species richness and abundance of benthic organisms (Dewicke et al. 1998), shallow Belgian coastal waters are indeed known as nursery areas for a series of epibenthic macro-crustaceans and flatfish species (Rabaut et al. 2010). Nevertheless, for the question on ESS, higher values are mostly found in the shallow subtidal, suggesting that the ESS selected (Appendices – Chapter 6 – Appendix A) are perhaps not equally capturing intertidal and shallow subtidal communities. For example, although the *Abra alba* community is extremely important in subtidal waters (Van Hoey et al. 2005, 2007a), the emphasis given to this species by naming it an ESS might have caused an underestimation of the overall ESS scores for intertidal subzones. Finally, a mismatch between the intertidal and shallow subtidal scores can be detected. Biologically *high* valued intertidal zones are not necessarily bordered by biologically *high* valued shallow subtidal zones and vice versa. Although there seems to be a gradual transition in macrobenthic assemblages from the lower intertidal to the shallow subtidal zone (Defeo & McLachlan 2005; Speybroeck et al. 2008a), the differences in these assemblages between both zones are substantial enough to lead to different scores by applying the same assessment questions.

Reliability of information apprises the level of certainty of the obtained BV scores, whereas data availability pinpoints subzones with more or less sampling effort, indicating where future surveys should be undertaken (Pascual et al. 2011). Hence, increasing reliability and sampling effort leads to a higher level of certainty of the final BV scores. The assessment questions chosen aimed at addressing the type of data integrated in this valuation. Most subzones displayed *medium* to *high* information reliability and have a *low* or *medium* data availability (Figure 3).

4.2 Using BV for solving space-use conflicts

Coastal defence

In addition to the trends previously discussed, another important pattern has been observed. *High/very high* biological values are consistently found in intertidal zones located immediately to the east of the three prominent Belgian harbours (figure 4). The major wind-driven and tidal currents and waves at the Belgian coast have a southwest-northwest direction (van der Molen & van Dijck 2000; Speybroeck et al. 2008a). As a consequence of the net sediment transport towards the northeast, current-induced erosion causes depletion of sediments to the west of these hard structures and sediment deposition at the east side, in a kinematic process already described and commonly addressed in coastal geophysics (Deronde et al. 2004). The east side of these prominent hard structures (also referred to as lee-side) is a sheltered area where hydrodynamics are less intense and sand deposition occurs. Hence, it creates a wealth in soft bottom habitats and proper environmental conditions for benthic colonization, which goes in accordance with the observed pattern.

The spatial correlation between the final BVM and the PSIPs (Appendices – Chapter 6 – Appendix G1 to G6) showed that areas for which no spatial plan exists are commonly addressed as blank or undesignated areas (Bogaert & Maes 2008) and as such cannot be legally considered under the scope of coastal spatial management. Lombardsijde beach, part of the nature reserve 'IJzermonding' (Figure 5a and 6b), is such an undesignated area but its *high/very high* BV scores emphasize the importance of a full-coverage coastal network of PSIPs, leaving no room for undesignated areas. Areas sensible to coastal flood (in red) but lacking biological data (no color) are identified within almost all of the PSIPs, e.g. the beach zone between Knokke-Heist and Zwin (figure 5c). Areas sensible to coastal flood and displaying *high/very high* biological value are also identified (Appendices – Chapter 6 – Appendix G1 to G6), e.g. Middelkerke (figure 5a), Oostende Oosteroever (figure 5b) and Knokke-Heist (figure 5c). If coastal defence activities are to be performed in these areas, appropriate (mitigation) measures have to be drafted. This stresses the need for acquiring more relevant biological data at the unstudied areas with high coastal flood risk. Some critical steps for an ecologically good practice of beach nourishment should be taken, particularly in areas of *high/very high* BV, such as: (1) selection of nourishment techniques in respect to local natural values; (2) selection of sand nourishment based on the sediment composition of the targeted area (grain size); (3) avoiding drastic alteration of the beach slope; (4) execution of nourishment activities during periods of low beach activity of birds or other mobile organisms; and (5) favoring the selection of smaller, phased projects as opposed to a single, wide project (Peterson et al. 2000; Speybroeck et al. 2006a).

An alternative nourishment solution, known as foreshore nourishment, involves the implementation of parallel sandbanks along the entire coast just at the submerged foreshore. These sandbanks constantly supply sand to the beach zone after progressive tidal regimes (Misdorp & Terwindt 1997). However, intertidal communities are much more adapted to extreme sudden changes in environmental conditions than subtidal ones (Speybroeck et al. 2005a), making them relatively more resilient to anthropogenic interventions such as beach nourishment. Additionally, habitat continuity from the low intertidal zone to the foreshore (Degraer et al. 1999a) is disrupted by these sandbanks, hindering repopulation of the low intertidal zone by subtidal organisms. The *high/very high* BV obtained for most shallow subtidal zones along the Belgian coast (figure 2) further stress the need for caution when contemplating coastal defence measures such as foreshore nourishment. Overall, it can be concluded by saying that these results highlight the potential usefulness of BVMs for coastal and marine spatial planning in Belgium, particularly if considered as baseline maps under a solid decision support system (figure 7).

Nature Conservation

The BV protocol has achieved good results as a tool for the implementation of the Habitats and Birds Directives in the Belgian Part of the North Sea (Derous et al. 2007d) and as a framework in the environmental status assessment, under the European Marine Strategy Framework Directive (Borja et al. 2011; Pascual et al. 2011). It could also be used as a baseline map for the implementation of the European Water Framework Directive, as the protocol incorporates most of the biological and physical characteristics required by the Directive (Derous et al. 2007d). To stress the usefulness of the BV protocol as support tool for the proposal of new or the extension of already existing protected areas, the integrated coastal BVM was displayed along with the main protected areas at the Belgian coast (Appendices – Chapter 6 – Appendix D2). It is clear that not all areas with a kind of protection status, have a high ecological value, as defined with the BV method. This can be due to several reasons, as explained below.

For the area of De Panne, both the birds and macrobenthos BVM show a *low* BV (Appendices – Chapter 6 – Appendix E1 and E2) leading to overall *low* BV scores (figure 6a). Nevertheless, the ecological importance of De Panne and the grey dunes of 'De Westhoek' have been widely acknowledged in literature (Bonte et al. 2004; Provoost et al. 2004; Vandenbohede & Lebbe 2004) and the latter is even a reserve considered to be properly managed from an ecological perspective (De Ruyck et al. 2001; Houston 2003). However, literature also allocates the ecological importance of both areas to the ecosystem components vascular plants and terrestrial arthropods. Since there was insufficient data for these components, they were not included in this analysis. As such, no significant conclusions regarding the biological value of De Panne and 'De Westhoek' can be made due to the lack of information on vascular plants and insects, the sparse distribution of subzones with data and the absence of shallow subtidal information.

High data availability in the Lombardsijde beach area of the Flemish nature reserve 'IJzermonding' supports a *medium/high* intertidal score and a *very high* shallow subtidal score (figure 6b). However, the beach of Lombardsijde is an undesignated area on the PSIPs since it falls under military jurisdiction. It was proposed for special management plans in 2000 given its high ecological importance (Herrier & Van Nieuwenhuyse 2005). The *very high* shallow subtidal scores of Lombardsijde beach justify and underline the ecological importance of extending the beach reserve seawards (figure 6b) by providing a straightforward and visual message to support this advice (Van Nieuwenhuyse 2003).

The Flemish nature reserve 'Baai van Heist' (figure 6c) attained a *very high* BV due to the birds' valuation. This was expected, as the development of the harbour of Zeebrugge in the 1980s created vast areas of sandy, sparsely vegetated and relatively undisturbed coastal areas, mimicking natural processes and attracting a great number of coastal breeders (Stienen & Van Waeyenberge 2002; Stienen & Van Waeyenberge 2004; Stienen et al. 2005). In fact, the distribution of species such as *Sterna albifrons* (Little tern) is now almost exclusively limited to this area and adjacent beaches (Courtens & Stienen 2004; Stienen et al. 2005).

Protected under various legislations and directives, 'Zwin' is one of the most important protected areas of the Belgian coast (figure 6c). Its ecological relevance is related not only to its role as a breeding/feeding/aggregation site for birds but also to the presence of rare and important species (Devos 2008; Herrier & Leten 2010; Charlier 2011; BirdLife 2013). The *medium* value obtained for 'Zwin' (Figure 6c) was certainly lower than expected. The value is strongly influenced by the results for the birds, suggesting that the birds' data are not covering the real situation. The *low* score for the shallow subtidal subzone of 'Zwin' (Figure 6c) was only valued on the basis of epibenthos and hyperbenthos, scoring very *low* and *low*, respectively. Although little can be discussed for these components separately, previous literature suggested a decline of species richness and abundance for hyperbenthic communities under estuarine influence (Dewicke et al. 2003). Being in such proximity to the Scheldt estuary, this might very well be the case for 'Zwin' but without a better spatial coverage of data, this remains a mere speculative conclusion.

Clearly, more comprehensive datasets need to be incorporated in future biological valuations of the Belgian coast, particularly for the beach of De Panne and the 'Zwin' area.

BV as tool for ecosystem based-marine spatial planning at the Belgian coast

Ecosystem-based definitions and strategies should only be used if they are able to inform management actions based on an intrinsic assessment of biological value (Arkema et al. 2006). BV can be a valuable tool within the scope of EB-MSP at the Belgian coast as it allows for the integration of 'nature' at an early stage of policy implementation, for both coastal flood risk and nature conservation space-use conflicts. The BVMs permit informing management decisions at a level that is closer to stakeholders, significantly attenuating conflicts and enabling a transparent involvement (Pomeroy & Douvere 2008; Fleming & Jones 2012). Still, BVMs should be further considered together with other criteria related to socio-economic and political/legal preconditions within an integrative decision support system for spatial planning (Derous et al. 2007c)(figure 7).

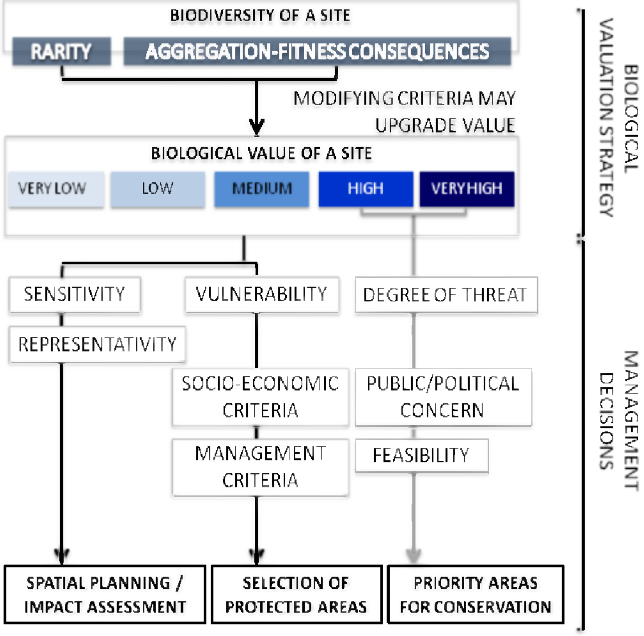


Figure 7: Overview of the BV concept and possible future steps to develop decision support management approaches (adapted from Derous et al 2007c)

Positive aspects of BV

When valuing marine biodiversity, it is important to capture as many attributes of biodiversity as possible, since biological structures and processes exist on different organizational levels (Zacharias & Roff 2000). Even though in this work the data available only addresses biological structures at the species/population and community levels, larger and more comprehensive datasets would eventually allow for the incorporation of all levels of biodiversity. Furthermore, the BV protocol also allows for the formulation and selection of different assessment questions, based on the ecological knowledge of the study area, and the inclusion of data regarding biological processes and functions (e.g. the presence of migratory routes and upwelling sites or overall productivity of a subzone), leading to more ecologically meaningful results.

BVMs only have a medium-term reliability and should be updated after a relevant period of time (several years) to reflect the medium-term variability in biological value and to meet the dynamics of the marine and coastal ecosystem. Unfortunately, the necessary high sampling intensity restrains a frequent update of BVMs making it impossible to reflect real inter-seasonal or inter-annual differences in biological value. For the time being, only maps based on data from a longer time period, giving a summary of the medium-term variability in value, can be developed (Derous et al. 2007d). A recalculation every five years seems appropriate given the amount of all new data that can be gathered within that time frame (Pascual et al. 2012).

Incorporating data on beach meiofauna, terrestrial arthropods and vascular plants could permit a more integrative and sound valuation of the coastal zone by addressing the beach ecosystem as a continuum from shallow subtidal waters to the foredunes. However, these ecosystem components are either only scarcely researched or restricted to the foredunes. In the latter case, this would hinder a good relative comparison between all studied zones (foredunes, intertidal and shallow subtidal zones). Limitations on data coverage can be overcome by mapping biophysical characteristics (Young et al. 2007) and subsequent habitat modeling based on, for example, grain size (Van Hoey et al. 2004; Degraer et al. 2008b; Willems et al. 2008), resulting in a sound extrapolation of benthic data to presently unsampled subzones.

Since the marine and coastal environment is very complex, several indicators have been designed to reduce the number of measurements and parameters that normally would be required to give an exact representation of the state of this environment. An indicator in ecology and environmental planning is defined as a component or a measure of environmentally relevant phenomena, e.g. pressures, states and responses, used to depict or evaluate environmental conditions or changes or to set environmental goals (Heink & Kowarik 2010). Indicators thus require detailed knowledge of what the natural state of a system should be, why the system is in a particular state, and which value-based criteria are necessary for applying the 'good' or 'bad' label (Mee et al. 2008). In general, indicators have to be SMART (specific, measurable, achievable, realistic and time-bounded) such that it will be apparent when they have been met, and when management measures have been successful. In moving towards a more functional approach, the need for indicators of overall health of the system still increases, at the expense of indicators of single aspects of the biota, e.g. species richness and biomass (Borja et al. 2010). Marine biological value is a multi-metric, integrative, system-level ecological indicator developed to be able to assess the intrinsic value of a certain area by integrating all available biological data on different organizational levels of biodiversity (from the species up to the ecosystem level) and for different ecosystem components (Derous et al. 2007d; Borja et al. 2011).

Limitations and Caveats of BV

The protocol followed in this work reflects the reasoning behind the development of the BV tool, and no fundamental changes to the original assessment questions and concept definitions (Derous 2007) have been undertaken. We highlight that misinterpretations could occur when the BVM is used without consultation of the reliability and availability maps, the underlying maps depicting the results of each assessment question separately per ecosystem component, the documentation of the valuation process or the integrated database. Despite these constraints, the availability of a BVM of the Belgian coast allows to answer policy questions related to the biological value of certain subzones in a transparent, objective way, where in the past, managers had to rely mainly on expert judgement (Derous et al. 2007a).

When first applied to the Belgian Part of the North Sea, species richness per subzone was corrected by applying a logistic regression analysis in which besides sampling effort (in terms of area surveyed), the distance to the coast and mean depth were also taken into account (Derous et al. 2007c). However, the BV protocol used here did not yet foresee for such correction, especially since distance to coast and mean depth would be irrelevant factors to be considered in the intertidal and shallow subtidal zone. For future applications, a correction for sampling effort differences among subzones could be designed and applied for questions related to species richness.

The relationship between the spatial coverage of data gathered and the number of subzones established strongly influences the selection for rare species in the BV protocol. Rare species in BV are defined as species appearing in less than 5 % of the studied subzones (Derous et al. 2007d), but this can be changed if properly justified. In this case, all species occur in more than 5 % of the subzones, resulting in a conflict within the selection of rare species. Therefore, rare species were defined as those appearing in less than 10 % of the studied subzones. This can be seen as a rather technical constraint of the protocol and it can be fixed by changing the calculation steps or changing the approach to the selection of rare species (Pascual et al. 2011). Clearly, further attention regarding this matter is fundamental to the successful improvement of the BV protocol.

5. Conclusions

The application of the biological valuation framework (Derous et al. 2007a; Derous et al. 2007b) for the shallow Belgian coastal zone was feasible and required minor adjustments. Spatial coverage and overall data availability were satisfactory and allowed for significant trends and patterns to be observed. Although the Belgian coast is entirely composed by sandy beaches, there is indeed biological diversity among distinct subzones and its intrinsic value needs to be properly assessed and taken into account. Spatial information on the intrinsic biological value of a given subzone within areas covered by PSIPs and/or coastal flood risk areas was presented in a straightforward manner, potentially enabling stakeholder's involvement. Similarly, BVMs provided a strong visual support to the proposal for the extension of some already existing nature reserves and to the need for more data to allow for significant conclusions regarding the biological value of other reserves. In both cases however, BVMs should be used along with other criteria defined within a sound decision-support system for spatial planning (Derous et al. 2007c). Important limitations to the applicability of this BV protocol have been identified, mostly related to the threshold for selection of rare species and the approach to calculating species richness. Notwithstanding these, the highlighted positive aspects strongly suggest that the potentialities of this integrative tool should not be underestimated. Further research on the applications of BV to coastal areas is still required to perfect and fine-tune the tool, enhancing the robustness of its results and consequently strengthening its application within spatial management strategies towards an integrative, ecosystem-based management of coastal areas worldwide.

Chapter 7: An ecosystem approach towards Belgian coastal policy

In this chapter, a general discussion, conclusions and future challenges are given. Beach research results are translated towards beach nourishment recommendations and policy guidelines for an ecosystem-based, integrated sandy beach management. Furthermore, beach fact sheets, criteria for a good ecological beach, a plea for a multi-disciplinary, integrated beach spatial plan and some future beach research suggestions have been made.

Abstract

The Belgian coastal zone hosts a complex of space-use and resource-use activities with a myriad of pressures impairing environmental conditions both on the coastline and on coastal waters. Specifically at the beach, predictions on sea level rise, intensified storms, accelerated erosion and flood risk for the North Sea have led to the drafting of the Belgian Integrated Coastal Safety Plan. The preferred coastal defence measure is beach nourishment as it safeguards the natural dynamics of the coast and has little impact on the beach ecology and tourism compared to other options. However, together with the multitude of human beach functions such as tourism and economic development, beach nourishment potentially threatens the natural characteristics of the beach ecosystem.

As management of the coastal zone is clearly a multi-faceted and complex endeavour, where the interests of several stakeholders need to be combined, coastal management desperately needs ecological dimensions. Hence, solid and meaningful biological and ecological information is needed. Clear and user-friendly management tools are essential to guide integrative and ecosystem-based strategies to sustainably manage ongoing space-use activities at the Belgian coast. From 1997 to 2011, relevant research data was gathered in 16 intertidal and 10 shallow subtidal coastal locations, over 8 years in 3 different seasons to (1) give an overview of the natural spatial and temporal variation in the Belgian coastal zone and (2) define the realized niches of the dominant intertidal and shallow subtidal macrobenthos. The *in situ* impact effects of an ecological nourishment were tested according to a Before After Control Impact (BACI) design (2008 – 2012) straddling the nourishment event (2009). The sediment preferences of the dominant Belgian intertidal beach macrofauna were experimentally tested both in single-species and combined-species conditions.

All these research results and data were used to (1) formulate research based guidelines for Belgian policy, especially regarding ecological beach nourishment, (2) develop an ecological model to predict the ecosystem response of beach nourishment scenarios at different trophic levels, (3) establish a scientifically sound and spatially based biological valuation of the Belgian coastal zone, using the marine biological valuation method (Derous et al. 2007a) and (4) produce beach records, encompassing all relevant data gathered on the 16 intertidal and 10 shallow subtidal studied coastal locations. These management tools will assist local decision makers and allow for the integration of 'nature' at an early stage of coastal policy implementation. Some future perspectives for Belgian coastal research are provided as well.

Keywords: beach nourishment, ecosystem based management, coastal policy, guidelines, tools, monitoring

1. Introduction

The Belgian coastal region is an extremely valuable social, ecological and economic environment, consisting mostly of sandy beaches with sea walls in front of the cities and dunes in between. The main long-term threats are linked on the one hand with the social and economic use of the land on or immediately behind the dunes and on the other with natural impacts like erosion and climate change. Storms and associated erosion present the most substantial universal hazard to beach ecosystems (McLachlan et al. 2013). Sea level rise due to climate change can cause flooding, accelerated coastal erosion and the loss of flat and low-lying coastal regions (Brown & McLachlan 2002), like the Belgian coastal region. Furthermore, it increases the likelihood, frequency and intensity of storm surges, enforces landward intrusion of salt water and endangers coastal ecosystems and wetlands. Projections by the Intergovernmental Panel on Climate Change (IPCC) for the end of the 21st century suggest a sea level rise between 18 – 59 cm above the average 1980 – 2000 level, with indications it might be even higher. This would cause waves to increase by 2 m at our coasts (Doody et al. 2004).

The threat of coastal flood risk might be not acute but the set-up of a precautionary principle driven design is vital from an ecological point of view. Within the Belgian legal system, coastal safety is already regarded as the most important priority in the decision making process for the Belgian coast, being a prime reason of public health concern. The recently approved Integrated Coastal Safety Plan (10 June 2011) contains a series of measures and alternatives to be taken between now and 2050, guarding against the dangers of a superstorm and preventing present and future flooding (Mertens et al. 2008). For the next years, Belgian beaches will thus face a multitude of coastal defence activities, including large-scale long-term beach nourishment projects.

Meanwhile, international and European legislation is trying to counteract the deterioration of the coastal environment in terms of biodiversity and ecosystem functioning. The coastal conservation and protection is laid down in the EU Water Framework Directive, the EU Bird and Habitat Directive, and in international treaties and recommendations. The Belgian sandy coast is indeed much more than just a biological desert providing a natural defence against the sea. Therefore, management of beaches should involve more caution than is often the case. Even though a significant proportion of the beach inhabiting organisms is adapted to the naturally high environmental stress of tides, waves and winds, this adaptation has its limitations (Speybroeck 2007). The Belgian coastal zone is also an important nursery area for juvenile fish and birds and falls under the habitat type 1140 (Mudflats and sandflats not covered by seawater at low tide; cf. NATURA 2000) of the European Habitat Directive (Annex II). Thus far, six intertidal beach zones have been proposed for the Natura 2000 framework. However, no restrictions on activities have been formulated yet and possible protection of these zones has not been incorporated in the Provincial Spatial Implementation Plans (PSIPs). There are some agreements, based on European legislation (precautionary principle in Convention on Biological Diversity), that stipulate that the loss or degradation of intertidal habitat due to impact activities like beach nourishment should be discussed and accordingly compensated. Depending on the impact effects, two compensation options are available: (1) when the nourishment activity is strongly impacting the area, leading to severe loss of valuable habitat, it can be compensated by creating ecological valuable habitat on another location or (2) by considering an ecological alternative nourishment, where the majority of the nourishment characteristics (sediment used, slope of the nourishment, timing, techniques) are ecologically adjusted. Therefore, guidelines for ecological adjustment of beach nourishment, leading to a minimization of the impact on the beach ecosystem are needed. However, a suitable assessment for every beach nourishment remains needed, in accordance with European legislation (CBD, Convention on Biological Diversity, 1992; Precautionary principle; Directive on Enviornmental Impact Assessment (Commission 2011)).

Flood risk management and coastal defence can deliver benefits for both people and nature. According to the 2002 EU Recommendation on Integrated Coastal Zone Management, the 2008 Marine Strategy Framework Directive and the recent proposal for a Directive establishing a framework for maritime spatial planning and integrated coastal management (Commission 2013b), the management of the coast has to be based on a comprehensive and integrated ecosystem approach (Janssen & Mulder 2005). This environmental management approach recognizes the full array of interactions within an ecosystem, including humans, rather than considering single issues, species or ecosystem services in isolation. It aims at maintaining an ecosystem in a healthy, productive and resilient condition so that it can provide the services humans want and need (McLeod 2005). Ecosystem based spatial planning is then the tool for its implementation by bridging the gap between science and practice and filling the current need of both governments and non-governmental organizations for more practical management tools (Douvere 2008). In essence, it is an integrated planning framework that informs the spatial distribution of activities in the area of interest in order to support current and future uses of its ecosystems and maintain the delivery of valuable ecosystem goods and services for future generations in a way that meets ecological, economic and social objectives (Foley et al. 2010). Since natural coastlines do not respect political borders, these coastal spatial plan initiatives should further develop into cross-border and regional plans to fully implement a sustainable coastal ecosystem based management.

Belgium was among the first countries to implement an operational, multiple-use marine spatial plan, covering its territorial sea and exclusive economic zone (Maes et al. 2005b). It aims at achieving both economic and ecological objectives, including the development of offshore wind farms, the delimitation of marine protected areas, a policy plan for sustainable sand and gravel extraction, the mapping of marine habitats, protection of wrecks valuable for biodiversity, and the management of land-based activities affecting the marine environment. Unfortunately, the Belgian beaches, from dunes to the mean low water level (MLW), are not (yet) incorporated in this plan because of the Belgian legal intricacies (figure 1). The federal government has jurisdiction over the entire Belgian part of the North Sea, including the Exclusive Economic Zone (EEZ, more than 12 nautical miles) and the Territorial Sea (between MLW and 12 nautical miles). Within this regard, the shallow subtidal coastal zone (between MLW and 1 nautical mile) falls under federal jurisdiction. The Flemish regional authority governs the inland territory, including inland waters and estuaries, and the coastal waters above the MLW, including the intertidal coastal zone. Environmental and coastal defence policy competences are thus shared between the federal and regional levels (Herrier et al. 2005). The Coordination Centre for Integrated Coastal Zone Management encourages and promotes sustainable and integrated coastal management by allowing a platform to discuss cross-sectorial themes between the federal, Flemish and provincial policy levels (Cliquet 2001).

Figure 1: Coastal legal system in Belgium (1NM: 1 nautical mile; MLW: mean low water level; MHW: mean high water level; WFD: Water Framework Directive, MSFD: Marine Strategy Framework Directive; ICZM: Integrated Coastal Zone Management) (Laporta 2012)

The current study highlights the recent Belgian beach research and proposes four coastal management tools: (1) a scientifically sound and spatially-based biological valuation map of the Belgian coastal zone, using the marine biological valuation method (Derous et al. 2007a), (2) an ecological model that can predict the ecosystem response of beach nourishment scenarios at different trophic levels, (3) research based guidelines for Belgian policy, especially regarding ecological beach nourishment and (4) beach fact sheets, encompassing all relevant data gathered on the 16 intertidal and 10 shallow subtidal studied coastal locations.

2. Beach research

In order to protect the coastal environment, one has to know what to protect (Janssen & Mulder 2005). Good knowledge of the Belgian beach ecosystem in both the intertidal and shallow subtidal zone provides us with a baseline condition. This is the condition of the natural resources and ecosystem services that would have existed if no impacts had occurred, estimated on the basis of historical data, reference data or control data. To this end, data gathered between 1997 and 2011 in 15 intertidal and 9 shallow subtidal coastal locations over 8 years in 3 different seasons was analysed (chapter 2). The partitioning of macrobenthic community structure within the Belgian beach ecosystem showed a large within beach variability, linked to elevation on the beach and median grain size of the sediment, in both the intertidal and shallow subtidal zone. Several spatial and temporal trends in abiotic factors (overall median grain size between 150 and 300 µm) and in macrobenthic species richness (intertidal: 0 – 19 species; shallow subtidal: 0 – 28 species), abundance (intertidal: 0 – 3989 individuals.m^{-2}; shallow subtidal: 0 – 1949 individuals.m^{-2}) and biomass (intertidal: 0 – 7 g AFDW.m^{-2}; shallow subtidal: 0 – 246 g AFDW.m^{-2}) were measured. The mean macrobenthic abundance in the intertidal and shallow subtidal zone fluctuates between 0 and 350 individuals.m^{-2} over the years. Between these minimum and maximum values the natural variation on Belgian beaches runs its course. Furthermore, the observed niches and interpolated occurrence of the dominant macrobenthic species of the Belgian beaches were defined as the area where these species really live during low tide, characterized by elevation on the beach and median grain size of the sediment (figure 2).

To document environmental impacts and assess the effectiveness of management actions, the natural noise in the system should be taken into account in order for any impact signal to be determined. All these findings assess the natural variability on the Belgian beaches and increase the strength, efficiency and accuracy of monitoring strategies to detect possible impact effects on the Belgian beaches.

Adaptive ecosystem based coastal management is the best mind-set for ecological intervention. We cannot control or manage populations or ecosystems, rather we control the level of human interaction with an intervention in natural systems. Optimizing the technical aspects of future nourishment projects is as such indispensable to maintain an ecologically healthy beach ecosystem. From March until September 2009, a nourishment was performed on the Belgian beach of Lombardsijde under optimal ecological conditions, e.g. phased nourishment project with nourished sand closely matching the original sediment and only moderate beach profile changes. The timing was suboptimal although the nourishment was originally planned during the more preferable winter season. In chapter 3, the *in situ* impact effects of this 'ecological' nourishment were tested according to a Before After Control Impact (BACI) design (2008-2012) straddling the nourishment event (2009) in Lombardsijde. As a temporal control, before-impact baseline data is necessary while selection of an area that will remain unimpacted serves as a spatial control (Grober 1992; Smith et al. 1993; Underwood 1994; Schlacher et al. 2012). A wider, higher and flatter intertidal beach with coarser sediment (from 216 ± 3.6 µm in 2008 to 280 ± 8.9 µm in spring 2010) was created and no return to the pre-nourishment abiotic conditions was visible three years after nourishment. The sediment grain size distribution had changed as well, showing slow recovery in the three post-nourishment years. The analysis of the macrobenthos community structure showed that nourishment under ecological optimal conditions does not yield any significant effects on both the intertidal and shallow subtidal beach ecosystem 6 months after the nourishment. Within this time frame, the macrobenthos community had recovered from the impact of the ecological nourishment. Ecological nourishment thus proves to be the least ecologically damaging way of combating erosion, compared to all other coastal engineering activities.

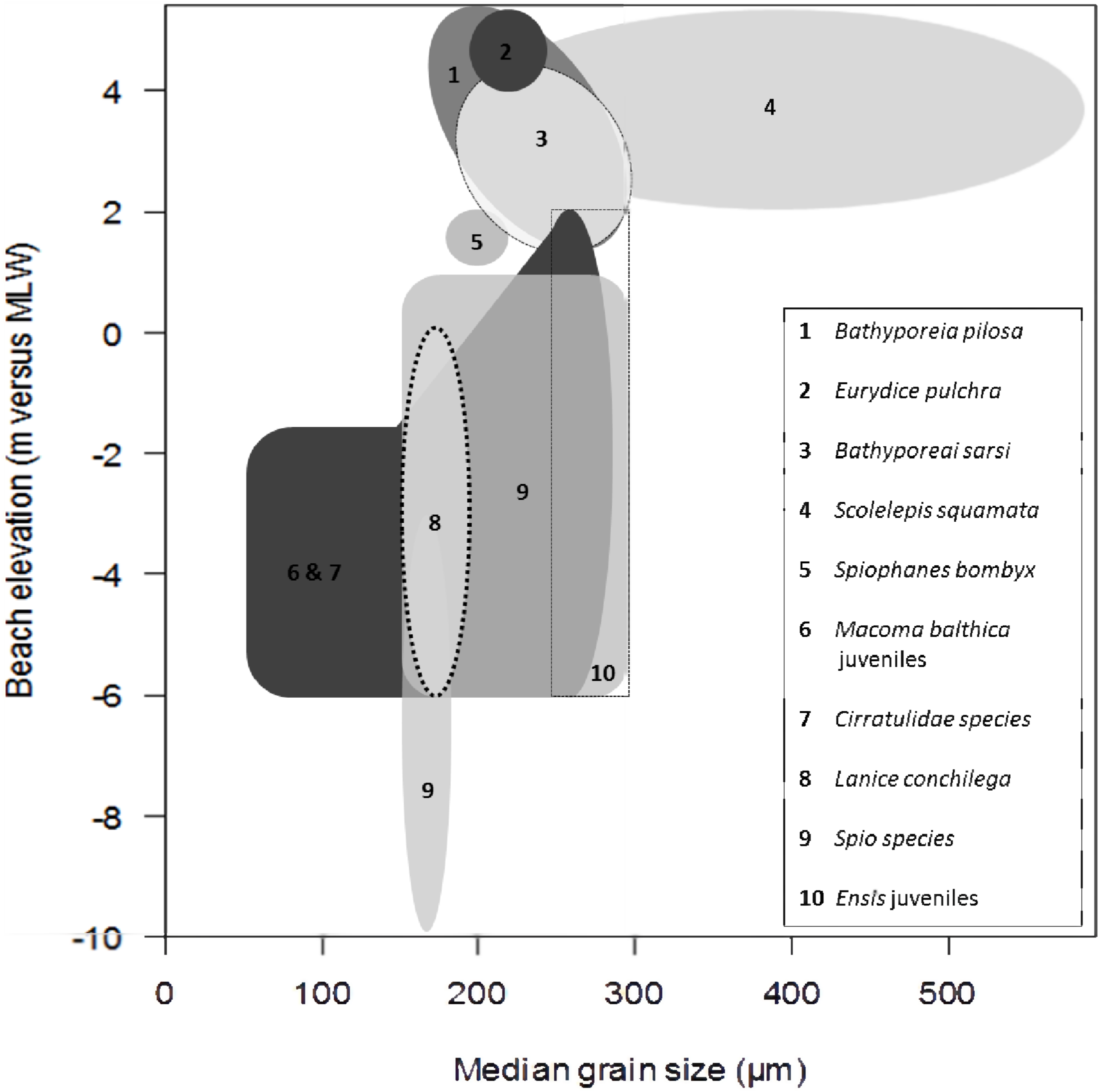


Figure 2: Observed niche and interpolated occurrence of the dominant Belgian macrobenthic species along an ‘average’ Belgian beach transect

The sediment preferences of the dominant Belgian intertidal beach macrofauna were experimentally tested both in single-species and combined-species conditions in chapter 4. Results of the experiments indicated that *Bathyporeia pilosa* and *Eurydice pulchra* prefer the finest sediment (< 250 µm), while *Bahyporeia sarsi* (250 – 355 µm) had a broader preference and also occurred in medium-coarse sediments. Interspecific competition between the sympatrically occurring amphipods was found to change the sediment selection of the amphipod *Bathyporeia pilosa* towards the coarser sediments where *Bathyporeia sarsi* occurred in lower frequencies. The polychaete *Scolelepis squamata* had the broadest preference (355 – 500 µm) and even showed a high occurrence in coarse sediments that are not naturally occurring on the Belgian sandy beaches. These preferences imply that beach nourishment with coarse sediment will have a major effect on *Bathyporeia pilosa* while effects of coarse sediments on *Scolelepis squamata* will be minor.

3. Research based guidelines for Belgian coastal policy

3.1 Monitoring guidelines

Good research starts with the collection of baseline environmental data to identify as much unknown variables as possible. Studies should be done at the appropriate spatial and temporal scales depending on the questions to be answered and they should be standardized to assure similar methodology throughout the long term dataset. Research surveys must be designed to take into account the fact that benthic fauna is extremely patchy in distribution and abundance (Eleftheriou & McIntyre 2008). The sampling method has to quantify, minimize and/or explain all scales of variability in the benthos and provide a solid base for ecological comparisons across, along and between shores or from year to year (Hayward 1994).

These few guidelines would allow for a good follow-up of beach conditions. On each beach, a sequence of samples (0.1026 m^2 sampling size with a depth of 0.15 m) arranged at uniform time intervals along the across-shore gradient has to be taken, both in the intertidal (15 samples with quadrat frame) and shallow subtidal zone (15 samples with Van Veen grab), preferably during two seasons, being spring and autumn. Physical characteristics of a beach act on beach macrobenthos at a single place and on a single time (Hacking 2007), making physical data at the time of sampling essential to predict beach macrobenthic communities. A sample of sand, sieved and analysed, gives an immediate and quite precise insight into the ecology of the habitat at the sampling location. The abiotic factors measured, especially beach height and median grain size, should provide a good overview of the physical characteristics of the Belgian beaches.

Traditionally, sandy beach ecologists have sampled along transects, e.g. shore-normal lines of samples from MHW to MLW. Due to the spatial autocorrelation among individual transect samples, they cannot be treated as replicates from one another. Data obtained from an individual transect should be pooled, thereby integrating the across-shore variability and providing a point estimate without confidence intervals. However, other disciplines often use stratified random sampling designs to quantify sources of variability in community descriptors along environmental axes. This technique allows for random sampling site selection within strata, horizontal layers of material harbouring a similar community, rather than by investigators, hereby ensuring that they are representative, unbiased and can be extrapolated to show the ecological condition of the entire stratum. Each set of macrobenthos samples collected within a stratum, is considered to consist of replicate samples of that stratum and should provide a representative view on the macrobenthos of the stratum. In theory, the samples within the intertidal or shallow subtidal zone could be regarded as replicates of one another. In reality, there is a macrobenthos zonation gradient present on the Belgian beaches, leading to smaller groups of 'replicates' according to the different beach height zones. The stratified random sampling approach is, however, often considered impractical on beaches because across-shore strata are difficult to define a priori (Schlacher, Schoeman et al. 2008). According to figure 2, two big strata can be detected in both the intertidal (between 2 and 6 m above MLW and between 0 and 2 m MLW) and shallow subtidal zone (between 0 and -6 m above MLW and below -6 m above MLW).

The number of samples taken in a given area is always a compromise between having sufficient replication at any site to allow statistical testing and having a wide enough coverage of sites in time and space to answer questions about the temporal and spatial patterns in the benthos (Gray & Elliott 2009). Capturing spatial and temporal variability on sandy beaches also requires true replication of sampling stations over appropriate scales. No single sample size is appropriate for all quantitative ecological studies but several authors (McLachlan 1983; Schlacher, Schoeman et al. 2008; Schoeman, Nel et al. 2008) postulated an aggregate area between 0.25 and 5 m^2 for accurate sampling to avoid underestimation of species richness. Using the quadrat frame and Van Veen grab (surface area, 0.1026 m^2), three replicates would suffice to reach the bear minimum. Replication of samples was however not feasible due to time constraints on work effort. Schlacher et al. (2008) suggest taking samples to a minimum depth of 0.25 m to capture the largest possible fraction of resident organisms. The samples in this study were taken to a maximum depth of 0.15 m.

Researching a greater number of transects and replicating samples in the intertidal and shallow subtidal zone, at different seasons, seems to be unnecessary to characterize the macrobenthos zonational patterns but could provide statistical power to detect these patterns (Elliott, Degraer et al. 1997). Moreover, conventional parametric approaches will probably be confounded by the autocorrelation in the abiotic and biotic variables, violating the assumptions of parametric analysis, making non-parametric analyses the preferred statistical option.

To research ecological impact of beach nourishment, it is advised to monitor at least two unimpacted beaches parallel to the impacted beach to provide for a reference framework. Sampling on all beaches should begin at least a year prior to nourishment during spring and autumn. Sampling should be restarted as soon as the nourishment activities are finished. A more intensive sampling scheme during the first year (1 week, 1 month, 3 months and 6 months) could monitor the short term development of the nourished beach while long term effects should be researched up to 3 years after nourishment.

3.2 Nourishment guidelines

In many instances, it is still assumed that the only reason for ensuring apparently pristine beaches is to attract tourists and holiday-makers. Technical aspects were therefore dominant in taking management decisions for coastal defence. Easily available sand with coarse grain size and a rather steep, more stable beach slope were the standards of any beach nourishment project. It has been shown (chapter 2) that Belgian beaches do harbor a healthy beach ecosystem, when given the chance. The research conducted in chapters 2, 3 and 4 provide us with sufficient findings to formulate guidelines for ecological adjustment of beach nourishment, based on the initial set of guidelines recommended by Speybroeck et al. (2006).

Nourishment sediment characteristics

Every effort should be made to ensure that the nourished sediment is similar to that occurring naturally, in hydraulic properties and characteristics, including grain size (McLachlan 1996; Hamm et al. 2002), clay/silt portion and shell content (Peterson & Manning 2001), to minimize environmental impacts (Greene 2002). Nourished sediment should be non-contaminated (Essink 1999). As Belgian beaches have an average sediment grain size range of 150 – 300 µm and all dominant macrobenthos do show a preference to these sediments (chapter4), it is advised to use fine to medium sand for beach nourishment (< 300 µm). The total amount of nourished sediment should be kept as small as possible (Speybroeck et al. 2006a).

Beach profile

In order to protect the biota, beach profiles should be changed as little as possible (Short & Wright 1983; Defeo & McLachlan 2005; McLachlan & Dorvlo 2005). Severe profile change will impact the ecosystem and determine the efficiency and the lifetime of the nourishment. Certain profiles can favor or reduce specific species and their habitats. These effects will only be temporarily as the profile will ultimately evolve towards the pre-nourishment conditions. A very steep slope however will enhance the risk of a complete community shift on the intertidal beach.

Belgian beaches are characterized by gentle slopes and fine sediment and harbor a specific beach ecosystem (Speybroeck et al. 2008a) while beaches with steep slopes and coarse sediment are inhabited by a less species-rich macrobenthic community (McLachlan & Dorvlo 2005). When the morphodynamic features of a beach are changed to such a degree that they resemble the features of a reflective beach, e.g. steep slope and coarse sediment, a shift to the less species-rich alternative community is very likely.

Nourishment location

As the Belgian sandy beaches can be considered as one ecosystem, disturbances on a local beach can be counterbalanced by the complete system. However, the different harbor inlets also divide the Belgian beach ecosystem into separate parts. Although the across-shore zonation is stronger than the along-shore differences (chapter 2), it seems wise to maintain a precautionary approach regarding the resilience of the entire Belgian beach ecosystem.

Due to the high technical effort characterizing beach nourishment, only a limited amount of the beach area (1 – 2 km) is impacted at once. These nourishment dimensions enable species to escape to adjacent areas and species from source populations on other beaches to recolonize the nourished beach. Hence, alternation of impacted and non-impacted beaches and phased nourishment (nourishing only parts of a beach at one time thereby expanding the nourishment area slowly) is essential for the maintenance of a healthy and well-balanced beach ecosystem.

Both foreshore and backshore nourishment are no real alternatives for beach nourishment. The impact effects of foreshore nourishment are not yet known and the shallow subtidal beach zone is a noted refuge, nursery and feeding area for epibenthos and hyperbenthos (Beyst et al. 2001b).

Moreover, foreshore nourishment becomes only effective after three to five years, providing no effective defence against short-terms coastal defence threats, for instance predicted storm or flooding events. Backshore nourishment involves sand deposition at the dune foot. Unfortunately, this sand is easily removed by waves and winds, creating a steep beach slope (Harte et al. 2002).

Nourishment timing and recovery period

When scheduling beach nourishment operations, it is important to avoid the breeding and recruitment season of all beach inhabitants, e.g. infauna, macro-crustaceans, marine fish and birds, since their occupancy of the intertidal beach and their recovery rates are then at their highest (Speybroeck et al. 2006a). The most opportune time of year for carrying out such work is during the winter months, as the reproductive cycle of most species begins in March and can extend beyond October. That way, the freshly nourished beach can quickly be recolonized by the recruits and seeds when reproduction starts in spring. As the winter period is also the less touristic season, it is considered the best period for nourishment both from a touristic and ecological point of view.

Several short nourishment projects in time (minimum one week in between) and space (leaving beach strips unnourished) are preferred over broad-scale, long lasting ones, especially in areas where short term morphological changes are unpredictable (Hillen & Roelse 1995)

If no further nourishment projects or other pressures are impacting the beach, the system should evolve towards the pre-nourishment conditions although it remains impossible to predict the timeframe of that evolution. Not only the specific characteristics and timing of the nourishment are determining factors, but also the specific features of the beach ecosystem. Recovery will only take place if the nourished beach possesses the right characteristics for planktonic dispersing larval stages and passively migrating adults to settle upon. However, some species can recolonize faster than others. Especially marine animals with pelagic larvae are swift colonizers, while crustaceans with brood care are slow colonizers. The post-nourishment monitoring data (chapter 3) suggest that at least in some cases nourishment under ecological optimal conditions can show no significant effects in the macrobenthos community structure 6 months after the nourishment (2010S). Within this short-term time frame, the macrobenthos community recovers from the impact of the ecological nourishment, showing no dispersal or recruitment limitations.

Nourishment technique

On Belgian beaches, most of the nourishment projects supply sand on the upper zone of the beach through pipes while bulldozers further divide the sediment over the entire beach. Schlacher et al. (2012) showed that this approach can have large ecological impacts that vary with elevation on the beach. The discovered patterns even suggest that burial, crushing and sediment compaction by the bulldozers were the most probable causative factors for these observed ecological impacts. The finishing work done by bulldozers is not always necessary as the action of the waves and tides restores the natural appearance of the beach in a relatively short period of time (Adriaanse & Coosen 1991). The most benign strategy is slow nourishment by sheeting a spray of sand and water (rainbow spraying). This allows beach organisms to keep up with the sediment overburdens as they are applied (Grober 1992; Schlacher et al. 2012).

4. Management tools for Belgian coastal policy

(Speybroeck et al. 2006a) indicated that an ecosystem vision on nourishment effects is generally missing. Hence, extensive scientific information on the complete beach ecosystem and clear and easy to use management and decision support tools are provided.

Predictive model for the effects of beach nourishment

The nourishment simulation model for the Belgian beach ecosystem, developed in chapter 5, integrates species envelope-based projections for the dominant macrobenthos species and mechanistic foodweb modules for higher trophic levels, e.g. epibenthos and birds. It enables the user to compare the effects of nourishment with varying technical features. According to the model, the sediment grain size is the most important factor determining beach-level diversity and production, with strong deterioration of the beach ecosystem after nourishment with too coarse sediment (e.g. >> than 300 µm). Therefore the gradient in sediment grain sizes that is advised for nourishment of fine-grained beaches is defined as 200

– 300 µm with the critical median sediment grain size set at 300 µm. Although the effect of nourishment slope was less strong compared to the sediment, nourishment slope did also affect species zonation patterns. For a uniform sediment grain size, high-shore nourishment was found to positively influence the abundances of high-shore species such as *Bathyporeia pilosa*. Patterns for higher trophic levels do not follow these decreasing patterns in macrobenthos abundance and biomass. Both the slope of the nourishment project as well as the sediment can be varied in the model, enabling the user to determine the combination with the lowest impact on the ecosystem. This first predicting model for nourishment effects can as such be a valuable tool in the selection process for compensation options.

Baseline maps depicting the ecological value of our beaches

In chapter 6, a scientifically sound and spatially based biological valuation of the Belgian coastal zone is given, using the marine biological valuation method (Derous et al. 2007a). Spatial coverage and overall data availability were satisfactory and allowed for significant trends and patterns to be observed. Although the Belgian coast is entirely composed by sandy beaches, there is indeed biological diversity among distinct subzones. A strong mosaic pattern of biological value along the coastline and a clear lack of (benthic) data at the eastern part of the Belgian coast was detected. Around 70 % of the shallow part of the subzones scored rather *high* biological values, compared with the intertidal part and *high/very high* biological values were consistently found in intertidal zones located immediately to the east of the harbours Nieuwpoort, Oostende and Zeebrugge. A detailed analysis of protected areas and areas under coastal flood risk indicated that the use of Biological Valuation Maps (BVMs) is very promising in order to differentiate between several impact values. BVMs provided a strong visual support to the proposal for the extension of some already existing nature reserves and to the need for more data to allow for significant conclusions regarding the biological value of other reserves. The designation of marine reserves adjacent to protected beaches is of the uttermost importance to achieve a successful and ecologically justified implementation of beach reserves (Herrier 2002). BVMs will allow for the integration of 'natural/ecological values' at an early stage of policy implementation, spatial planning and nature conservation.

Beach fact sheets of all studied Belgian beaches, combining all research information

The beach fact sheets in Appendices – Chapter 7 – Beach Fact sheets provide all information gathered during this PhD research (chapter 2 – 6) on the 16 intertidal and 10 shallow subtidal studied coastal locations. In a clear and easy to work with format, each beach record gives an overview of its location, legal circumstances (Provincial Spatial Implementation Plan and nature conservation status), possible coastal defence activities, biological value and current scientific knowledge.

5. Conclusions

5.1 Beach Spatial Planning = science + policy

The Belgian coastal zone should be evaluated against all beach functions, including coastal environmental protection, coastal defence and tourism to obtain an integrated beach spatial plan. In some coastal areas a conflict is present between locations sensible to coastal flood and locations displaying *high*/*very high* biological value, e.g. Middelkerke, Oostende-East and Knokke-Heist (chapter 6). If coastal defence activities are to be performed in these areas, appropriate (mitigation) measures should be drafted.

Beside the delineation of Habitat 2000 areas, European legislation also forces the member states to define a good ecological status for these areas and to formulate conservation objectives. As a high human impact has been influencing the beach ecosystems in the past, it is difficult to determine the best possible quality of a Belgian beach. The current most valuable beaches are not necessarily the best possible beaches as we do not know their (possibly better) condition in the past. On the other hand, beaches that now show a lower intrinsic value could have the potential for valuable nature development. Hence, the definition of a good ecological beach (a healthy beach of habitat type 1140) has to be formulated in a human impacted time and space, making it a very hard exercise. Nevertheless, such a definition is essential for formulating conservation objectives. The criteria for a good ecological beach have to indicate what ecosystem components and processes in ecosystem functioning need to be available on a healthy beach and what state they have to be in. Figure 3 gives a preliminary overview of criteria based on my own research. Future research can elaborate and specify these criteria.

There is still a pressing need for better communication and cooperation between scientific institutions involved. Currently, the Research Institute for Nature and Forest (INBO) is responsible for suitable assessments and beach protection but not all necessary information is available to this institution. Therefore, a good communication and regular deliberation between Belgian beach ecology experts and INBO is essential for the best possible assessments and conservation objectives. Regarding coastal defence activities, all research conducted during this PhD research amounted to updated guidelines for ecological adjustment of beach nourishment as summarized in figure 4. Furthermore, relevant marine scientific institutes should be more visible in determining important guidelines for spatial planning. Although uncertainty is inherent to the scientific process, research institutions should dare to make statements and predictions, necessary for an integrated spatial beach planning.

A multi-disciplinary, integrated beach spatial plan, combined with the marine spatial plan of the shallow subtidal Belgian zone should be the ideal scenario for the Belgian coast. Integrated consultation and deliberation with all stakeholders, institutes and authorities involved will become an important issue in the future. However, this will always be a tremendous task as local authorities will not be keen on ceding power to higher authorities. Nevertheless, narrowing local authority power and making decisions on higher jurisdiction levels will be the only solution for establishing a long-term, integrated and sustainable beach and coastal spatial plan for the entire Belgian coast.

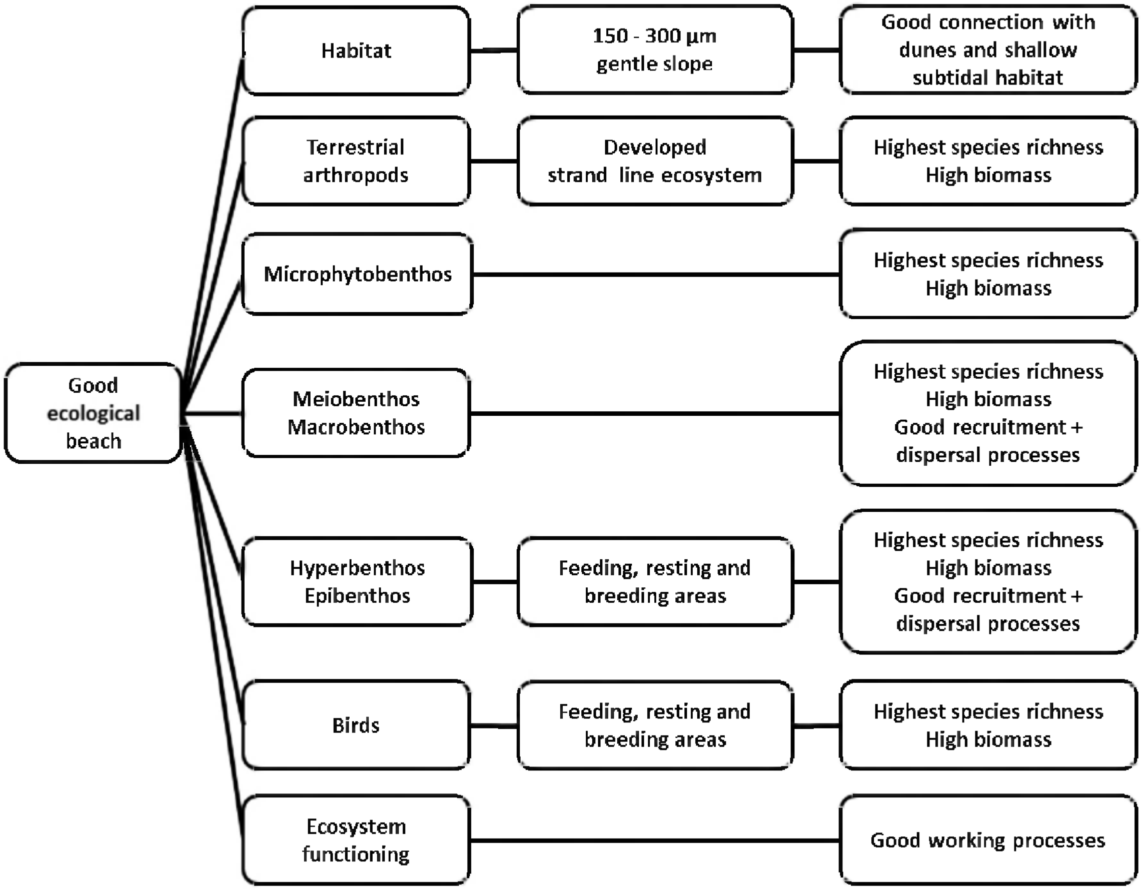


Figure 3: Criteria for a good ecological beach

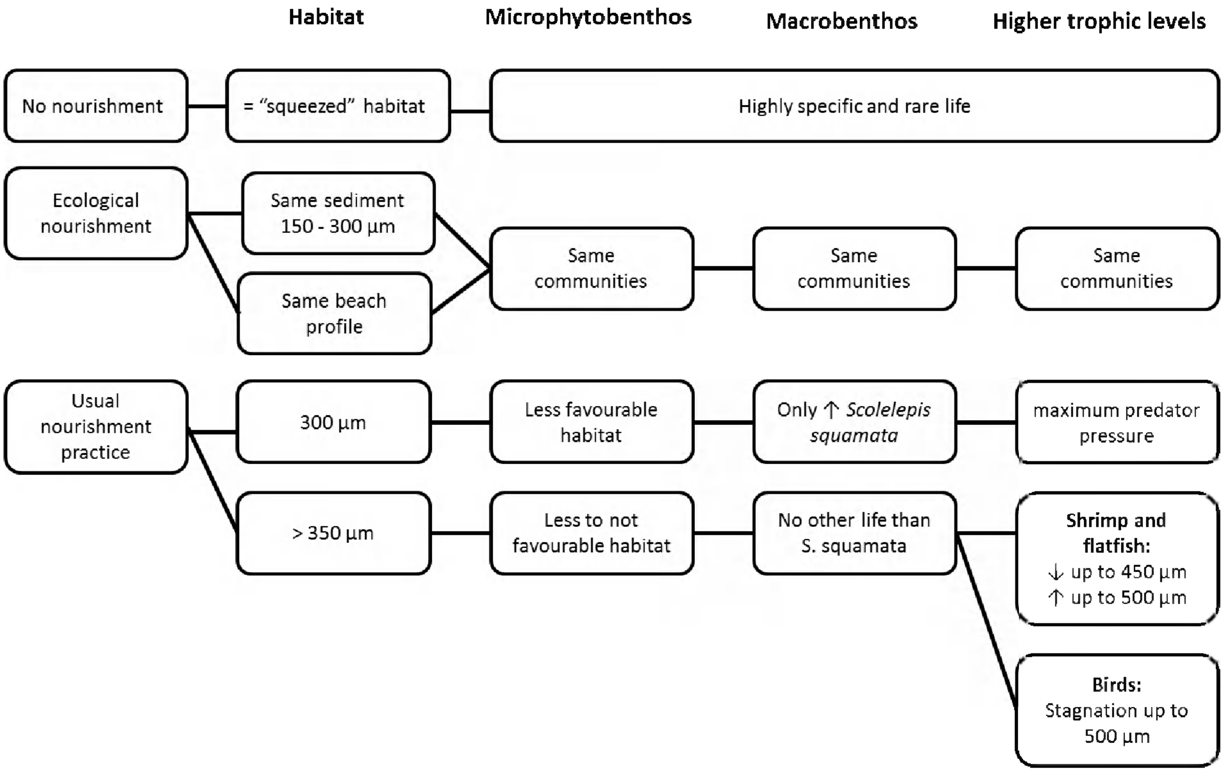


Figure 4: Guidelines for coastal defence, in particular ecological nourishment

Sacrificing bits of nature (urbanised beaches) in order to protect the Belgian beach ecosystem (benthic protected areas)

As highly touristic, (semi-)urban, top priority coastal defence beaches with a high percentage of development along the coast are heavily threatened by coastal erosion and sea level rise and need the most protection, beach nourishment will be applied repeatedly on these beaches. Furthermore, to enlarge the 'lifetime' of the nourishment, both steep slopes and coarse sediments will be used, leading to negative impacts on the ecosystem of these beaches. However, these beach ecosystems are already strongly impacted and consequently impoverished by beach cleaning, trampling, pollution and presence of coarse material due to previous (local) nourishment projects. Moreover, these impacts suppress any possible development of healthy beach ecosystems. Therefore one could suggest to 'sacrifice' touristic and top priority coastal defence beaches in the light of nature protection in order to focus on the protection of ecologically more valuable beaches (determined in chapter 6). That way, both the intrinsic value of the beach ecosystem could be protected and human use of the beach can be kept on the biologically less valuable beaches.

When beach nourishment is executed under ecological optimal conditions, following the guidelines for coastal defence, compensation measures are minimal. The loss of touristic, top priority coastal defence and/or ecologically low valuable beaches should not be considered at the same level as the loss of ecologically valuable beaches, urging for compensation measures to be more attuned to the economic reality of the beach. Communication between institutions that monitor the impact of beach nourishment and institutions that have a more advising role is such crucial to formulate valuable guidelines for compensation or ecological nourishment in the suitable assessments of announced nourishment projects. Furthermore, a good knowledge on the response of the beach ecosystem following nourishment is essential (chapter 3 and 5) and a suitable environmental impact assessment for every beach nourishment remains needed, in accordance with European legislation (CBD, Convention on Biological Diversity, 1992; Precautionary principle(Commission 2011)).

A good connection between biologically valuable beaches and connections with both protected dune areas and protected (shallow) subtidal areas ascertains the protection of an overall valuable beach ecosystem. Concerning beach conservation, one of the first tasks will be to evaluate and designate the area of the proposed habitat type 1140. In the evaluation process, the connectivity with (shallow) subtidal protected areas and dune reserves is essential. As this condition is generally fulfilled for the proposed areas, the location of these areas is well-considered but the ecological value should still be evaluated with the recently available BVM of the Belgian beaches (chapter 6). This evaluation shows that the most important flaws of the proposed Habitat 2000 areas are the absence of protected areas in the central part of the Belgian coastal zone and the non-incorporation of ecological valuable beach areas located immediately to the east of the three prominent Belgian harbours. The major wind-driven and tidal currents and waves at the Belgian coast have a southwest-northwest direction (van der Molen & van Dijck 2000; Speybroeck et al. 2008a). The east side of these prominent hard structures (also referred to as lee-side) is a sheltered area where hydrodynamics are less intense and sand deposition occurs, creating a wealth in soft-bottom habitats and proper environmental conditions for benthic colonization. Furthermore, connectivity of protected beach zones should be well considered. Therefore, the central part of the Belgian coastal zone should be better covered with protected beaches, in connection with the protected dunes in this area.

Combination of coastal defence techniques

In some conditions, the best approach for coastal defence is a combination of management measures. The construction of groins may be beneficial in some situations, not only to protect the physical (nourished) beach, but also to provide refuge for certain shorebirds and other threatened species. However, these hard structures introduce a new, not naturally occurring habitat and ecosystem into and onto the Belgian beaches (Engledow et al. 2001). Such constructions need careful planning and execution and a thorough knowledge of sand transport and budgets in the area is mandatory. Moreover, beach nourishment, foreshore nourishment, the construction of technically highly enhanced dykes that do not disturb the view, dune creation, reshaping, brushwood hedges and the construction of groins can all be combined to obtain a sound beach safety plan that includes a minimum of costs and work while minimally impacting the beach ecosystem.

Beach nourishment can encourage further development along unstable shorelines which can further reduce future alternative management options, such as shoreline retreat (Greene 2002). The project 'Vlaamse Baaien' aims at developing a Masterplan for the Flemish Coastal Zone by 2050, regarding five basic concepts : safety, naturalness, attractiveness, sustainability and development. One of the proposed projects involves the creation of barrier islands in front of the Belgian coast. These islands can have different functions, including coastal protection, green energy support ('Socket at Sea' principle, a place to store energy at sea), maritime safety support and development of durable energy. However, functions on these islands will only move the actual coastline further seawards since these islands will then suffer the impact of erosion and climate change instead of the Belgian beaches. Consequently, coastal defence will be necessary on these islands. Research on barrier islands showed profound impacts of beach nourishment. A steeper beach profile is created when sand is stacked on the beach during the nourishment process and this condition can lead to greater wave energy on the beach and greater beachside erosion (Kaufman & Pilkey 1983). It can also preclude wave overwash, leading to further erosion on the soundside. Under normal conditions, barrier islands move slowly landward with rising sea level (Pilkey & Clayton 1989; Pilkey 1998). Some scientists have predicted that efforts to keep these dynamic areas in a fixed location through for instance beach nourishment, will ultimately result in their demise.

Beach Ecosystem vision

The Belgian beach ecosystem has been thoroughly described by Speybroeck (2008). The necessity for a good ecosystem approach and a solid ecosystem interpretation led to the development of a nourishment model (chapter 5). It was the first attempt to link different trophic levels to one another in a food web framework. Based on the baseline information on Belgian beaches (chapter 2), the impact effects of both an ecological nourishment (chapter 3) and the effect of coarser median grain size on the intertidal macrobenthos (chapter 4), a better beach ecosystem vision was incorporated in the model in chapter 5 and as a result, the nourishment impact on the distribution and zonation of microphytobenthos, macrobenthos, fish and birds was modeled. The presence of a lot of birds or a high macrobenthos biomass on the beach can be a deceiving indicator for the beach ecosystem health. Indeed, the model shows that after nourishment with coarse sediment (> 300 µm), both total macrobenthos biomass as well as bird abundances increase. However, this is the result of the decrease in biodiversity and the increase of the abundance of one opportunistic macrobenthos species, *Scolelepis squamata*, resulting in the attraction of trophically linked bird species. The quality of the beach ecosystem and the importance of the biodiversity in the functioning of the beach ecosystem is as such not visible by only assessing for instance the biomass flow through trophic levels. Nutrient sediment cycles (nitrogen, carbon…) also play a significant role through primary production, microbial cycles and so on. Hence, the observation and evaluation of a too limited selection of ecosystem variables will hamper a good ecosystem approach. The combination of at least biodiversity and biomass provides for a better assessment of the beach ecosystem quality.

The intertidal Belgian beaches represent the largest nursery area for both marine fish as well as birds along the Belgian coast (Beyst et al. 1999a; Vanermen et al. 2009). Hence, degradation of the intertidal beach will heavily impact these higher trophic levels (Stienen & Van Waeyenberge 2004; Stienen et al. 2005). Nourishment impact on biologically valuable beaches with rich feeding grounds for birds and fish (chapter 6) will have an important effect on the populations of higher trophic species of the whole area. It is however impossible at the moment to exactly quantify the minimum impact area of valuable beaches that will have a meaningful effect on the higher trophic levels of the ecosystem.

The combination of one major beach nourishment project, followed by a foreshore nourishment for maintenance, will probably be a bad option for juvenile epibenthos and hyperbenthos feeding on the beach. The major beach nourishment will render the intertidal nursery area (temporarily) unavailable while the foreshore nourishment will impact the alternative nursery area, the shallow subtidal. Moreover, the shallow subtidal cannot be used as a refuge for marine intertidal species during the beach nourishment so this combination can have a negative local effect on these benthic communities. The preliminary results of the Ameland reports contest this statement but more research is needed. The combination of beach and foreshore nourishment needs careful consideration and should be evaluated on a site-specific basis.

5.2 Future beach research

If we continue monitoring the Belgian beaches in a standardized way, we will succeed in building a long term dataset at meta-analysis scale. For the understanding of large-scale patterns, intensive long-term sampling in a few areas would be meaningless, and a large number of snapshot samples covering a wide range of conditions is more appropriate (McLachlan & Dorvlo 2005). Regarding future monitoring, we suggest surveying every beach of interest in both the intertidal and shallow subtidal zone, preferably replicated in an appropriate manner. Seasonal variation can be monitored in spring and autumn although yearly monitoring in autumn will suffice as well. Prior to future research, pilot studies could be performed to determine the relative efficiency, accuracy and precision of: (1) combinations of sample size, depth of sampling and sieve mesh size, and of (2) macrobenthos sampling stratification in order to assess feasibility of a stratified random sampling design and hence a possible reduction in the required number of samples. It might also be interesting to gain insight in the hydrodynamic and turbidity conditions of the sandy beach ecosystem.

For examining most environmental impacts and many other ecological hypotheses, the temporal scales of change are not known and can seldom be predicted. *In situ* monitoring in the field also goes hand in hand with environmental heterogeneity, unpredictable biotic and abiotic environmental fluctuations and sampling variances, making the detection of impact effects difficult and arduous. In spite of all this, monitoring still remains the best way forward as other techniques have their constraints as well, e.g. experiments representing an artificial environment or models with possible incomplete algorithms predicting unrealistic patterns and trends. Ideally, research in the field should be combined with experimental research to study the tolerance and preference of species for certain beach dependent factors, like beach slope, turbidity and silt fraction, and to study migration and recruitment patterns.

Much research remains to be done on ecological relationships between macrobenthos and other trophic beach levels (meiobenthos, hyperbenthos, epibenthos, microphytobenthos and birds). Gaining a comprehensive understanding of how these communities fit into the larger beach ecosystem and food web will be necessary to fully assess the impact of anthropogenic activities. Cumulative ecological effects of beach nourishment in both space and time remain hardly unknown (Greene 2002; Speybroeck et al. 2006a) and research on foreshore nourishment as a prime and cumulative defence technique is of the utmost importance in the near future. All this information could lead to a better understanding of the sandy beach ecosystem and its resilience to withstand impacts, not in the least the impact of nourishment.

Cited literature

Cited literature

Adin R, Riera P (2003) Preferential food source utilization among stranded macroalgae by Talitrus saltator (Amphipod, Talitridae): a stable isotopes study in the northern coast of Brittany (France). Estuarine, Coastal and Shelf Science 56:91-98

Adriaanse LA, Coosen J (1991) Beach and dune nourishment and environmental aspects. Coastal Engineering 16:129-146

Airoldi L, Abbiati M, Beck MW, Hawkins SJ, Jonsson PR, Martin D, Moschella PS, Sundelöf A, Thompson RC, Aberg P (2005) An ecological perspective on the deployment and design of low-crested and other hard coastal defence structures. Coastal Engineering 52:1073-1087

Alheit J, Naylor E (1976) Behavioural basis of intertidal zonation in *Eurydice pulchra* Leach. J Exp Mar Biol Ecol 23:135-144

Anfuso G, Benavente J, Gracia FJ (2001) Morphodynamic responses of nourished beaches in SW Spain. Journal of Coastal Conservation 7:71-80

Araújo MB, Peterson AT (2012) Uses and misuses of bioclimatic envelope modeling. Ecology 93:1527-1539

Arkema KK, Abramson SC, Dewsbury BM (2006) Marine ecosystem-based management: from characterization to implementation. Frontiers in Ecology and the Environment 4:525-532

Baeye M, Fettweis M, Voulgaris G, Van Lancker V (2010) Sediment mobility in response to tidal and wind-driven flows along the Belgian inner shelf, southern North Sea. Ocean dynamics 61:611-622

Bahn V, McGill BJ (2012) Testing the predictive performance of distribution models. Oikos 122:321-331

Balvanera P, Pfisterer AB, Buchmann N, He J-S, Nakashizuka T, Raffaelli D, Schmid B (2006) Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecology Letters 9:1146-1156

Basco DR (1999) Overview of Beach Engineering in the United States of America p116

Beaumont NJ, Austen MC, Atkins JP, Burdon D, Degraer S, Dentinho TP, Derous S, Holm P, Horton T, van Ierland E, Marboe AH, Starkey DJ, Townsend M, Zarzycki T (2007) Identification, definition and quantification of goods and services provided by marine biodiversity: Implications for the ecosystem approach. Marine Pollution Bulletin 54:253-265

Beaumont NJ, Austen MC, Mangi SC, Townsend M (2008) Economic valuation for the conservation of marine biodiversity. Marine Pollution Bulletin 56:386-396

Beck MW, Heck KL, Able KW, Childers DL, Eggleston DB, Gillanders BM, Halpern BS, Hays CG, Hoshino K, Minello TJ (2003) The role of nearshore ecosystems as fish and shellfish nurseries. Issues in Ecology 11

Becuwe M, Lingier P, Deman R, De Putter G, De Vos K, Rappé G, Sys P (2006) Ecologische atlas van de Paarse Strandloper en de Steenloper aan de Vlaamse kust 1947-2005: Aantalsevolutie, terreingebruik, activiteitsritme en gedrag van de Paarse Strandloper (Calidris maritima) en de Steenloper (Arenaria interpres) aan de Vlaamse kust, 1947-2005= Numbers, habitat use, activity system and behaviour of the Purple Sandpiper (Calidris maritima) and the Turnstone (Arenaria interpres) on the Flemish coast, Belgium, 1947-2005. VLIZ Special Publication 33:183

Belpaeme K, Konings P (2004) The Coastal Atlas Flanders: Belgium, Vol. Publication of the Coordination Center for Coastal Zone Management, Belgium.

Beukema JJ (1992) Dynamics of juvenile shrimp *Crangon crangon* in a tidal-flat nursery of the Wadden Sea after mild and cold winters. Mar Ecol Prog Ser 83:157-165

Beukema JJ (1997) Caloric values of marine invertebrates with an emphasis on the soft parts of marine bivalves. Oceanogr Mar Biol 35:387-414

Beyst B, Buysse D, Dewicke A, Mees J (2001a) Surf Zone Hyperbenthos of Belgian Sandy Beaches: Seasonal Patterns. Estuarine, Coastal and Shelf Science 53:877-895

Beyst B, Cattrijsse A, Mees J (1999a) Feeding ecology of juvenile flatfishes of the surf zone of a sandy beach. Journal of Fish Biology 55:1171-1186

Beyst B, Hostens K, Mees J (2001b) Factors influencing fish and macrocrustacean communities in the surf zone of sandy beaches in Belgium: temporal variation. Journal of Sea Research 46:281-294

Beyst B, Mees J, Cattrijsse A (1999b) Early postlarval fish in the hyperbenthos of the Dutch Delta (south-west Netherlands). VLIZ Collected Reprints: Marine and Coastal Research in Flanders 29

Beyst B, Vanaverbeke J, Vincx M, Mees J (2002) Tidal and diurnal periodicity in macrocrustaceans and demersal fish of an exposed sandy beach, with special emphasis on juvenile plaice Pleuronectes platessa. Mar Ecol Prog Ser 225:263-274

Bindoff NL, Willebrand J, Artale V, Cazenave A, Gregory J, Gulev S, Hanawa K, Le Quéré C, Levitus S, Nojiri Y, Shum CK, Talley LD, Unnikrishnan A (2007) Observations: Oceanic Climate Change and Sea Level. In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA

Bird ECF (2000) Coastal Geomorphology: An Introduction., Vol. John Wiley, Chichester

BirdLife I (2013) Important Bird Areas factsheet: Zwin. Downloaded from http://www.birdlife.org on 16/01/2013

Bogaert DCM, Maes F (2008) Who rules the coast?: policy processes in Belgian MPAs and beach spatial planning, Vol. Maklu

Bonte D, Baert L, Maelfait JP (2002) Spider assemblage structure and stability in a heterogeneous coastal dune system (Belgium). The Journal of Arachnology 30:331-343

Bonte D, Criel P, Vanhoutte L, Van Thournout I, Maelfait JP (2004) The importance of habitat productivity, stability and heterogeneity for spider species richness in coastal grey dunes along the North Sea and its implications for conservation. Biodiversity and Conservation 13:2119-2134

Bonte D, Hoffmann M (2005) Are coastal dune management actions for biodiversity restoration and conservation underpinned by internationally published scientific research? VLIZ Special Publication 19

Borja Á, Elliott M, Carstensen J, Heiskanen A-S, van de Bund W (2010) Marine management: Towards an integrated implementation of the European Marine Strategy Framework and the Water Framework Directives. Marine Pollution Bulletin 60:2175-2186

Borja A, Galparsoro I, Irigoien X, Iriondo A, Menchaca I, Muxika I, Pascual M, Quincoces I, Revilla M, Germán Rodríguez J, Santurtún M, Solaun O, Uriarte A, Valencia V, Zorita I (2011) Implementation of the European Marine Strategy Framework Directive: A methodological approach for the assessment of environmental status, from the Basque Country (Bay of Biscay). Marine Pollution Bulletin 62:889–904

Boulez S (2002) Het macrobenthos van zwinnen op Vlaamse zandstranden: zonatie en tidale variatie. Ghent University

Brampton A (1992) Beaches-the natural way to coastal defence. Coastal zone planning and management, ICE:221-229

Brazeiro A (2001) Relationship between species richness and morphodynamics in sandy beaches: what are the underlying factors? Marine Ecology Progress Series 224:35-44

Brooks SP, Gelman A (1998) General methods for monitoring convergence of iterative simulations. Journal of computational and graphical statistics 7:434-455

Brown AC, McLachlan A (2002) Sandy shore ecosystems and the threats facing them: some predictions for the year 2025. Environmental Conservation 29:62-77

Brown JH, Maurer BA (1986) Body size, ecological dominance and Cope's rule. Nature 324:248-250

Buchanan JB, Sheader M, Kingston PF (1978) Sources of Variability in the Benthic Macrofauna off the South Northumberland Coast, 1971-1976. Journal of the Marine Biological Association of the United Kingdom 58:191-209

Burke L, Kura Y, Kassem K, Revenga C, Spalding M, McAllister D, Caddy J (2001) Coastal ecosystems, Vol. World Resources Institute Washington, DC

Buyle G (2002) Het hyper-en epibenthos van zwinnen op het strand van Koksijde: zonale en tidale variatie. Ghent University

Byrnes MR, Hammer RM, Thibaut TD, Snyder DB (2004) Physical and biological effects of sand mining offshore Alabama, USA. J Coast Res 20:6-24

Carl J, Sparrevohn CR, Nicolajsen H, Støttrup J (2008) Substratum selection by juvenile flounder Platichthys flesus (L.): effect of ephemeral filamentous macroalgae. J Fish Biol 72:2570-2578

Castro G, Myers J, Place AR (1989) Assimilation efficiency of sanderlings (*Calidris alba*) feeding on horseshoe crab (*Limulus polyphemus*) eggs. Physiological Zoology 62:716-731

Cattrijsse A, Vincx M (2001) Biodiversity of the benthos and the avifauna of the Belgian coastal waters: summary of data collected between 1970 and 1998.

Charlier RH (2011) The Zwin: From Golden Inlet to Nature Reserve. Journal of coastal Research 27:746-756

Charlier RH, Chaineux MCP, Morcos S (2005) Panorama of the history of coastal protection. Journal of Coastal Research:79-111

Charlier RH, Decroo D, De Meyer CP, B. L (1998) To feed or not to feed, that is often the question. International Journal of Environmental Studies 55:1-23

Cisneros KO, Smit AJ, Laudien J, Schoeman DS (2011) Complex, dynamic combination of physical, chemical and nutritional variables controls spatio-temporal variation of sandy beach community structure. PloS one 6:e23724

Clarke KR, Somerfield PJ, Gorley RN (2008) Testing of null hypotheses in exploratory community analyses: similarity profiles and biota-environment linkage. Journal of Experimental Marine Biology and Ecology 366:56-69

Cliquet A (2001) Coastal zone management in Belgium. Revue juridique de l'environnement:85-106

Cliquet A, Bogaert D, De Waen D, Maes F (2007) The designation of marine protected areas in Belgium: From government to governance?

Cliquet A, Decleer K (2007) Natura 2000 in de Belgische Noordzee: meer dan een'papieren'bescherming. De Levende Natuur-édition 108

Cliquet A, Kervarec F, Bogaert D, Maes F, Queffelec B (2010) Legitimacy issues in public participation in coastal decision making processes: Case studies from Belgium and France. Ocean & Coastal Management 53:760-768

Cliquet A, Maes F (2001) Beleidsondersteunend onderzoek voor een geïntegreerd kustzonebeleid in België: welk geïntegreerd kustzonebeleid? In: in: Mees JeaE, 4: pp. 12-17 (eds) Beheer van kust en zee: beleidsondersteunend onderzoek in Vlaanderen: Studiedag, Oostende, 9 november 2001. VLIZ Special Publication, 4

Cliquet A, Van Poucke L, Queffelec B, O'Hagan AM, Malfait E, Maes F (2011) Adaptation in international and European legislation.

Coe WR (1956) Fluctuations in populations of littoral marine invertebrates. J mar Res 15:212-232

Collet S (2002) Appropriation of marine resources: from management to an ethical approach to fisheries governance. Social science information 41:531-553

Commission E (2011) Environmental Impact Assessment Directive, Brussels

Commission E (2013a) Commission takes initiative to support Blue Growth through sustainable management of marine and coastal areas (IP/13/222), 13 March 2013.

Commission E (2013b) Proposal for a Directive of the European Parliament and of the Council establishing a framework for maritime spatial planning and integrated coastal management p34

Connell JH (1983) On the prevalence and relative importance of interspecific competition: evidence from field experiments. Am Nat 122:661-696

Cooke BC, Jones AR, Goodwin ID, Bishop MJ (2012) Nourishment practices on Australian sandy beaches: A review. Journal of environmental management

Copejans E, Smits M (2011) De wetenschap van de zee: over een onbekende wereldoceaan, Vol. Uitgeverij Acco

Costanza R (1999) The ecological, economic, and social importance of the oceans. Ecological Economics 31:199-213

Costanza R, d'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill RV, Paruelo J, Raskin RG, Sutton P, van den Belt M (1998) The value of the world's ecosystem services and natural capital. Ecological Economics 25:3-15

Courtens W, Stienen EWM (2004) Voorstel tot afbakening van een vogelrichtlijngebied voor het duurzaam in stand houden van de broedpopulaties van kustbroedvogels te Zeebrugge-Heist. Advies van het Instituut voor Natuurbehoud

Crain CM, Halpern BS, Beck MW, Kappel CV (2009) Understanding and managing human threats to the coastal marine environment. Annals of the New York Academy of Sciences 1162:39-62

Creutzberg F, Wapenaar P, Duineveld G, Lopez Lopez N (1984) Distribution and density of the benthic fauna in the southern North Sea in relation to bottom characteristics and hydrographic conditions. Rapport et Proces-verbaux des Reunions, Conseil International pour l'Exploration de la Mer 183

Crowder L, Norse E (2008) Essential ecological insights for marine ecosystem-based management and marine spatial planning. Marine Policy 32:772-778

Crowder LB, Osherenko G, Young OR, Airamé S, Norse EA, Baron N, Day JC, Douvere F, Ehler CN, Halpern BS (2006) Resolving mismatches in US ocean governance. Science 313:617

Culter JK, Mahadevan S (1982) Long-term Effects of Beach Nourishment on the Benthic Fauna of Panama City Beach, Florida, DTIC Document

Cummings VJ, Pridmore RD, Thrush SF, Hewitt JE (1995) Post-settlement movement by intertidal benthic macroinvertebrates: Do common New Zealand species drift in the water column? . N Zeal J Mar Freshw Res 29:59-67

D'Hondt I (1999) Tidale, diurnale en semilunaire ritmes in het hyperbenthos van het strand van Lombardsijde. Ghent University

Dahl E (1952) Some aspects of the ecology and zonation of the fauna on sandy beaches. Oikos 4:1-27

Dahl F (1971) Die Tierwelt Deutschland und angrenzenden Meerseteile., Vol 58. Veb. Gustav Fischer Verlag, Jena

Dankers N, Binsbergen M, Zegers K (1983) De effecten van zandsuppletie op de fauna van het strand van Texel en Ameland, Rijksinstituut voor Natuurbeheer-Texel, p 12

Dauer DM (1983) Functional morphology and feeding behavior of *Scolelepis squamata* (Polychaeta: Spionidae). Mar Biol 77:279-285

Davies JL (1964) A morphogenic approach to world shorelines. Zeitschrift für Geomorphologie 8:127-142

Davis RAJ (1994) The Evolving Coast, Vol. Scientific American Library, New York

De Backer A (2001) De macrofauna ter hoogte van de vloedmerklijn op Vlaamse stranden. Ghent University

de Groot RS, Brander L, van der Ploeg S, Costanza R, Bernard F, Braat L, Christie M, Crossman N, Ghermandi A, Hein L, Hussain S, Kumar P, McVittie A, Portela R, Rodriguez LC, ten Brink P, van Beukering P (2012) Global estimates of the value of ecosystems and their services in monetary units. Ecosystem Services 1:50-61 de Groot RS, Wilson MA, Boumans RMJ (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecological Economics 41:393-408

De Meulenaer B (2006) Voedselecologie van strandvogels: Drieteenstrandloper (*Calidris alba*). University of Gent

De Moor G (1979) Recent beach evolution along the Belgian North Sea coast. Bulletin van de Belgische Vereniging voor Geologie 88:143-157

De Moor G (1986) Geomorfologisch onderzoek op het Belgisch kontinentaal plat. Tijdschrift van de Belgische Vereniging voor Aardrijkskundige Studies= Bulletin de la Société Belge d'Etudes Géographiques 55

De Moor G (2006) Het Vlaamse strand. Geomorfologie en Dynamiek, Vlaams Instituut voor de Zee, Oostende, p 154

De Moor G, Bloome E (1988) General nature of the coastline. Artificial structures and shorelines 10:115

De Neve L (1996) Het effect van antropogene invloeden op de macrobenthische gemeenschappen van de intergetijdenzone ter hoogte van Koksijde. Ghent University

De Ruyck AMC, Ampe C, Langohr R (2001) Management of the Belgian coast: Opinions and solutions. Journal of Coastal Conservation 7:129-144

De Smet WMA (1978) Zeehonden langs de Belgische kust en in de binnenwateren. De Wielewaal 44

De Wolf P (2002) Kusterosie en verzanding van Het Zwin. VLIZ Special Publication 10

De Wolf P, Fransaer D, van Sieleghem J, Houthuys R (1993) Morphological Trends of the Belgian Coast Shown by 10 Years of Remote-Sensing Based Surveying Coastlines of the Southern North Sea. ASCE, p 245-257

Decleer KE (2007) Europees beschermde natuur in Vlaanderen en het Belgisch deel van de Noordzee: habitattypen, dier-en plantensoorten. Report No. 9040302677, Mededelingen van het Instituut voor Natuur-en Bosonderzoek (INBO M. 2007.01), Brussel, p 584

Defeo O (1996) Recruitment variability in sandy beach macroinfauna: much to learn yet. Revista Chilena de Historia Natural 69:615-630

Defeo O, Brazeiro A, deAlava A, Riestra G (1997) Is sandy beach macrofauna only physically controlled? Role of substrate and competition in isopods. Estuar Coast Shelf Sci 45:453-462

Defeo O, Jaramillo E, Lyonnet A (1992) Community structure and intertidal zonation of the macroinfauna on the Atlantic coast of Uruguay. J Coast Res 8:830-839

Defeo O, McLachlan A (2005) Patterns, processes and regulatory mechanisms in sandy beach macrofauna: a multi-scale analysis. Marine Ecology-Progress Series 295:1-20

Defeo O, McLachlan A (2011) Coupling between macrofauna community structure and beach type: a deconstructive meta-analysis. Mar Ecol Prog Ser 433:29-41

Defeo O, McLachlan A, Schoeman DS, Schlacher TA, Dugan J, Jones A, Lastra M, Scapini F (2009) Threats to sandy beach ecosystems: A review. Estuarine Coastal and Shelf Science 81:1-12

Degraer S, Braeckman U, Haelters J, Hostens K, Jacques T, Kerckhof F, Merckx B, Rabaut M, Stienen E, Van Hoey G, Van Lancker V, Vincx M (2009) Studie betreffende het opstellen van een lijst met potentiële Habitatrichtlijn gebieden in het Belgische deel van de Noordzee. Eindrapport in opdracht van de Federale Overheidsdienst Volksgezondheid, Veiligheid van de Voedselketen en Leefmilieu, Directoraat-generaal Leefmilieu Brussel, België 93

Degraer S, Courtens W, Haelters J, Hostens K, Jacques T, Kerckhof F, Stienen E, Van Hoey G (2010) Bepalen van instandhoudingsdoelstellingen voor de beschermde soorten en habitats in het Belgische deel van de Noordzee, in het bijzonder in beschermde mariene gebieden. Eindrapport in opdracht van de Federale Overheidsdienst Volksgezondheid, Veiligheid van de Voedselketen en Leefmilieu, Directoraat-generaal Leefmilieu Brussel, België:132

Degraer S, Moerkerke G, Rabaut M, Van Hoey G, Du Four I, Vincx M, Henriet J-P, Van Lancker V (2008a) Very-high resolution side-scan sonar mapping of biogenic reefs of the tube-worm Lanice conchilega. Remote Sensing of Environment 112:3323-3328

Degraer S, Mouton I, De Neve L, Vincx M (1999a) Community structure and intertidal zonation of the macrobenthos on a macrotidal, ultra-dissipative sandy beach: summer-winter comparison. Estuaries 22:11

Degraer S, Van Lancker VRM, Moerkerke G, Van Hoey G, Vanstaen K, Vincx M, Henriet JP (2003a) Evaluation of the ecological value of the foreshore habitat-model and macrobenthic side-scan sonar interpretation: extension along the Belgian coastal zone: final report, Ministry of the Flemish Community, Environment and Infrastructure. Department Waterways and Marine Affairs Administration, Coastal Waterways

Degraer S, Verfaillie E, Willems W, Adriaens E, Vincx M, Van Lancker V (2008b) Habitat suitability modelling as a mapping tool for macrobenthic communities: An example from the Belgian part of the North Sea Continental Shelf Research 28:369-379

Degraer S, Vincx M, Meire P, Offringa H (1999b) The macrozoobenthos of an important wintering area of the common scoter (Melanitta nigra). Journal of the Marine Biological Association of the UK 79:243-251

Degraer S, Volckaert A, Vincx M (2003b) Macrobenthic zonation patterns along a morphodynamical continuum of macrotidal, low tide bar/rip and ultra-dissipative sandy beaches. Estuarine, Coastal and Shelf Science 56:10

Degraer S, Wittoeck J, Appeltans W, Cooreman K, Deprez T, Hillewaert H, Hostens K, Mees J, Vanden Berghe E, Vincx M (2006) The macrobenthos atlas of the Belgian part of the North Sea. Belgian Science Policy, Brussels:164

del Norte-Campos AGC, Temming A (1994) Daily activity, feeding and rations in gobies and brown shrimp in the northern Wadden Sea. Marine Ecology-Progress Series 115:41-41

Depuydt F (1972) De Belgische strand-en duinformaties in het kader van de geomorfologie der zuidoostelijke Noordzeekust. Verhandelingen van de Koninklijke Academie voor Wetenschappen, Letteren en Schone Kunsten van België Klasse der Wetenschappen 122

Deronde B, Houthuys R, Debruyn W, Fransaer D, Lancker VV, Henriet J (2006) Use of Airborne Hyperspectral Data and Laserscan Data to Study Beach Morphodynamics along the Belgian Coast. Journal of Coastal Research 22:1108-1117

Deronde B, Houthuys R, Henriet JP, van Lancker V (2008) Monitoring of the sediment dynamics along a sandy shoreline by means of airborne hyperspectral remote sensing and LIDAR: a case study in Belgium. Earth Surface Processes and Landforms 33:280-294

Deronde B, Houthuys R, Sterckx S, Debruyn W, Fransaer D (2004) Sand dynamics along the Belgian coast based on airborne hyperspectral data and lidar data. EARSeL eProceedings 3:26

Derous S, Agardy T, Hillewaert H, Hostens K, Jamieson G, Lieberknecht L, Mees J, Moulaert I, Olenin S, Paelinckx D, Rabaut M, Rachor E, Roff J, Stienen EWM, van der Wal JT, van Lancker V, Verfaillie E, Vincx M, Weslawski JM, Degraer S (2007a) A concept for biological valuation in the marine environment. Oceanologia 49:99-128

Derous S, Austen M, Claus S, Daan N, Dauvin J-C, Deneudt K, Depestele J, Desroy N, Heessen H, Hostens K, Marboe AH, Lescrauwaet A-K, Moreno MP, Moulaert I, Paelinckx D, Rabaut M, Rees H, Ressurreicao A, Roff J, Santosi PT, Tatarek A, Ter Hofstede R, Vincx M, Zarzycki T, Degraer S, Speybroeck J, Stienen EWM (2007b) Building on the concept of marine biological valuation with respect to translating it to a practical protocol: Viewpoints derived from a joint ENCORA-MARBEF initiative. Oceanologia 49:579-586

Derous S, Courtens W, Cuvelier D, Deckers P, Deneudt K, Hillewaert H, Hostens K, Mees J, Moulaert I, Stienen E (2007c) BWZee: A biological valuation map for the Belgian part of the North Sea: Final report, Research in the framework of the BELSPO programme “Global change, ecosystems and biodiversity” – SPSD II, BELSPO, Belgium, p 162

Derous S, Vincx M, Degraer S (2007d) Marine Biological Valuation as a decision-support tool for Marine Management. Universiteit Gent

Devos K (2008) Numbers and population trends of waders along the Belgian North Sea coast. The European Non-Estuarine Coastal Waterbird Survey International Wader Studies 18:5-11

Dewicke A, Beyst B, Mees J (1998) Fish larvae in the hyperbenthal of Belgian coastal waters. Biologisch Jaarboek (Dodonaea) 65:134–135

Dewicke A, Cattrijsse A, Mees J, Vincx M (2003) Spatial patterns of the hyperbenthos of subtidal sandbanks in the southern North Sea. Journal of Sea Research 49:27-45

Dexter DM (1979) Community structure and seasonal variation in intertidal Panamanian sandy beaches. Estuarine and Coastal Marine Science 9:543-558

DFO (2004) Identification of Ecologically and Biologically Significant Areas, DFO Canadian Science Advisory Secretariat.

Diaz RJ, Cutter Jr GR, Hobbs Iii CH (2004) Potential impacts of sand mining offshore of Maryland and Delaware: Part 2-Biological considerations. J Coast Res:61-69

Doody JP (2005) Shoreline management-conservation, management or restoration? VLIZ Special Publication 19

Doody P, Ferreira M, Lombardo S, Lucius I, Misdorp R, Niesing H, Salman A, Smallegange M (2004) Living with coastal erosion in Europe: sediment and space for sustainability: Results from the EUROSION study, Vol

Douvere F (2008) The importance of marine spatial planning in advancing ecosystem-based sea use management. Marine Policy 32:762-771

Douvere F, Ehler CN (2009) New perspectives on sea use management: Initial findings from European experience with marine spatial planning. Journal of Environmental Management 90:77-88

Dugan JE, Defeo O, Jaramillo E, Jones AR, Lastra M, Nel R, Peterson CH, Scapini F, Schlacher T, Schoeman DS (2010) Give beach ecosystems their day in the sun. Science 329:1146

Dugan JE, Jaramillo E, Hubbard DM, Contreras H, Duarte C (2004) Competitive interactions in macroinfaunal animals of exposed sandy beaches. Oecologia 139:630-640

Dugan JE, McLachlan A (1999) An assessment of longshore movement in Donax serra Röding (Bivalvia: Donacidae) on an exposed sandy beach. J Exp Mar Biol Ecol 234:24

Ehler C (2008) Conclusions: Benefits, lessons learned, and future challenges of marine spatial planning. Marine Policy 32:840-843

Ehler C, Douvere F (2009) Marine Spatial Planning, A Step-by-Step Approach toward Ecosystem-based Management. IOC Manuals and Guides 53

Eleftheriou A, McIntyre A (1976) The Intertidal Fauna of Sandy Beaches. A Survey of the Scottish Coast, National Technical Information Service, Springfield VA 22161, p 65

Eleftheriou A, McIntyre A (2008) Methods for the study of marine benthos, Vol. Wiley-Blackwell

Elliott B, Degraer S, Bursey M, M. V (1997) Intertidal zonation of macroinfauna on a dissipative, sandy beach at De Panne (Belgium): a pilot study. Biologisch Jaarboek Dodonaea 64:17

Engledow H, Spanoghe G, Volckaert A, Coppejans E, Degraer S, Vincx M, Hoffmann M (2001) Onderzoek naar de fysische karakterisatie en de biodiversiteit van strandhoofden en andere harde constructies langs de Belgische kust, Instituut voor Natuurbehoud, Universiteit Gent, Brussel, Gent, p 110

Essink K (1999) Ecological effects of dumping of dredged sediments; options for management. Journal of Coastal Conservation 5:22

Evans D (2012) The EUNIS habitats classification-past, present & future. Revista de Investigación Marina 19:28-29

Feagin RA, Sherman DJ, Grant WE (2005) Coastal erosion, global sea-level rise, and the loss of sand dune plant habitats. Frontiers in Ecology and the Environment 3:359-364

Fernandes RSR, Soares-Gomes A (2006) Community structure of macrobenthos in two tropical sandy beaches with different morphodynamic features, Rio de Janeiro, Brazil. Marine Ecology-an Evolutionary Perspective 27:160-169

Fettweis M, Baeye M, Francken F, Lauwaert B, Van den Eynde D, Van Lancker V, Martens C, Michielsen T (2011) Monitoring the effects of disposal of fine sediments from maintenance dredging on suspended particulate matter concentration in the Belgian nearshore area(southern North Sea). Marine Pollution Bulletin 62:258-269

Fettweis M, Francken F, Van den Eynde D, Verwaest T, Janssens J, Van Lancker V (2010) Storm influence on SPM concentrations in a coastal turbidity maximum area with high anthropogenic impact(southern North Sea). Continental Shelf Research 30:1417-1427

Finkl CW (2002) Long-Term Analysis of Trends in Shore Protection Based on Papers Appearing in the Journal of Coastal Research, 1984-2000. Journal of coastal Research 18:24

Finkl CW, Walker HJ (2002) Beach nourishment. Engineered coasts:1-23

Fleming DM, Jones PJS (2012) Challenges to achieving greater and fairer stakeholder involvement in marine spatial planning as illustrated by the Lyme Bay scallop dredging closure. Marine Policy 36:370-377

Foley MM, Halpern BS, Micheli F, Armsby MH, Caldwell MR, Crain CM, Prahler E, Rohr N, Sivas D, Beck MW, Carr MH, Crowder LB, Emmett Duffy J, Hacker SD, McLeod KL, Palumbi SR, Peterson CH, Regan HM, Ruckelshaus MH, Sandifer PA, Steneck RS (2010) Guiding ecological principles for marine spatial planning. Marine Policy 34:955-966

Forero Parra CE (2007) A biological valuation map for the Dutch Continental Shelf.

Fremout A (2002) Overzicht van de tijwaarnemingen langs de Belgische Kust. Periode 1991-2000 voor Nieuwpoort, Oostende en Zeebrugge, Ministerie van de Vlaamse Gemeenschap - Department Leefmilieu en Infrastructuur - Administratie Waterwegen en Zeewezen - Afdeling Waterwegen Kust - Hydrografie, Oostende, p 38

Gheskiere T, Hoste E, Kotwicki L, Degraer S, Vanaverbeke J, Vincx M (2002) The sandy beach meiofauna and free-living nematodes from De Panne (Belgium). Bulletin van het KBIN, Biologie 72-suppl.:7

Gheskiere T, Hoste E, Vanaverbeke J, Vincx M, Degraer S (2004) Horizontal zonation patterns and feeding structure of marine nematode assemblages on a macrotidal, ultra-dissipative sandy beach (De Panne, Belgium). Journal of Sea Research 52:211-226

Gheskiere T, Vincx M, Urban-Malinga B, Rossano C, Scapini F, Degraer S (2005) Nematodes from wave-dominated sandy beaches: diversity, zonation patterns and testing of the isocommunities concept. Estuarine, Coastal and Shelf Science 62:365-375

Ghilarov AM (2000) Ecosystem functioning and intrinsic value of biodiversity. Oikos 90:408-412

Gibson R, Robb L (2000) Sediment selection in juvenile plaice and its behavioural basis. J Fish Biol 56:1258-1275

Gibson RN, Robb L (1996) Piscine predation on juvenile fishes on a Scottish sandy beach. Journal of fish biology 49:120-138

Gibson RN, Yoshiyama RM (1999) Intertidal fish communities. in: Horn, MH et al (Ed) (1999) Intertidal fishes: life in two worlds:264-296

Gilliland PM, Laffoley D (2008) Key elements and steps in the process of developing ecosystem-based marine spatial planning. Marine Policy 32:787-796

Gmelig Meyling AW, De Bruyne AW (1994) Zicht op zee: waarnemingen van veranderingen in de nabije kustzone door strandmonitoring met 381 strandwachten, p 79

Godet L, Toupoint N, Olivier F, Fournier J, Retière C (2008) Considering the functional value of common marine species as a conservation stake: the case of sandmason worm Lanice conchilega (Pallas 1766)(Annelida, Polychaeta) beds. Ambio: A Journal of the Human Environment 37:347-355

Goldberg WM (1988) Biological effects of beach restoration in South Florida: the good, the bad, and the ugly. Proc 1988 Fla Shore and Beach Preserv Assn Gainsville, Florida

Govaere JCR, Van Damme D, Heip C, De Coninck LAP (1980) Benthic communities in the Southern Bight of the North Sea and their use in ecological monitoring. Helgoländer Meeresuntersuchungen 33:507-521

Granek EF, Polasky S, Kappel CV, Reed DJ, Stoms DM, Koch EW, Kennedy CJ, Cramer LA, Hacker SD, Barbier EB, Aswani S, Ruckelshaus M, Perillo GME, Silliman BR, Muthiga N, Bael D, Wolanski E (2009) Ecosystem Services as a Common Language for Coastal Ecosystem-Based Management. Blackwell Publishing Inc, p 207-216

Granéli E, Turner JT (2002) Top-down regulation in ctenophore-copepod-ciliate-diatom-phytoflagellate communities in coastal waters: A mesocosm study. Marine Ecology Progress Series 239

Gray JS, Elliott M (2009) Ecology of marine sediments: from science to management, Vol. OUP Oxford

Green RH (1993) Application of repeated measures designs in environmental impact and monitoring studies. Australian Journal of Ecology 18:81-98

Greene K (2002) Beach Nourishment: A Review of the Biological and Physical Impacts ASMFC Habitat Management Series # 7 Washington DC:69

Griffiths CL, Stenton-Dozey JME, Koop K (1983) Kelp wrack and the flow of energy through a sandy beach ecosystem. Developments in Hydrobiology 19

Grimm V, Berger U, DeAngelis DL, Polhill JG, Giske J, Railsback SF (2010) The ODD protocol: A review and first update. Ecological modelling 221:2760-2768

Grober LE (1992) The ecological effects of beach replenishment Duke University

Grootaert P, Pollet M (2004) Vliegen als onontbeerlijke elementen in de ecologie van strand en duinen. Mededelingen van het Instituut voor Natuurbehoud 22:208-219

Guisan A, Zimmermann NE (2000) Predictive habitat distribution models in ecology. Ecological Modelling 135:147-186

Günther CP (1992) Dispersal of intertidal invertebrates: a strategy to react to disturbances of different scales? . Netherlands Journal of Sea Research 30:12

Hackney CT (1996) Review of Synthesis of Data on Surf Zone Fishes and Invertebrates in the South Atlantic Bight and the Potential Impacts from Beach Renourishment, Vol. University of North Carolina

Haelters J (1999) Zeehonden aan de Belgische kust in de zomer van 1998. De Strandvlo: Driemaandelijks Tijdschrift van De Strandwerkgroep België 19

Haghebaert G (1989) Coleoptera from marine habitats. In: K Wouters and L Baert (eds) Proceedings of the Symposium "Invertebrates of Belgium". KBIN, Brussel.301-308

Hamm L, Capobianco M, Dette HH, Lechuga A, Spanhoff R, Stive MJF (2002) A summary of European experience with shore nourishment. Coastal Engineering 47:28

Hanson H, Brampton A, Capobianco M, Dette HH, Hamm L, Laustrup C, Lechuga A, Spanhoff R (2002) Beach nourishment projects, practices, and objectives - a European overview. Coastal Engineering 47:31

Harley CDG, Randall Hughes A, Hultgren KM, Miner BG, Sorte CJB, Thornber CS, Rodriguez LF, Tomanek L, Williams SL (2006) The impacts of climate change in coastal marine systems. Ecology Letters 9:228-241

Harte M, Huntjes PMJM, Mulder S, Raadschelders EW (2002) Zandsuppleties en Europese richtlijnen, Rijksinstituut voor Kust en Zee/RIKZ. Ministerie van Verkeer en Waterstaat

Hartmann-Schröder G (1996) Annelida, Borstenwürmer, Polychaeta, Vol 58. Gustav Fischer Verlag, Jena

Harvey M, Gauthier D, Munro J (1998) Temporal changes in the composition and abundance of the macro-benthic invertebrate communities at dredged material disposal sites in the anse a Beaufils, baie des Chaleurs, eastern Canada. Marine pollution bulletin 36:41-55

Haynes D, Quinn GP (1995) Temporal and spatial variability in community structure of a sandy intertidal beach, Cape Paterson, Victoria, Australia. Marine and Freshwater Research 46:931-942

Hayward PJ (1994) Animals of sandy shores, Vol 21. The Richmond Publishing Co. Ltd

Heink U, Kowarik I (2010) What are indicators? On the definition of indicators in ecology and environmental planning. Ecological Indicators 10:584-593

Herrier JL (2002) De Baai van Heist: van verloren hoek tot eerste Vlaamse strandreservaat. VLIZ Special Publication 9

Herrier JL, Killemaes I (2001) Synopsis of the Flemish coastal dune conservation policy. In: Houston JAE, S.E.; Rooney, P.J. (Ed.) (2001) (eds) Coastal dune management: shared experience of European conservation practice: Proceedings of the European Symposium Coastal Dunes of the Atlantic Biogeographical Region Southport, northwest England, p 316-325

Herrier JL, Killemaes I, Noels C (2005) Purchase of dunes: the first step towards nature restoration along the Flemish coast. VLIZ Special Publication 19

Herrier JL, Leten M (2010) Restoration of the tidal lagoon of the Zwin Proceedings of the 7th European Conference on Ecological Restoration Avignon, France, p 23-27

Herrier JL, Van Nieuwenhuyse H (2005) The Flemish coast: life is beautiful! VLIZ Special Publication 19

Hiddink JG, Marijnissen SAE, Troost K, Wolff WJ (2002) Predation on 0-group and older year classes of the bivalve *Macoma balthica*: interaction of size selection and intertidal distribution of epibenthic predators. J Exp Mar Biol Ecol 269:223-248

Hill AE (1991) A mechanism for horizontal zooplankton transport by vertical migration in tidal currents. Marine Biology 111:485-492

Hillen R, Roelse P (1995) Dynamic preservation of the coastline in the Netherlands. Journal of Coastal Conservation 1:17-28

Hostens K, Mees J (1999) The mysid-feeding guild of demersal fishes in the brackish zone of the Westerschelde estuary. Journal of Fish Biology 55:704-719

Houghton JT, Ding Y, Griggs DJ, Noguer M, van der LINDEN PJ, Dai X, Maskell K, Johnson C (2001) Climate change 2001: the scientific basis, Vol 881. Cambridge University Press, Cambridge

Houston J (2003) The conservation of sand dunes in the Atlantic Biogeographical Region: the contribution of the LIFE programme. In: Publication:29–44 VS (ed) International Conference on Nature Restoration Practices in European Coastal Habitats. in Herrier J.-L., J. Mees, A. Salman, J. Seys, H. Van Nieuwenhuyse and I. Dobbelaere (Eds) (2005) Proceedings “Dunes and Estuaries 2005, Koksijde, Belgium

Incera M, Lastra M, López J (2006) Effect of swash climate and food availability on sandy beach macrofauna along the NW coast of the Iberian Peninsula. Mar Ecol Prog Ser 314:25-33

Ivlev VS (1961) Experimental ecology of the feeding of fishes, Vol. Yale University Press, New Haven, Connecticut, USA

James RJ, Fairweather PG (1996) Spatial variation of intertidal macrofauna on a sandy ocean beach in Australia. Estuarine, Coastal and Shelf Science 43

Janssen GM, Mulder S (2005) Zonation of macrofauna across sandy beaches and surf zones of the Dutch coast. Oceanologia 47:265-282

Janssens F (2002) A checklist of the Belgian springtails. Nomina Collembola Belgica. http://www.collembola.be

Jaramillo E, McLachlan A (1993) Community and population responses of the macroinfauna to physical factors over a range of exposed sandy beaches in south-central Chile. Estuarine, Coastal and Shelf Science 37

Jonckheere I (2011) Gewone zeehonden Phoca vitulina Linnaeus, 1758 langs de Westkust. De Strandvlo: Driemaandelijks Tijdschrift van De Strandwerkgroep België 31

Jones AR, Gladstone W, Hacking NJ (2007) Australian sandy-beach ecosystems and climate change: ecology and management. Australian Zoologist 34:190-201

Jones AR, Murray A, Lasiak TA, Marsh RE (2008) The effects of beach nourishment on the sandy beach amphipod Exoediceros fossor: impact and recovery in Botany Bay, New South Wales, Australia. Marine Ecology 29:28-36

Jones DA (1968) The functional morphology of the digestive system in the carnivorous intertidal isopod *Eurydice*. J Zool 156:363-376

Jones DA (1969) Genus *Eurydice* (Crustacea-Isopoda) in Aegean Sea including *E. longispina* sp nov Cahiers De Biologie Marine 10:15-29

Jordan LKB, Banks KW, Fisher LE, Walker BK, Gilliam DS (2010) Elevated sedimentation on coral reefs adjacent to a beach nourishment project. Marine pollution bulletin 60:261-271

Josefson AB, Forbes TL, Rosenberg R (2002) Fate of phytodetritus in marine sediments: functional importance of macrofaunal community. Mar Ecol Prog Ser 230:71-85

Kaufman W, Pilkey DF (1983) The beaches are moving: The drowning of America's shoreline, Vol. Duke University Press Books

Kersten M, Piersma T (1987) High levels of energy expenditure in shorebirds: metabolic adaptations to an energetically expensive way of life. Ardea 75:175-187

Khayrallah NH, Jones AM (1980) The ecology of *Bathyporeia pilosa* (Amphipoda, Haustoriidae) in the Tay Estuary. 1. Factors influencing the distribution on Tayport and Tentsmuir beaches. Proceedings of the Royal Society of Edinburgh Section B-Biological Sciences 78:S109-S119

Kleypas JA, Buddemeier RW, Archer D, Gattuso J-P, Langdon C, Opdyke BN (1999) Geochemical consequences of increased atmospheric carbon dioxide on coral reefs. Science 284:118-120

Knott DM, Calder DR, Van Dolah RF (1983) Macrobenthos of sandy beach and nearshore environments at Murrells Inlet, South Carolina, U.S.A. Estuarine, Coastal and Shelf Science 16:573-590

Knox GA (2001) The Ecology of Seashores, Vol. CRC Press

Koehler H, Munderloh E, Hofmann S (1995) Soil microarthropods (Acari, Collembola) from beach and dune: characteristics and ecosystem context. Journal of Coastal Conservation 1

Komar PD (2009) Beach processes and sedimentation.

Koot J (2009) Interannual patterns in predation pressure by juvenile plaice *Pleuronectes platessa* L. on intertidal macrobenthos in the Dutch Wadden Sea, Royal Netherlands Institute for Sea Research, Texel, p 21

Kuang C, Pan Y, Zhang Y, Liu S, Yang Y, Zhang J, Dong P (2011) Performance Evaluation of a Beach Nourishment Project at West Beach in Beidaihe, China. Journal of coastal Research 27:769-783

Kuipers BR, Dapper R (1984) Nursery function of Wadden Sea tidal flats for the brown shrimp *Crangon crangon*. Marine ecology progress series 17:171-181

Laffoley DA, Maltby E, Vincent MA, Mee L, Dunn E, Gilliland P, Hamer JP, Mortimer D, Pound D (2004) The ecosystem approach: coherent actions for marine and coastal environments. A report to the UK Government, Vol. English Nature, Peterborough

Laporta LB (2012) Marine biological valuation of the Belgian coast., Ghent University

Leatherman SP (1979) Barrier island handbook, Vol. National Park Service, Cooperative Research Unit, The Environmental Institute, University of Massachusetts at Amherst

Leewis L, van Bodegom PM, Rozema J, Janssen GM (2012) Does beach nourishment have long-term effects on intertidal macroinvertebrate species abundance? Estuarine, Coastal and Shelf Science 113:172-181

Levinton JS (1995) Marine biology: Function, biodiversity, ecology. Oxford University Press, Inc, New York, NY (USA) 423:1995

Lincoln R (1979) British Marine Amphipoda: Gammaridea, Vol. British Museum (Natural History), London

Lockwood SJ (1984) The daily food intake of 0-group plaice (*Pleuronectes platessa* L.) under natural conditions: changes with size and season. J Conseil 41:181-193

Löffler M, Coosen J (1995) Ecological Impact of Sand Replenishment Healy & Doody (eds) Directions in European Coastel Management Samara Publishing Ltd, Cardigan 291-299

M'Harzi AM, Tackx M, Daro MH, Kesaulia I, Caturao R, Podoor N (1998) Winter distribution of phytoplankton and zooplankton around some sandbanks of the Belgian coastal zone. Journal of Plankton Research 20:2031-2052

Maelfait H, Belpaeme K (2007) Het kustkompas, indicatoren als wegwijzers voor een duurzaam kustbeheer, Coördinatiepunt Duurzaam Kustbeheer, Oostende, p 80

Maelfait H, Belpaeme K (2009) The Belgian Coastal Atlas: moving from the classic static atlas to an interactive data-driven atlas. Journal of Coastal Conservation 14:13-19

Maelfait H, Belpaeme K, Lescrauwaet AK, Mees J (2006) Indicators as guides for integrated coastal zone management. VLIZ Special Publication 30

Maelfait J-P, Baert Lo, Janssen M, Alderweireldt M (1998) A Red list for the spiders of Flanders. Bulletin van het Koninklijk Belgisch Instituut voor Natuurwetenschappen, Entomologie 68:131-142

Maes F, De Batist M, Van Lancker VRM, Leroy D, Vincx M (2005a) Towards a Spatial Structure Plan for Sustainable Management of the Sea: Mixed actions-Final report: SPSD II (MA/02/006).

Maes F, Schrijvers J, Vanhulle A (2005b) A Flood of Space. Towards a spatial structure plan for the sustainable management of the North Sea, Vol. Belgian Science Policy, Brussels

Maria TF, De Troch M, Vanaverbeke J, Esteves AM, Vanreusel A (2011a) Use of benthic *vs* planktonic organic matter by sandy-beach organisms: A food tracing experiment with ^{13}C labelled diatoms. Journal of Experimental Marine Biology and Ecology 407:309-314

Maria TF, Esteves AM, Vanaverbeke J, Vanreusel A (2011b) The effect of the dominant polychaete *Scolelepis squamata* on nematode colonisation in sandy beach sediments: An experimental approach. Estuarine, Coastal and Shelf Science 94:272-280

Maria TF, Vanaverbeke J, Esteves AM, De Troch M, Vanreusel A (2012a) The importance of biological interactions for the vertical distribution of nematodes in a temperate ultra-dissipative sandy beach. Estuarine, Coastal and Shelf Science 97:114-126

Maria TF, Vanaverbeke J, Gingold R, Esteves AM, Vanreusel A (2012b) Tidal exposure or microhabitats: what determines sandy-beach nematode zonation? a case study of a macrotidal ridge-and-runnel sandy beach in Belgium. Marine ecology 34:207–217

Martens PM (1983) Three new species of Minoninae (Turbellaria, Proseriata, Monocelididae) from the North Sea, with remarks on the taxonomy of the subfamily. Zoologica scripta 12:153-160

Martens PM (1984) Comparison of three different extraction methods for Turbellaria. Marine Ecology Progress Series 14

Martens PM, Schockaert ER (1986) The importance of turbellarians in the marine meiobenthos: a review. Hydrobiologia 132:295-303

Martin D, Bertasi F, Colangelo MA, de Vries M, Frost M, Hawkins SJ, Macpherson E, Moschella PS, Satta MP, Thompson RC (2005) Ecological impact of coastal defence structures on sediment and mobile fauna: evaluating and forecasting consequences of unavoidable modifications of native habitats. Coastal Engineering 52:1027-1051

Masselink G, Short AD (1993) The effect of tide range on beach morphodynamics and morphology: a conceptual beach model. Journal of Coastal Research:785-800

McDermott JJ (1987) The distribution and food-habits of *Nephtys bucera* Ehlers, 1868, Polychaeta, Nephtyidae) in the surf zone of a sandy beach. Proceedings of the Biological Society of Washington 100:21-27

McLachlan A (1983) Sandy Beach Ecology - A review. In: McLachlan A. ET (eds) Sandy Beaches as Ecosystems. Junk, the Hague, p 59

McLachlan A (1990) Dissipative beaches and macrofauna communities on exposed intertidal sands J Coast Res 6:57-71

McLachlan A (1996) Physical factors in benthic ecology: effects of changing sand particle size on beach fauna. MEPS 131:13

McLachlan A (2001) Coastal Beach Ecosystems In: Encyclopedia of Biodiversity, Vol 1. Academic Press, p 741-751

McLachlan A, Brown AC (2006) The ecology of sandy shores, Vol. Academic Press, Burlington, MA, USA

McLachlan A, De Ruyck A, Hacking N (1996a) Community structure on sandy beaches: patterns of richness and zonation in relation to tide range and latitude. Revista Chilena de Historia Natural 69:451-467

McLachlan A, Defeo O, Jaramillo E, Short AD (2013) Sandy beach conservation and recreation: Guidelines for optimising management strategies for multi-purpose use. Ocean & Coastal Management 71:256-268

McLachlan A, Dorvlo A (2005) Global patterns in sandy beach macrobenthic communities. Journal of Coastal Research:674-687

McLachlan A, Dugan JE, Defeo O, Ansell AD, Hubbard DM, Jaramillo E, Penchaszadeh PE (1996b) Beach clam fisheries. Oceanography and Marine Biology: an annual review 34

McLachlan A, Jaramillo E (1995) Zonation on sandy beaches. Oceanography and Marine Biology - an Annual Review 33:305-335

McLeod KL, Lubchenco J. , Palumbi S. R., Rosenberg A. A. (2005) Scientific Consensus Statement on Marine Ecosystem-Based Management. Signed by 221 academic scientists and policy experts with relevant expertise and published by the Communication Partnership for Science and the Sea

Mee LD, Jefferson RL, Laffoley DdA, Elliott M (2008) How good is good? Human values and Europe's proposed Marine Strategy Directive. Marine Pollution Bulletin 56:187-204

Mees J, Abdulkerim Z, Hamerlynck O (1994) Life history, growth and production of Neomysis integer in the Westerschelde estuary (SW Netherlands). Marine Ecology Progress Series 109

Mees J, Jones MB (1997) The hyperbenthos. Oceanography and marine biology 35:221-256

Melvin SM, Griffin CR, Macivor LH (1991) Recovery strategies for piping plovers in managed coastal landscapes. Coastal Management 19:21-34

Mertens T, Trouw K, Vermander J, Verwaest T, Bolle A, Rouck JD (2008) An Integrated Master Plan for Flanders Future Coastal Safety

Miller DC, Muir CL, Hauser OA (2002) Detrimental effects of sedimentation on marine benthos: what can be learned from natural processes and rates? Ecological Engineering 19:211-232

Misdorp R, Terwindt J (1997) Coastal zone management experiences in the Netherlands.

Modde T, Ross ST (1981) Seasonality of fishes occupying a surf zone habitat in the northern Gulf of Mexico. Fish Bull 78:911-922

Moffett MD, McLachlan A, Winter PED, Ruyck AMC (1998) Impact of trampling on sandy beach macrofauna. Journal of Coastal Conservation 4:87-90

Mouton I (1996) De macrobenthische gemeenschappen van de intergetijdenzone ter hoogte van het natuurreservaat “De Westhoek” (De Pann-Bray Dunes). Ghent University

Mumby PJ, Steneck RS, Hastings A (2012) Evidence for and against the existence of alternate attractors on coral reefs. Oikos 122:481–491

Nasir N, Poxton M (2001) Substratum preferences of juvenile flat fish. Cybium 25:109-117

Naughton SP, Saloman CH (1978) Fishes of the nearshore zone of St. Andrew Bay, Florida, and adjacent coast Northeast Gulf Sci 2:43-55

Nelson WG (1993) Beach Restoration in the Southeastern US: Environmental Effects and Biological Monitoring. Ocean & Coastal Management 19:26

Newell RC, Seiderer LJ, Hitchcock DR (1998) The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the seabed. Oceanogr Mar Biol Ann Rev 36:127-178

Nicholls RJ, Cazenave A (2010) Sea-level rise and its impact on coastal zones. science 328:1517-1520

Nicholls RJ, de la Vega-Leinert AC (2008) Implications of sea-level rise for Europe's coasts: An introduction. Journal of Coastal Research:285-287

Nicolaisen W, Kanneworff E (1969) On the burrowing and feeding habits of the amphipods *Bathyporeia pilosa* Lindstrom and *Bathyporeia sarsi* Watkin. Ophelia 6:231-250

Nicolaisen W, Kanneworff E (1983) Annual variations in vertical distribution and density of *Bathyporeia pilosa* Lindström and *Bathyporeia sarsi* Watkin at Julebaek (North-Sealand, Denmark). Ophelia 22:237-251

Noss RF (1990) Indicators for Monitoring Biodiversity: A Hierarchical Approach. Conservation Biology 4

Oksanen M (1997) The moral value of biodiversity. Ambio:541-545

Oliver JS, Slattery PN (1976) Effects of dredging and disposal on some benthos at Monterey Bay, California. Tech Pap U S Army Coast Eng Res Cent

Paine RT (1980) Food webs: linkage, interaction strength and community infrastructure. J Anim Ecol 49:667-685

Parr T, Diener D, Lacy S (1978) Effects of beach replenishment on the nearshore sand fauna at Imperial beach, California U.S. Army, Corps of Engineers, Coastal Engineering Research Center, Fort Belvoir

Pascual M, Borja A, Eede SV, Deneudt K, Vincx M, Galparsoro I, Legorburu I (2011) Marine biological valuation mapping of the Basque continental shelf (Bay of Biscay), within the context of marine spatial planning. Estuarine, Coastal and Shelf Science 95:186-198

Pascual M, Borja A, Franco J, Burdon D, Atkins JP, Elliott M (2012) What are the costs and benefits of biodiversity recovery in a highly polluted estuary? Water research 46:205-217

Pearce D, Moran D (1994) The economic value of biodiversity.

Pearson TH, Mannvik H-P (1998) Long-term changes in the diversity and faunal structure of benthic communities in the northern North Sea: natural variability or induced instability? Hydrobiologia 375/376:13

Perry CT, Murphy GN, Kench PS, Smithers SG, Edinger EN, Steneck RS, Mumby PJ (2013) Caribbean-wide decline in carbonate production threatens coral reef growth. Nature communications 4:1402

Persson LE (1982) Seasonal migration of *Bathyporeia pilosa* Lindström in the southern Baltic. Ophelia 21:205-213

Peterson CH, Hickerson DHM, Johnson GG (2000) Short-term consequences of nourishment and bulldozing on the dominant large invertebrates of a sandy beach. Journal of coastal Research 16:11

Peterson CH, Manning L (2001) How beach nourishment affects the habitat value of intertidal beach prey for surf fish and shorebirds and why uncertainty still exists. Proceedings of the Coastal Ecosystems and Federal Activities Technical Training Symposium

Phillips MR, Jones AL (2006) Erosion and tourism infrastructure in the coastal zone: Problems, consequences and management. Tourism Management 27:517-524

Pikitch EK, Santora C, Babcock EA, Bakun A, Bonfil R, Conover DO, Dayton P, Doukakis P, Fluharty D, Heneman B (2004) Ecosystem-based fishery management. Science 305:346-347

Pilkey OH (1998) The North Carolina shore and its barrier islands: restless ribbons of sand, Vol. Duke University Press

Pilkey OH, Clayton TD (1989) Summary of beach replenishment experience on US East Coast barrier islands. Journal of Coastal Research:147-159

Pilkey OH, Dixon KL (1998) The Corps and the shore, Vol. Island Press, Washington, D.C.

Pomeroy R, Douvere F (2008) The engagement of stakeholders in the marine spatial planning process. Marine Policy 32:816-822

Powilleit M, Kleine J, Leuchs H (2006) Impacts of experimental dredged material disposal on a shallow, sublittoral macrofauna community in Mecklenburg Bay (western Baltic Sea). Marine pollution bulletin 52:386-396

Provoost S, Ampe C, Bonte D, Cosyns E, Hoffmann M (2004) Ecology, management and monitoring of grey dunes in Flanders. Journal of Coastal Conservation 10:33-42

Provoost S, Jones MLM, Edmondson SE (2011) Changes in landscape and vegetation of coastal dunes in northwest Europe: a review. Journal of Coastal Conservation 15:207-226

Rabaut M, Guilini K, Van Hoey G, Vincx M, Degraer S (2007) A bio-engineered soft-bottom environment: The impact of Lanice conchilega on the benthic species-specific densities and community structure Estuarine, Coastal and Shelf Science

Rabaut M, Van de Moortel L, Vincx M, Degraer S (2010) Biogenic reefs as structuring factor in Pleuronectes platessa (Plaice) nursery. Journal of Sea Research 64:102-106

Radenac G, Miramand P, Tardy J (1997) Search for impact of a dredged material disposal site on growth and metal contamination of *Mytilus edulis* (L.) in Charente-Maritime (France). Marine pollution bulletin 34:721-729

Rappé G (1983) Het voorkomen van de grijze zeehond langs de Belgische kust. De Strandvlo: Driemaandelijks Tijdschrift van De Strandwerkgroep België 3

Rappé G (1996) Verspreiding en populatiedynamiek van thalassochore zaadplanten aan de Belgische kust. Dumortiera 64-65:8-13

Rappé G (1997) Vloedmerken. In: Hermy M, De Blust G (eds) Punten en lijnen in het landschap. Marc Van de Wiele, Brugge, p 48-49

Regeringsbesluit (2010) Besluit van 23 juli 2010 van de Vlaamse Regering tot vaststelling van gewestelijke instandhoudingsdoelstellingen voor Europees te beschermen soorten en habitats. , Belgisch Staatsblad, 5 november 2010

Rego T (2007) A Biological Valuation of the Pico e Faial Channel. University of Azores, Portugal

Reilly FJJ, Bellis VJ (1983) The ecological impact of beach nourishment with dredged materials on the intertidal zone at Bogue Banks, north Carolina, U.S. Army, Corps of Engineers, Coastal Engineering Research Center, Fort Belvoir

Ricciardi A, Bourget E (1999) Global patterns of macroinvertebrate biomass in marine intertidal communities. Marine Ecology Progress Series 185

Riddell KJ, Young SW (1992) The management and creation of beaches for coastal defence. Water and Environment Journal 6:588-597

Roberts RD, Forrest BM (1999) Minimal impact from long-term dredge spoil disposal at a dispersive site in Tasman Bay, New Zealand. New Zealand journal of marine and freshwater research 33:623-633

Robertson AI, Mann KH (1980) The role of isopods and amphipods in the initial fragmentation of eelgrass detritus in Nova Scotia, Canada. Marine Biology 59:63-69

Rodil IF, Compton TJ, Lastra M (2012) Exploring Macroinvertebrate Species Distributions at Regional and Local Scales across a Sandy Beach Geographic Continuum. PloS one 7:e39609

Rodil IF, Lastra M, Sanchez-Mata AG (2006) Community structure and intertidal zonation of the macroinfauna in intermediate sandy beaches in temperate latitudes: North coast of Spain. Estuar Coast Shelf Sci 67:267-279

Roode N, Baarse G, Ash J, Salado R (2008) Coastal flood risk and trends for the future in the North Sea region: results and recommendations of Project Safecoast: Synthesis report, The Hague, p 136

Rosa LC, Bemvenuti CE (2006) Temporal variability of the estuarine macrofauna of the Patos Lagoon, Brazil. Revista de Biología Marina y Oceanografía 41:1-9

Ross ST, McMichael Jr RH, Ruple DL (1987) Seasonal and diel variation in the standing crop of fishes and macroinvertebrates from a Gulf of Mexico surf zone. Estuarine, Coastal and Shelf Science 25:391-412

Ruddick K, Ovidio F, Van den Eynde D, Vasilkov A (1998) The distribution and dynamics of suspended particulate matter in Belgian coastal waters derived from Avhrr imagery Proceedings of the ninth Conference Satellite Meteorology and Oceanography, Paris, p 626-629

Sabbe K (1997) Systematics and ecology of intertidal benthic diatoms of the Westerschelde estuary (The Netherlands). Ph. D. thesis, University of Ghent: 206 pp

Salmane I (2000) Fauna of soil-dwelling predatory Gamasina mites (Acari: Mesostigmata) in seashore habitats of the Kurzeme coast, Latvia. Ekologia(Bratislava)/Ecology(Bratislava) 19:87-96

Saloman CH, Naughton SP (1979) Fishes of the littoral zone, Pinellas County, Florida. Florida Sci 42:85-93

Saloman CH, Naughton SP, Taylor JL (1982) Benthic community response to dredging borrow pits, Panama City Beach, Florida, National Marine Southeast Fisheries Center, Panama City

Salvat B (1964) Les conditions hydrodynamiques interstitielles des sédiments meubles intertidaux et la répartition verticale de la faune endogée. Comptes Rendus de l'Académie des Sciences Paris 259:1576–1579

Savage C, Field JG, Warwick RM (2001) Comparative meta-analysis of the impact of offshore marine mining on macrobenthic communities versus organic pollution studies. Mar Ecol Prog Ser 221:265-275

Schlacher TA, Dugan J, Schoeman DS, Lastra M, Jones A, Scapini F, McLachlan A, Defeo O (2007) Sandy beaches at the brink. Diversity and Distributions 13:556-560

Schlacher TA, Noriega R, Jones A, Dye T (2012) The effects of beach nourishment on benthic invertebrates in eastern Australia: Impacts and variable recovery. Science of The Total Environment 435-436:411-417

Schlacher TA, Schoeman DS, Dugan J, Lastra M, Jones A, Scapini F, McLachlan A (2008) Sandy beach ecosystems: key features, sampling issues, management challenges and climate change impacts. Mar Ecol 29:70-90

Schoener TW (1983) Field experiments on interspecific competition. Am Nat 122:240-285

Seys J (2001) Sea-and coastal bird data as tools in the policy and management of Belgian marine waters. University of Ghent

Short AD (1991) Macro-meso tidal beach morphodynamics: An overview. Journal of Coastal Research 7:417-436

Short AD (2006) Australian beach systems-nature and distribution. Journal of Coastal Research:11-27

Short AD, Wright LD (1983) Physical variability of sandy beaches. Developments in hydrobiology 19

Slim P, Löffler M (2007) Kustveiligheid en natuur: een overzicht van kennis en kansen. Alterra-Rapport 1485:1-92

Slott JM, Murray AB, Ashton AD, Crowley TJ (2006) Coastline responses to changing storm patterns.

Smit C, Wolff W (1981) Birds of the Wadden Sea, Vol. Balkema, Rotterdam

Smith EP (2002) BACI design. Encyclopedia of environmetrics

Smith EP, Orvos DR, Cairns Jr J (1993) Impact assessment using the before-after-control-impact (BACI) model: concerns and comments. Canadian Journal of Fisheries and Aquatic Sciences 50:627-637

Snickars M, Gullström M, Sundblad G, Bergström U, Downie AL, Lindegarth M, Mattila J (2013) Species-environment relationships and potential for distribution modelling in coastal waters. Journal of Sea Research

Sokal RR, Rohlf FJ (1995) Biometry WH Freeman, Vol, New York

Somerfield PJ, Rees HL, Warwick RM (1995) Interrelationships in community structure between shallow-water marine meiofauna and macrofauna in relation to dredgings disposal. Mar Ecol Prog Ser 127:103-112

Souza JRB, Gianuca NM (1995) Zonation and seasonal-variation of the intertidal macrofauna on a sandy beach of Parana state, Brazil. Scientia Marina 59:103-111

Spanoghe G (1999) Aantallen en verspreiding, gedrag en habitatkeuze van meeuwen (Laridae) aan de Vlaamse Kust in het winterhalfjaar. Licentiaatsthesis Universiteit Gent, Gent

Spanoghe G, Devos K (2002) Totaaltellingen van meeuwen langs de Belgische kust. Natuur Oriolus 68

Speybroeck J (2007) Ecology of macrobenthos as a baseline for an ecological adjustment of beach nourishment. UGent

Speybroeck J, Alsteens L, Vincx M, Degraer S (2007) Understanding the life of a sandy beach polychaete of functional importance--Scolelepis squamata (Polychaeta: Spionidae) on Belgian sandy beaches (northeastern Atlantic, North Sea) Estuarine, Coastal and Shelf Science

Speybroeck J, Bonte D, Courtens W, Gheskiere T, Grootaert P, Maelfait JP, Mathys M, Provoost S, Sabbe K, Stienen E (2005a) How may beach nourishment affect the sandy beach ecosystem? The case of Belgian beaches In: Herrier J-L, Mees J, Salman A, Seys J, Van Nieuwenhuyse H, Dobbelaere I (eds) 'Dunes and Estuaries 2005' – International Conference on Nature Restoration Practices in European Coastal Habitats, Koksijde, Belgium, p 557-568

Speybroeck J, Bonte D, Courtens W, Gheskiere T, Grootaert P, Maelfait JP, Mathys M, Provoost S, Sabbe K, Stienen EWM (2006a) Beach nourishment: an ecologically sound coastal defence alternative? A review Aquatic Conservation: Marine and Freshwater Ecosystems 16:419-435

Speybroeck J, Bonte D, Courtens W, Gheskiere T, Grootaert P, Maelfait JP, Provoost S, Sabbe K, Stienen EWM, Van Lancker V, Van Landuyt W, Vincx M, Degraer S (2008a) The Belgian sandy beach ecosystem: a review. Marine Ecology - an Evolutionary Perspective 29:171-185

Speybroeck J, Bonte D, Dasseville R, Gheskiere T, Grootaert P, Lionard M, Maelfait JP, Sabbe K, Stienen EWM, Van den Broeck K (2005b) Biological evaluation of eleven beach zones along the Flemish coast-BEST: Final report 181

Speybroeck J, Bonte D, Grootaert P, Maelfait JP, Stienen E, Vandomme V, Vanermen N, Vincx M, Degraer S (2006b) Studie over de impact van zandsuppleties op het ecosysteem – fase 2 – Dossiernummer 204.295 Ministerie van de Vlaamse Gemeenschap Departement Leefmilieu en Infrastructuur Agentschap voor Maritieme Dienstverlening en Kust, Afdeling Kust p118

Speybroeck J, Degraer S, Vincx M (2003) Ecologische monitoring kustverdedigingsproject Oostende (t0 situatie, fase 1) – Dossiernummer 202.290, Ministerie van de Vlaamse Gemeenschap, Departement Leefmilieu en Infrastructuur, Administratie Waterwegen en Zeewezen, Afdeling Waterwegen Kust (AWK) p64

Speybroeck J, Van Tomme J, Vincx M, Degraer S (2008b) In situ study of the autecology of the closely related, co-occurring sandy beach amphipods Bathyporeia pilosa and Bathyporeia sarsi Helgoland Marine Research

Spiegelhalter D, Thomas A, Best N, Lunn D (2003) WinBUGS user manual. Cambridge: MRC Biostatistics Unit

Stauble DK, Nelson WG (1985) Guidelines for beach nourishment: a necessity for project management. Proceedings of the Fourth Symposium on Coastal and Ocean Management "Coastal Zone '85" ASCE/Baltimore, MD/July 30 - Aug 1, 1985:20

Stenton-Dozey JME, Griffiths CL (1983) The fauna associated with kelp stranded on a sandy beach. Developments in Hydrobiology 19

Stienen E, Van Waeyenberge J (2004) Kustbroedvogels: teruggedrongen dynamiek en toegenomen verstoring. In Provoost, S et al (Ed) (2004) Levende duinen: een overzicht van de biodiversiteit aan de Vlaamse kust Mededelingen van het Instituut voor Natuurbehoud 22:176-183

Stienen EW, Van Waeyenberge JZ, 14 juni 2002. VLIZ Special Publication 9:1–9. (2002) Verstoken van verstoring: het belang van de "Baai van Heist" als rust- en broedgebied voor vogels. In: Mees J (ed) Academische studiedag: 5 Jaar strand-natuurreservaat "De Baai van Heist" - De Vlaamse stranden: steriele zandbakken of natuurpatrimonium?

Stienen EWM, Courtens W, Van Waeyenberge J, Kuijken E (2005) Harbouring nature: port development and dynamic birds provide clues for conservation. VLIZ Special Publication 19

Stoner AW, Ottmar ML (2003) Relationships between size-specific sediment preferences and burial capabilities in juveniles of two Alaska flatfishes. J Exp Mar Biol Ecol 282:85-101

Stuer V (2002) Trofische interacties in relatie tot zwinnen van de Belgische kust (Schipgatduinen Koksijde): epibenthos en avifauna. University of Gent

Sundbäck K, Persson LE (1981) The effect of microbenthic grazing by an amphipod, Bathyporeia pilosa, Lindström. Kieler Meeresforschungen Sonderheft 5:573-575

Szaniawska A (1983) Seasonal changes in energy content of *Crangon crangon* L. (Crustacea, Decapoda). POL ARCH HYDROBIOL 30:45-56

Turbeville DB, Marsh GA (1982) Benthic Fauna of an Offshore Borrow Area in Broward County, Florida, DTIC Document

Underwood AJ (1991) Beyond BACI: experimental designs for detecting human environmental impacts on temporal variations in natural populations. Marine and Freshwater Research 42:569-587

Underwood AJ (1992) Beyond BACI: the detection of environmental impacts on populations in the real, but variable, world. Journal of Experimental Marine Biology and Ecology 161:145-178

Underwood AJ (1994) On Beyond BACI: Sampling Designs that Might Reliably Detect Environmental Disturbances. Ecological Applications 4:4-15

Underwood GJC, Kromkamp J (1999) Primary production by phytoplankton and microphytobenthos in estuaries. Advances in ecological research 29:93-153

Vader W (1965) Intertidal distribution of haustoriid amphipods in The Netherlands. Bot gothoburg 3:233-246

Vader WJM (1966) Een overzicht van de zandbewonende amphipoden uit het Oosterschelde gebied. Het Zeepaard

Van Cauwenberghe C, Dekker L, Schuurman A (1993) M2 tidal reduction method for coastal zones, Ministerie van Openbare Werken. Dienst der Kust, Oostende, p 1-12, 17 app.

Van Dalfsen JA, Essink K (2001) Benthic community response to sand dredging and shoreface nourishment in Dutch coastal waters. In: Kroncke ITMSJ (eds) Burning Issues of North Sea Ecology, Vol 31, p 329-332

van der Molen J, van Dijck B (2000) The evolution of the Dutch and Belgian coasts and the role of sand supply from the North Sea. Global and Planetary Change 27:223-244

Van Dolah RF, Calder DR, Knott DM (1984) Effects of dredging and open-water disposal on benthic macroinvertebrates in a South Carolina estuary. Estuaries and Coasts 7:28-37

Van Ginderdeuren K, Maene S, Vincx M, Degraer S (2007) Ecologische monitoring kustverdedigingsproject Oostende (t0 situatie, fase 3) – Dossiernummer 205.240, Ministerie van de Vlaamse Gemeenschap, Departement Leefmilieu en Infrastructuur, Agentschap voor Maritieme Dienstverlening en Kust (MD&K) p103

Van Gompel J (1983) Jaarverslag zeezoogdieren 1982. De Strandvlo: Driemaandelijks Tijdschrift van De Strandwerkgroep België 3:47-49

Van Hoey G (2006) Spatio – temporal variability within the macrobenthic Abra alba community, with emphasis on the structuring role of Lanice conchilega. UGent

Van Hoey G, Degraer S, Vincx M (2004) Macrobenthic community structure of soft-bottom sediments at the Belgian Continental Shelf. Estuarine, Coastal and Shelf Science

Van Hoey G, Derweduwen J, Hillewaert H, Hostens K, Pecceu E, Wittoeck J (2010) Ecological status evaluation of the quality element macro-invertebrates for the Belgian Coast (2007-2009). Report, ILVO-Animal Sciences-Fisheries

Van Hoey G, Vincx M, Degraer S (2005) Small- to large-scale geographical patterns within the macrobenthic Abra alba community Estuarine, Coastal and Shelf Science

Van Hoey G, Vincx M, Degraer S (2007a) Temporal variability in the Abra alba community determined by global and local events Peer reviewed article Journal of Sea Research

Van Hoey G, Ysebaert T, Herman P (2007b) Update of the assessment of the Belgian coastal waters with level 3 of the BEQI (Benthic Ecosystem Quality Index) method, Belgium, p 28

Van Lancker V, Du Four I, Verfaillie E, Deleu S, Schelfaut K, Fettweis M, Van den Eynde D, Francken F, Monbaliu J, Giardino A (2007) Management, research and budgetting of aggregates in shelf seas related to end-users (Marebasse).

Van Landuyt W, Hoste I, Vanhecke L, Van den Bremt P, Vercruysse W, De Beer De (2006) Atlas van de Flora van Vlaanderen en het Brussels gewest, Vol. Nationale Plantentuin en het Instituut voor Natuur- en Bosonderzoek i.s.m. Flo.Wer vzw, Brussel

Van Nieuwenhuyse H (2003) Beheersplan voor het Vlaams Natuurreservaat Ter Yde te Koksijde (Oostduinkerke).

van Riel MC, Healy EP, van der Velde G (2007) Interference competition among native and invader amphipods. Acta oecologica 31:282-289

Van Tomme J (2013) Biotic interactions within sandy beach ecosystems, with implications for an ecologically-sound beach nourishment Ghent University

Van Tomme J, Van Colen C, Degraer S, Vincx M (2012a) Encounter competition partly explains the segregation of the sandy beach amphipods *Bathyporeia pilosa* and *Bathyporeia sarsi*. A mesocosm experiment. Journal of Experimental Marine Biology and Ecology 438:118-124

Van Tomme J, Vanden Eede S, Speybroeck J, Degraer S, Vincx M (2012b) Macrofaunal sediment selectivity considerations for beach nourishment programmes. Marine Environmental Research 84:10-16

Van Waeyenberge J, Stienen EWM, Offringa H (2001) Overwinterende zee-eenden voor de Belgische kust. Vogelnieuws: Ornithologische Nieuwsbrief van het Instituut voor Natuurbehoud 1:20-23

Vanaverbeke J, Beuselinck B, Braeckman U, Van Colen C, De Troch M, Moens T, Rigeaux A, Schram D, Van Gansbeke D, Vincx M, Degraer S (2008) Ecologische Monitoring Kustverdedigingsproject Oostende (t0-situatie, fase 3). Aanvullende studie: Kwantificering en kwalificering van organisch materiaal in mariene sedimenten: hun onderlinge relaties en de relatie tot de korrelgrootteverdeling. Dossiernummer 205.240, MDK

Vanden Eede S (2007) Marine biological valuation of the Isles of Scilly archipelago. Universiteit Gent, MARELAC

Vanden Eede S, Bonte D, Vincx M (2013, in prep.) Assessment of the ecological characteristics of the Belgian beaches prior to the implementation of the Belgian Master Plan for Coastal Safety.28 + 25

Vanden Eede S, Colson L, Vincx M (2013) Ecologische monitoring OW plan Oostende en geïntegreerd kustveiligheidsplan, Lombardsijde en Oostende (t1 situatie – tweede fase – 2012). Eindrapport – Dossiernummer 211.095, Vlaamse Overheid, Agentschap voor Maritieme Dienstverlening en Kust; Afdeling Kust, p 118

Vanden Eede S, Vincx M (2010) Ecologische monitoring natuurinrichtingsproject Lombardsijde (t1 situatie, 2009) – Dossiernummer 209.090, Vlaamse Overheid, Agentschap voor Maritieme Dienstverlening en Kust, Afdeling Kust p71

Vanden Eede S, Vincx M (2011a) Ecological monitoring of the beach nourishment project Lombardsijde. VLIZ Special Publication 48

Vanden Eede S, Vincx M (2011b) Ecologische monitoring natuurinrichtingsproject Lombardsijde en OW-plan Oostende (t1 situatie, 2010) – Dossiernummer 209.090, Vlaamse Overheid, Agentschap voor Maritieme Dienstverlening en Kust, Afdeling Kust p91

Vanden Eede S, Vincx M, Degraer S (2008) Ecologische monitoring natuurinrichtingsproject Lombardsijde (t0 situatie, fase 4) – Dossiernummer 208.245, Vlaamse Overheid, Agentschap voor Maritieme Dienstverlening en Kust, Afdeling Kust p49

Vandenbohede A, Lebbe L (2004) Density dependent groundwater flow model of the shore and dune area of the Westhoek nature reserve (Belgium).

Vandepitte L, Decock W, Mees J (2010a) Belgian Register of Marine Species, compiled and validated by the VLIZ Belgian Marine Species Consortium. VLIZ Special Publication 46

Vandepitte L, Vanhoorne B, Kraberg A, Anisimova N, Antoniadou C, Araújo R, Bartsch I, Beker B, Benedetti-Cecchi L, Bertocci I (2010b) Data integration for European marine biodiversity research: creating a database on benthos and plankton to study large-scale patterns and long-term changes. Hydrobiologia 644:1-13

Vandewalle S (2009) Levensgeschiedenis, habitat en zonatiepatronen van de Agaatpissebed (*Eurydice pulchra*)(Crustacea: Isopoda: Cirolanidae) op Belgische stranden. Ghent University

Vanermen N, Stienen EWM, De Meulenaer B, Van Ginderdeuren K, Degraer S (2009) Low dietary importance of polychaetes in opportunistic feeding Sanderlings *Calidris alba* on Belgian beaches. Ardea 97:81-87

Veloso VG, Caetano CHS, Cardoso RS (2003) Composition, structure and zonation of intertidal macroinfauna in relation to physical factors in microtidal sandy beaches in Rio de Janeiro state, Brazil. Scientia Marina 67:393-402

Veloso VG, Cardoso RS (2001) Effect of morphodynamics on the spatial and temporal variation of macrofauna on three sandy beaches. J Mar Biol Ass UK 81:369-375

Veloso VrG, Silva ES, Caetano CHS, Cardoso RS (2006) Comparison between the macroinfauna of urbanized and protected beaches in Rio de Janeiro State, Brazil. Biological Conservation 127:510-515

Verfaillie E, Du Four I, Van Meirvenne M, Van Lancker V (2009) Geostatistical modeling of sedimentological parameters using multi-scale terrain variables: application along the Belgian Part of the North Sea. International journal of geographical information science 23:135-150

Verfaillie E, Van Lancker V, Van Meirvenne M (2006) Multivariate geostatistics for the predictive modelling of the surficial sand distribution in shelf seas. Continental Shelf Research 26:2454-2468

Vermeersch G, Anselin A, Devos K, Herremans M, Stevens J, Gabriëls J, Van Der Krieken B (2004) Atlas van de Vlaamse broedvogels 2000-2002. Mededelingen van het Instituut voor Natuurbehoud 2004(23), Vol 23, Brussel

Volckaert AM (1998) Het macrobenthos van de intergetijdenzone van de Vlaamse stranden. Ghent University

Welvaert G (2005) Ecologische monitoring kustverdedigingsproject Oostende (t0 situatie, fase 2) – Rapportnummer 04/A2084-3, Ministerie van de Vlaamse Gemeenschap, Departement Leefmilieu en Infrastructuur, Administratie Waterwegen en Zeewezen, Afdeling Waterwegen Kust p166

Wentworth CK (1922) A Scale of Grade and Class Terms for Clastic Sediments. The Journal of Geology 30:377-392

Weslawski JM, Warzocha J, Wiktor J, Urbañski J, Bradtke K, Kryla L, Tatarek A, Kotwicki L, Piwowarczyk J (2009) Biological valorisation of the southern Baltic Sea (Polish Exclusive Economic Zone). Oceanologia 51:415-435

Wilber DH, Clarke DG, Ray GL, Burlas M (2003) Response of surf zone fish to beach nourishment operations on the northern coast of New Jersey, USA. Marine Ecology Progress Series 250

Willems W, Goethals P, Van den Eynde D, Van Hoey G, Van Lancker V, Verfaillie E, Vincx M, Degraer S (2008) Where is the worm? Predictive modelling of the habitat preferences of the tube-building polychaete Lanice conchilega Ecological Modelling

Wilson E (1986) Biodiversity National Forum on Biodiversity Washington DC

Witt J, Schroeder A, Knust R, Arntz WE (2004) The impact of harbour sludge disposal on benthic macrofauna communities in the Weser estuary. Helg Mar Res 58:117-128

Worm B, Barbier EB, Beaumont N, Duffy JE, Folke C, Halpern BS, Jackson JBC, Lotze HK, Micheli F, Palumbi SR (2006) Impacts of biodiversity loss on ocean ecosystem services. Science 314:787-790

Young OR, Osherenko G, Ekstrom J, Crowder LB, Ogden J, Wilson JA, Day JC, Douvere F, Ehler CN, McLeod KL (2007) Solving the crisis in ocean governance: Place-based management of marine ecosystems. Environment: science and policy for sustainable development 49:20-32

Young RW, Bryant EA (1992) Catastrophic wave erosion on the southeastern coast of Australia: Impact of the Lanai tsunamis ca. 105 ka? Geology 20:199-202

Ysebaert T, Meire P (1989) Uitkomstsukses en foerageergedrag van kokmeeuwen in drie kolonies gelegen langs de Westerschelde, p 95

Zacharias MA, Roff JC (2000) A Hierarchical Ecological Approach to Conserving Marine Biodiversity. Blackwell Science Inc, p 1327-1334

Zeidler RB (1987) Discussion of "Suspended Sediment Transport and Beach Profile Evolution" by William R. Dally and Robert G. Dean (February, 1984, Vol. 110, No. 1). Journal of Waterway, Port, Coastal, and Ocean Engineering 113:89-95

Zwarts L, Wanink JH, Ens BJ (1996) Predicting seasonal and annual fluctuations in the local exploitation of different prey by Oystercatchers *Haematopus ostralegus*: a ten year study in the Wadden Sea. Ardea 84A:401-440

Appendices

Appendices – Chapter 1

A myriad of laws and Directives apply to the BPNS and its waters, habitats and species (Cliquet et al. 2011). A detailed overview of Belgium's most important international and European environmental obligations is given below.

International obligations

The **Convention on Wetlands of International Importance, especially as Waterfowl Habitat** (Ramsar 1971) provides the framework for national action and international cooperation for the conservation and wise use of wetlands and their resources. The Ramsar Convention is the only global environmental treaty that deals with a particular ecosystem. Wetlands are broadly defined. The term comprises human-made sites such as fish ponds, rice paddies, reservoirs and salt pans and numerous natural sites, including lakes, rivers, swamps, marshes, wet grasslands, peatlands, oases, estuaries, deltas, tidal flats, near-shore marine areas, mangroves and coral reefs. The wise use of wetlands is defined as the maintenance of their ecological character, achieved through the implementation of ecosystem approaches, within the context of sustainable development.

The **Convention on the Conservation of Migratory Species of Wild Animals** (CMS or Bonn Convention 1979) aims to conserve terrestrial, aquatic and avian migratory species throughout their range, under the auspices of the United Nations Environment Program (UNEP). Migratory species threatened with extinction are listed in Appendix I. CMS promotes concerted action within and between Member States to strictly protect these animals, conserving or restoring the places where they live, mitigating obstacles to migration and controlling other factors that might endanger them. Migratory species that need or would significantly benefit from international co-operation are listed in Appendix II. Global or regional cooperation between Member States are encouraged. CMS always acts as a framework Convention. This cooperation may range from legally binding treaties or agreements to less formal instruments, such as Memoranda of Understanding, and can be adapted to the requirements of particular regions. Several Agreements have been concluded to date under the auspices of CMS. ASCOBANS aims to conserve Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas while AEWA tries to protect African-Eurasian Migratory Waterbirds.

The **United Nations Convention on the Law of the Sea** (UNCLOS 1982) is a comprehensive regime of law and order for the world's oceans and seas, governing all uses and resources. All problems of ocean space are closely interrelated and need to be addressed as a whole. UNCLOS governs all aspects of ocean space, such as delimitation, environmental control, marine scientific research, economic and commercial activities, technology transfer and settlement of disputes relating to ocean matters. Coastal States have sovereign rights in a 200 nautical mile exclusive economic zone (EEZ) with respect to natural resources and certain economic activities, and exercise jurisdiction over marine science research and environmental protection. All other States have freedom of navigation in the EEZ, as well as freedom to lay submarine cables and pipelines. Unfortunately, these maritime zones do not coincide with ecosystem boundaries. UNCLOS also includes a number of provisions concerning the marine environment in relation to pollution, alien species, global and regional cooperation and highly migratory species.

The **Convention on Biological Diversity** (CBD, Rio de Janeiro 1992) is the most comprehensive and significant international instrument addressing the threats to marine and coastal biodiversity. CBD prescribes the ecosystem approach to sustainably protect, understand and use marine resources. The implementation must be consistent with the UNCLOS. Objectives are to be met through the implementation of a number of measures including (1) development of national strategies, (2) integration of biodiversity considerations into sectoral and cross-sectoral plans, (3) establishment of monitoring programs and (4) extensive measures for *in situ* and *ex situ* conservation, e.g. controlling alien organisms and restoring degraded ecosystems. As an important element of the overall approach, CBD requires all Member States to establish a system of protected areas and to develop guidelines for their selection, establishment and management. The Jakarta Mandate on Marine and Coastal Biodiversity (1995) formed a valuable part of the implementation of the CBD. The five identified thematic issues focus on the relationships between conservation, the use of biological diversity and fishing activities: integrated marine and coastal area management, marine and coastal protected areas, sustainable use of marine and coastal living resources, mariculture and alien species. In summary, the CBD aims at achieving a significant reduction in the current rate of biodiversity loss and a sustainable development of marine ecosystems through the application of the ecosystem approach by 2010. During the World Summit on Sustainable Development (WSSD; Johannesburg 2002), this target was confirmed in the plan of implementation and Member States committed themselves to promote integrated, multisectoral, coastal and ocean management at the national level. The goals and aims of CBD and other biodiversity-related conventions, particularly CMS and Ramsar are mutually reinforcing. CMS is even recognized as the lead partner to CBD in conserving migratory species.

Full implementation of the **United Nations Agenda 21 program** (UN 1992) was also affirmed during the WSSD in 2002. Agenda 21 clearly prescribes new precautionary and anticipatory approaches to marine and coastal area management and development, at the national, (sub)regional and global levels. Among the program areas are integrated management and sustainable development of coastal and marine areas, environmental protection, sustainable use and conservation of marine and coastal living resources. Chapter 17 deals with the protection of oceans and sets a number of concrete objectives, some of which relate directly to spatially managed areas (SMAs) and marine spatial planning (MSP): (1) integrate policy and decision making process, including all involved sectors, to promote compatibility and a balance of uses, (2) identify existing and projected uses of coastal areas and their interactions and (3) apply preventive and precautionary approaches in project planning and implementation, including prior assessment and systematic observation of the impacts of major projects.

The **Convention for the Protection of the marine Environment of the North-East Atlantic** (OSPAR 1992) has worked to identify threats to and protection of the marine environment of the North-East Atlantic and its Regions: Arctic Waters, the Greater North Sea including the English Channel, the Celtic Seas, the Bay of Biscay and Iberian Coast, and the Wider Atlantic including the waters surrounding the Azores. It started in 1972 with the Oslo Convention against dumping and was broadened to cover land-based sources and the offshore industry by the Paris Convention of 1974. Both conventions were unified, up-dated and extended by the OSPAR Convention. The new annex on biodiversity and ecosystems was adopted in 1998 to cover non-polluting human activities that can adversely affect the sea. OSPAR has defined broad environmental goals which collectively aim at a clean, healthy, biologically diverse and productive sea. Commonly agreed criteria, methodological standards and monitoring guidelines have been developed for evaluating the status of the marine environment and the impacts of human activities. These tools apply coherently across the entire OSPAR maritime area, while taking into account environmental differences between regions. Building on a 35-year track of experience and long-standing cooperation with the International Council for the Exploration of the Sea (ICES), a common science basis has been developed. It supports holistic and thematic evaluations of the quality status of the North-East Atlantic against defined targets and allows for an integrated ecosystem assessment.

European obligations

The **Birds Directive** (BD 1979, changed in 2009: Directive 2009/147/EC) states that the population of the natural wild birds (species referred to in Article 1) should be maintained at a level which corresponds in particular to ecological, scientific and cultural requirements, while taking account of economic and recreational requirements. The preservation, maintenance and re-establishment of their biotopes and habitats should include the creation of new protected areas and biotopes, the upkeep and management of these areas in accordance with their ecological needs inside and outside protected zones and the re-establishment of destroyed biotopes. The objectives are ensured by the selection, designation and protection of a network of **Special Protection Areas** (SPAs) designated to protect wild birds throughout Europe.

The aims of the **Convention on the Conservation of European Wildlife and Natural Habitats** (Bern Convention 1982) are to conserve wild flora and fauna and their natural habitats, and to promote conservation cooperation between several states. Particular emphasis is given to endangered and vulnerable (migratory) species. The Convention lists protected species in four Appendices. Appendix I and II list respectively strictly protected flora and fauna species. Appendix III lists protected fauna species and Appendix IV lists prohibited means and methods of killing, capture and other forms of exploitation.

The convention led to the creation in 1998 of the Emerald network of Areas of Special Conservation Interest (ASCIs) throughout the territory of Member States.

To fulfill its obligations arising from the Bern Convention, particularly in respect of habitat protection, the European Union produced the **Habitats Directive** (HD 1992: Directive 92/43/EEC). This directive ensures biodiversity through the conservation, maintenance and restoration of natural habitats and wild fauna and flora of community interest. The objectives are ensured by the selection, designation and protection of a network of **Special Areas of Conservation** (SACs) throughout Europe. Potential SACs were selected depending upon a list of Annex I habitats and Annex II species of community importance. Member States were asked to draw up national lists of Sites of Community Importance (SCIs) based on a set of criteria listed in Annex III. These national SCIs were screened on their contribution to maintaining or re-establishing a natural habitat (Annex I) or species (Annex II). The criteria for this screening are (1) the relative value of the site at national level, (2) the geographical situation of the site in relation to migration routes of species in Annex II and whether or not it belongs to a continuous ecosystem situated on both sides of one or more internal SCI frontiers, (3) total area of the site, (4) the number of natural habitat types (Annex I) and species (Annex II) present on the site and (5) the global ecological value of the site for the biogeographical regions. Sites chosen after this stage become designated SACs within 6 years.

Natura 2000 is the centerpiece of the European Union (EU) nature and biodiversity policy. It is a European wide network of important ecological sites established under the HD (SACs) and BD (SPAs). Member States are fulfilling their obligation under the CBD by contributing to the Natura 2000 network. They are committed to provide an update report on their protected sites to the EU every six years. The aim of the Natura 2000 network is to protect most valuable and vulnerable habitats and species across their natural range and ensure that they are restored to, or maintained at, a favorable conservation status. Natura 2000 fully recognises man as an integral part of nature and stimulates partnerships between nature and man. Many sites in the Natura 2000 Network are valuable precisely because of the way they have been managed up to now. Considering that the majority of Natura 2000 sites are likely to be in private ownership and used for purposes other than nature conservation, it is also essential that future management is sustainable, both ecologically and socioeconomically.

The conservation objectives can be achieved by drawing up a set of statutory, administrative or contractual measures. Management plans reflect these objectives and measures, and reconcile them with sustainable economic development, safety issues and accessibility. They also create opportunities to integrate recurring and routine maintenance activities. Monitoring schemes should be established to discover trends and follow up short and long term evolution, such as morphological dynamics and sediment circulation and redistribution. Applying the principles of adaptive management, objectives and measures can then be revisited where and when necessary.

However, past, present and future activities in SACs can cause deterioration of natural habitats or disturbance of species for which the area has been designated. If appropriate, the negative impact should be brought to an end either by stopping the activity or taking mitigating measures. Being an integral part of the specifications of a plan or project, mitigating measures aim at minimizing or even cancelling the negative effects. Compensatory measures constitute the very last resort. They can consist of (1) recreating a habitat on a new or enlarged site, to be incorporated into Natura 2000, (2) improving a habitat on part of the site or on another Natura 2000 site, proportional to the loss due to the project or (3) in exceptional cases, proposing a new site under the HD. The result has to be operational when the damage is effective on the project site unless it can be proved that this simultaneity is not necessary to ensure the contribution of this site to the Natura 2000 network.

Whilst the designation of Natura 2000 sites in coastal and inshore waters is considered to be fairly advanced, there are still important gaps in the network regarding the offshore marine environment, due to the difficulties in obtaining scientific knowledge on distribution and abundance of species and habitats. In an area so difficult to patrol, the cooperation of all operating interest groups is paramount to the success of conservation measures proposed.

The de facto extension of Natura 2000 to non-EU countries is currently represented by the Emerald network of Areas of Special Conservation Interest (ASCIs, see Bern Convention).

The **Water Framework Directive** (WFD 2000) rationalises and updates existing water legislation by setting common European wide objectives for all surface waters within 1 nautical mile from the coastline (1852 m), namely groundwater, rivers, lakes, transitional and coastal waters. The WFD aims at an integrated and coordinated long-term sustainable water management based on a high level of protection of the aquatic environment. With regard to their water needs, terrestrial ecosystems and wetlands directly depending on the aquatic ecosystem also fall within the scope of the WFD. The main environmental objectives are to achieve and maintain 'good ecological potential (GEP)' for heavily modified water bodies and 'good ecological status (GES)' for all surface waters by 2015. Ecological status is an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters. The ecological status classification is based on biological and physical-chemical monitoring results. Regarding the chemical status, the WFD scope is extended to cover all territorial waters. In terms of quality elements for transitional and coastal waters, the biological quality elements (BQEs) include phytoplankton, macroalgae, angiosperms, benthic invertebrates and fish. Physical-chemical elements comprise pollutants and general conditions, e.g. transparency, oxygenation conditions and nutrients. The normative classification definitions (Annex V) provide a general description on how the critical biological components, such as taxa composition, diversity, abundance… change as response to environmental degradation and pressures. Member States translate these descriptors into practical quality targets and specific quantitative metrics, e.g. various diversity indices, biomass metrics or metrics describing numbers of sensitive versus non-sensitive species in the marine environment. To achieve good quality status, the WFD contains provisions for the coordinated elaboration of River Basin

Management Plans (RBMPs). As part of a RBMP, a monitoring network has to be established to provide a coherent and comprehensive overview of the ecological and chemical status.

The **Integrated Coastal Zone Management** (ICZM 2002) concept was born in 1992 during the Earth Summit of Rio de Janeiro. The policy regarding ICZM is set out in the proceedings of the summit within Agenda 21. Nowadays, ICZM is a dynamic, multidisciplinary and iterative process to promote integrated sustainable management, regarding all aspects of the coastal zone, including geographical and political boundaries. It covers the full cycle of information collection, planning in its broadest sense, decision making, policy areas, managing and monitoring of implementation. ICZM seeks, over the long-term, to balance environmental, economic, social, cultural and recreational objectives, all within the limits set by natural dynamics. The strategic approach emphasizes both the protection of the coastal environment and its integrity and functioning, based on an ecosystem approach, and the threat posed by climate change or unsustainable economic activities and employment options. It will consider local traditional activities and customs that do not present a threat to sensitive natural areas and to the maintenance status of the wild coastal species.

In 2006, the **Biodiversity Action Plan** was drawn up by the European Commission. The Action Plan underlines the importance of biodiversity and ecosystems protection as a prerequisite for sustainable development. For the first time, all relevant economic sectors and policy areas are included in a single strategy document and given responsibility for its implementation. It sets out a comprehensive program of actions and targets which would enable the EU to meet its commitment to halt biodiversity loss by 2010. In light of the expiry of the 2010 biodiversity targets, the new 2050 EU biodiversity vision and 2020 target were agreed upon).

The **Marine Strategy Framework Directive** (MSFD 2008) establishes a framework for the protection, preservation and restoration of marine ecosystems. It promotes the integration of environmental considerations into all relevant policy areas and delivers the environmental pillar of the future EU maritime policy. The ultimate aim of MSFD is to provide diverse and dynamic oceans and seas which are clean, healthy and productive by achieving or maintaining a good environmental status (GEnvS) by the year 2020. GEnvS could be considered as a point, somewhere between an undisturbed (reference) situation and the present situation, assuming that the current situation is not considered to be ‘good’. Annex 1 of the MSFD provides 11 qualitative descriptors of GEnvS: (1) biological diversity, (2) non-indigenous species, (3) commercial fish, (4) food webs, (5) eutrophication, (6) sea floor integrity, (7) hydrography, (8) contaminants, (9) contaminants in food, (10) marine litter, (11) energy, including noise. However, GEnvS can be regarded as an ethical concept which is highly dependent on worldview and existing national and international commitments (Mee et al. 2008). While scientific knowledge can be helpful to describe ecosystem changes, society has to decide to what extent these changes are acceptable. To be operational, quantification of the 11 qualitative descriptors is required. For those purposes, a set of characteristics, criteria and methodological standards for GES should be defined by each Member State. Consistency is needed to allow for comparison between marine (sub)regions. The next step should then include the establishment of environmental targets and monitoring programs. Environmental targets should comprise qualitative or quantitative statements on the desired condition of the different components of, and pressures and impacts on, marine waters.

The geographical scope of the MSFD overlaps with the Natura 2000 sites and WFD in transitional and coastal waters. Although the principal aim of the WFD and the Natura 2000 directives is to protect ecosystems, their objectives, measures and tools are not entirely complementary. Attention should be given to the synergies between them where both apply. The WFD clearly states that if the conservation objectives of the HD are more stringent than the requirements of the WFD then those of the former will apply. The same is also true of the converse. The MSFD, however, does not apply to transitional waters such as estuaries. Its environmental status only applies in the coastal waters insofar as particular aspects are not already addressed through the WFD, e.g. litter, noise, cetaceans. These two more recent directives complement the other nature directives by putting increased emphasis on the role of ecosystems. Integrated WFD, MSFD and Natura 2000 management plans should be established where possible.

Directives regarding environmental assessments

Contrary to most environmental legislation, **environmental impact assessments** do not lay down any measurable environmental standards. They establish a sustainable process of identifying, predicting, evaluating and mitigating the relevant environmental impacts from projects, plans and programs prior to decisions being taken and commitments made at national, regional or local level. The **Environmental Impact Assessment** Directive (EIA 1985) is the general procedure for individual projects while the **Strategic Environmental Assessment** Directive (SEA 2001) is necessary for public plans or programs, without any referral to policies. Bear in mind that national defence, civil emergencies, finance and budget projects, plans and programs are excluded from both process Directives.

For all projects listed in Annex I, an EIA is mandatory. For projects listed in Annex II (e.g. agriculture, energy, food industry), the competent national authorities follow a screening procedure (criteria listed in Annex III), based on a case-by-case examination, nationally set thresholds or criteria. If the screening concludes that an EIA is not needed, the decision is published and the process ends. A SEA is indispensable for plans and programs either requiring an appropriate assessment under the HD or setting the framework for future development consent of projects listed in the EIA Directive, e.g. agriculture, forestry, fisheries, energy, industry, transport, waste or water management, telecommunications, tourism, town and country planning or land use. For themes not included above, Member States have to carry out a screening procedure (criteria listed in Annex II) to determine whether the projects, plans and programs are likely to have significant environmental effects. If there are significant effects, an SEA is needed.

An environmental report of a sufficient standard must be prepared on the detrimental impacts that are likely to result. Environmental authorities and the public should consult this report ensuring objective information and public participation in the decision making process.

Appendices – Chapter 2

Table A: Original species list; Species with one of the following symbols were not taken into account in the analyses: E=epibenthos; H=hyperbenthos, M=meiobenthos, *= in < 0.5 % of the samples.

Species list	Species code		Species list	Species code	
Abra alba	Abraalba		*Caligidae sp*	Calispec	*
Actinaria spec.	Actispec		*Callionymus lyra*	Calllyra	E
Ammothella longipes	Ammolong	*	*Cancer pagurus*	Cancpagu	E
Ammodytes tobianus	Ammotobi	E	*Capitella capitata*	Capicapi	
Ampharetidae species	Amphaspec		*Capitella species*	Capispec	*
Amphipoda species	Amphipspec	*	*Caprellidae species*	Caprspec	*
Amphipholis species	Amphispec	E	*Carcinus maenas*	Carcmaen	E
Angulus fabula	Angufabu		*Cerastoderma edule*	Ceraedul	
Angulus fabula juveniles	Angufabujuv	*	*Cerebratulidae species*	Cerespec	*
Angulus pygmaeus	Angupygm	*	*Chaetozone setosa*	Chaeseto	
Angulus tenuis	Angutenu		*Cirratulidae species*	Cirrspec	
Anthozoa species	Anthspec	E	*Copepoda species*	Copespec	M
Aonides oxycephala	Aonioxyc	*	*Corophium species*	Corospec	
Aonides paucibranchiata	Aonipauc	*	*Crangon crangon*	Crancran	E
Aphelochaeta marioni	Aphemari		*Crangon crangon juveniles*	Crancranjuv	H
Arenicola marina	Arenmari		*Crepidula fornicata*	Crepforn	*
Arrhis phyllonyx	Arrhphyl	*	*Cumacea species*	Cumaspec	*
Asterias rubens	Asterube	E	*Cumopsis goodsir*	Cumogood	
Asterias species juvenielen	Astespecjuv	E	*Cumopsis longipes*	Cumolong	*
Atylus falcatus	Atylfalc		*Decapoda juveniles*	Decajuv	H
Atylus swammerdami	Atylswam		*Decapoda species*	Decaspec	E
Atylus vedlomensis	Atylvedl	*	*Diastylis bradyi*	Diasbrad	
Autolytus prolifera	Autoprol	*	*Diastylis laevis*	Diaslaev	*
Autolytus species	Autospec		*Diastylis lucifera*	Diasluci	*
Bathyporeia elegans	Batheleg		*Diastylis rathkei*	Diasrath	
Bathyporeia guilliamsoniana	Bathguil		*Diastylis rathkei juveniles*	Diasrathjuv	*
Bathyporeia nana	Bathnana	*	*Diastylis rugosa*	Diasrugo	*
Bathyporeia pelagica	Bathpela		*Diastylis species*	Diasspec	*
Bathyporeia pilosa	Bathpilo		*Diogenes pugilator*	Diogpugi	E
Bathyporeia sarsi	Bathsars		*Donax vittatus*	Donavitt	
Bathyporeia species	Bathspec	*	*Donax vittatus juveniles*	Donavittjuv	
Bathyporeia tenuipes	Bathtenu	*	*Echinocardium cordatum*	Echicord	E
Bivalvia juveniles	Bivjuv	*	*Echinocyamus pusillus*	Echipusi	E
Bodotria arenosa	Bodoaren	*	*Enchytraeidae species*	Enchspec	*
Bodotria pulchella	Bodopulc		*Ensis arcuatus*	Ensiarcu	*
Branchiostoma lanceolatum	Branlanc	E	*Ensis directus*	Ensidire	

Table A: Original species list; Species with one of the following symbols were not taken into account in the analyses: E=epibenthos; H=hyperbenthos, M=meiobenthos, *= in < 0.5 % of the samples.

Species list	Species code	
Ensis juveniles	Ensijuv	
Ensis species	Ensispec	
Eteone flava	Eteoflav	
Eteone longa	Eteolong	
Eteone species	Eteospec	*
Eumida bahuniensis	Eumibahu	*
Eumida sanguinea	Eumisang	
Eumida juveniles	Eumijuv	
Eurydice affinis	Euryaffi	
Eurydice pulchra	Eurypulc	
Euspira pulchella	Eusppulc	*
Gammarus species	Gammspec	
Gastrosaccus species	Gastspec	H
Gastrosaccus spinifer	Gastspin	H
Gattyana cirrhosa	Gattcirr	
Glycera species	Glycspec	
Goniada maculata	Gonimacu	*
Harmothoe glabra	Harmglab	*
Harmothoe nodosa	Harmnodo	*
Harmothoe species	Harmspec	*
Haustorius arenarius	Hausaren	
Hesionides arenaria	Hesiaren	*
Hesionura elongata	Hesielon	*
Hesioninae species	Hesiosp	*
Heteromastus filiformis	Hetefili	
Holothuroidea species	Holospec	E
Idotea linearis	Idotline	H
Iphinoe trispinosa	Iphitris	
Jassa falcata	Jassfalc	
Jassa herdmani	Jassherd	
Jassa species	Jassspec	
Lagis koreni	Lagikore	
Lanice conchilega	Laniconc	
Leptomysis gracilis	Leptgrac	*
Leucothoe incisa	Leucinci	*
Leucothoe lilljeborgi	Leuclill	*
Leucothoe spinicarpa	Leucspin	*
Liocarcinus arcuatus	Liocarcu	E
Liocarcinus holsatus	Liochols	E
Liocarcinus marmoreus	Liocmarm	E
Liocarcinus vernalis	Liocvern	E
Macoma balthica	Macobalt	
Macoma balthica juveniles	Macobaltjuv	
Mactra stultorum	Mactstul	*
Maerella tenuimana	Maertenu	*
Magelona species	Magespec	
Malacoceros species	Malaspec	*
Malmgreniella castanea	Malmcast	*
Malmgreniella juveniles	Malmjuv	
Megaluropus agilis	Megaagil	*
Melita species	Melispec	*
Mesopodopsis slabberi	Mesoslab	H
Microprotopus maculatus	Micrmacu	
Microphtalmus similis	Micrsimi	*
Modiolula phaseolina	Modiphas	*
Mya arenaria	Mya aren	*
Kurtiella bidentata	Mysebide	
Mysida species	Mysispec	H
Mytilus edulis	Mytiedul	
Nassarius reticulatus	Nassreti	*
Nematoda species	Nemaspec	M
Nemertea species	Nemespec	
Neomysis integer	Neominte	H
Nephtys assimilis	Nephassi	
Nephtys caeca	Nephcaec	
Nephtys cirrosa	Nephcirr	
Nephtys hombergii	Nephhomb	
Nephtys juveniles	Nephjuv	
Nephtys longosetosa	Nephlong	*
Nephtys hombergii	Nephomb	
Nephtys species	Nephspec	
Nereis longissima	Nerelong	

Table A: Original species list; Species with one of the following symbols were not taken into account in the analyses: E=epibenthos; H=hyperbenthos, M=meiobenthos, *= in < 0.5 % of the samples.

Species list	Species code	
Nereis juveniles	Nerejuv	*
Nereis virens	Nerevire	*
Notomastus latericeus	Notolate	
Oligochaeta species	Oligspec	
Ophelia borealis	Ophelima	*
Ophelia rathkei	Opherath	
Ophiura albida	Ophialbi	E
Ophiura juveniles	Ophijuv	E
Ophiura ophiura	Ophiophi	E
Ophiura species	Ophispec	E
Orchestia cavimana	Orchcavi	*
Orchomenella nana	Orchnana	*
Owenia fusiformis	Owenfusi	
Pagurus bernhardus	Pagubern	E
Paraonis fulgens	Parafulg	
Pariambus typicus	Paritypi	
Perioculodes longimanus	Perilong	*
Pholoe inornata	Pholinor	*
Pholoe minuta	Pholminu	
Photis reinhardi	Photrein	*
Phtisica marina	Phtimari	*
Phyllodoce species	Phylspec	
Pinnotheres pisum	Pinnpisu	*
Podarkeopsis helgolandica	Podahelg	*
Poecilochaetus serpens	Poecserp	
Polydora species	Polyspec	
Polynoidae species	Polynspec	*
Pomatoschistus lozanoi	Pomaloza	E
Pomatoschistus minutus	Pomaminu	E
Pontocrates altamarinus	Pontalta	
Pontocrates arenarius	Pontaren	
Portumnus latipes	Portlati	E
Psammodrilus balanoglossoides	Psambala	
Pseudocuma gilsoni	Pseugils	*
Pseudocuma longicorne	Pseulong	
Pygospio elegans	Pygoeleg	
Sagitta species	Sagispec	
Schistomysis kervillei	Schikerv	H
Schistomysis ornata	Schiorna	H
Schistomysis spiritus	Schispir	H
Scoloplos armiger	Scolarmi	
Scolelepis bonnieri	Scolbonn	*
Scolelepis squamata	Scolsqua	
Sigalion mathildae	Sigamath	
Solea solea	Solesole	E
Spiophanes bombyx	Spiobomb	
Spionidae species	Spionsp	
Spio species	Spiospec	
Spisula subtruncata	Spissubt	
Stenothoe marina	Stenmari	*
Stenothoe monoculoides	Stenmono	*
Sthenelais boa	Stheboa	*
Streblospio benedicti	Strebene	*
Syllis species	Syllspec	*
Synchelidium haplocheles	Synchapl	*
Talitrus saltator	Talisalt	
Tellimya ferruginosa	Tellferr	*
Thia scutellata	Thiascut	*
Turbellaria species	Turbspec	M
Urothoe brevicornis	Urotbrev	
Urothoe poseidonis	Urotpose	
Urothoe pulchella	Urotpulc	
Urothoe species	Urotspec	*

Table B1: Summary of a permutative linear model based on an Euclidean distance matrix, partitioning multivariate variation in abiotic structure in the intertidal and shallow subtidal zone zone

Main Model	**Intertidal zone**					**Shallow subtidal zone**				
	df	**SS**	**Pseudo-F**	**R^2**	**P(perm)**	**df**	**SS**	**Pseudo-F**	**R^2**	**P(perm)**
Beach	14	5.678	129.389	0.568	**0.001**	8	4.524	125.966	0.454	**0.001**
Year	6	2.188	116.337	0.219	**0.001**	6	2.774	102.987	0.278	**0.001**
Season	2	0.055	8.731	0.005	**0.001**	2	0.132	14.655	0.013	**0.001**
Residuals	663	2.078		0.208		565	2.536		0.255	
Total	685	9.998		1		581	9.965		1	

Table B2: Summary of a permutative linear model based on an Bray-Curtis distance matrix, partitioning multivariate variation in macrobenthic community structure in the intertidal and shallow subtidal zone zone

Main Model	**Intertidal zone**					**Shallow subtidal zone**				
	df	**SS**	**Pseudo-F**	**R^2**	**P(perm)**	**df**	**SS**	**Pseudo-F**	**R^2**	**P(perm)**
Beach	14	5.678	129.389	0.568	**0.001**	8	31.067	15.618	0.134	**0.003**
Year	6	2.188	116.337	0.219	**0.001**	6	27.396	18.363	0.118	**0.001**
Elevation	279	120.627	5.913	0.444	**0.001**	370	116.927	1.271	0.505	**0.023**
Median grain size	321	95.482	4.068	0.351	**0.001**	181	52.393	1.164	0.226	0.149
Residuals	65	4.753		0.017		16	3.978		0.017	
Total	685	271.737		1		581	231.762		1	

Table C: Overview of abiotic and biotic mean values per beach, over all sampled years

INTERTIDAL ZONE	**Codes**	**Beach elevation (m versus MLW)**	**Median grain size (µm)**	**Silt fraction (%)**	**Total organic matter (%)**	**Total organic carbon (%)**	**Carbonate content (%)**	**Species richness (number of species)**	**Abundance (number of individuals.m^{-2})**	**Biomass (g AFDW.m^{-2})**
De Panne	1	2.85	198.51	0.001	0.61		0.58	4.96	455.70	0.00
Schipgatduinen	2	2.70	242.96	0.001	0.74		0.47	4.64	63.21	0.00
Koksijde-Oostduinkerke	3	2.55	199.45	0.003	0.15	0.96	12.48	4.07	111.13	0.59
Nieuwpoort	4	2.15	208.64	0.000	0.54	0.77	11.18	4.76	112.77	0.53
Middelkerke	5	2.35	256.18	0.008	0.28	0.97	12.22	3.86	52.06	0.40
Raversijde	6	2.57	210.89	0.000	0.57		0.42	7.20	196.10	0.00
Mariakerke	7	2.61	344.64	0.000	0.62	0.08	13.97	2.53	204.08	0.62
Oostende-Center	8	2.07	278.70	0.009	0.27	1.15	17.54	3.10	89.93	0.18
Oostende-East	10	2.25	241.54	0.009	0.45	0.57	11.91	4.96	76.32	0.33
Bredene	11	2.17	273.02	0.000	0.48	0.09	8.97	3.90	52.02	0.38
Wenduine	12	1.93	258.06	0.000	0.12	0.55	8.91	4.32	36.93	0.81
Blankenberge	13	2.32	249.42	0.000	0.55	0.06	5.59	3.28	30.93	0.14
Fonteintjes	14	2.70	240.65	0.000	0.53		0.20	4.95	79.21	0.00
Heist	15	2.93	255.42	0.395	0.56		0.33	6.41	191.33	0.00
Zwinduinen en Polders	16	2.32	321.92	0.000	0.70		1.93	5.75	67.21	0.00

SHALLOW SUBTIDAL ZONE	**Codes**	**Beach elevation (m versus MLW)**	**Median grain size (µm)**	**Silt fraction (%)**	**Total organic matter (%)**	**Total organic carbon (%)**	**Carbonate content (%)**	**Species richness (number of species)**	**Abundance (number of individuals.m^{-2})**	**Biomass (g AFDW.m^{-2})**
Koksijde-Oostduinkerke	3	-5.75	190.58	2.38	0.24	0.84	10.57	7.53	45.83	5.97
Nieuwpoort	4	-3.09	184.52	3.60	1.79	0.63	10.20	10.58	222.95	12.58
Middelkerke	5	-2.61	174.73	1.73	0.62	1.01	11.30	6.40	37.47	3.63
Mariakerke	7	-3.69	191.22	3.96	1.40	0.18	9.42	10.13	166.05	45.20
Oostende-Center	8	-4.90	179.38	4.25	0.56	0.70	12.35	7.44	76.82	2.00
Oostende-Fairway	9	-6.46	181.27	11.11	0.34	1.06	11.16	5.67	32.98	0.63
Oostende-East	10	-2.28	176.43	4.23	1.57	0.65	12.94	8.03	86.94	5.63
Bredene	11	-2.24	218.58	0.30	1.04	0.12	7.18	8.18	84.08	10.41
Wenduine	12	-6.68	136.30	32.76	0.98	1.68	16.19	4.73	67.11	8.39

Table D: Overview of abiotic and biotic mean values per year, over all sampled beaches

INTERTIDAL ZONE	**Beach elevation (m versus MLW)**	**Median grain size (µm)**	**Silt fraction (%)**	**Total organic matter (%)**	**Total organic carbon (%)**	**Carbonate content (%)**	**Species richness (number of species)**	**Abundance (number of individuals.m^{-2})**	**Biomass (g AFDW.m^{-2})**
1997	2.72	242.34	0.065		0.63	0.62	5.59	193.07	0.00
2002	2.50	248.85	0.016			0.48	3.34	93.60	0.00
2004		222.33	0.011	0.17	12.08	0.11	5.27	49.14	0.57
2006	2.88	277.39	0.004	1.69	13.75		3.27	94.06	0.40
2008	2.65	206.35	0.000	1.43	11.90	0.28	3.62	178.38	0.53
2009	2.34	199.42	0.000	1.14	11.31	0.63	4.14	82.41	0.35
2010	2.25	254.89	0.000	0.15	10.25	0.56	4.95	82.83	0.50
2011	2.23	256.33	0.000	0.10	10.79	0.56	3.90	83.22	0.42

SHALLOW SUBTIDAL ZONE	**Beach elevation (m versus MLW)**	**Median grain size (µm)**	**Silt fraction (%)**	**Total organic matter (%)**	**Total organic carbon (%)**	**Carbonate content (%)**	**Species richness (number of species)**	**Abundance (number of individuals.m^{-2})**	**Biomass (g AFDW.m^{-2})**
1997									
2002	-3.65	181.09	6.42	0.84			6.95	33.91	0.00
2004		172.45	2.37	0.34	0.32	12.58	7.34	73.47	1.76
2006	-4.02	171.03	12.32		1.62	12.64	6.54	48.51	6.86
2008	-3.46	186.48	0.00	0.38	1.32	11.04	7.36	67.37	3.71
2009	-2.38	181.92	1.13	1.49	0.64	9.56	8.68	312.04	6.71
2010	-2.31	205.79	1.16	1.53	0.21	9.12	9.29	197.29	20.40
2011	-2.78	186.49	6.73	2.18	0.28	10.62	10.41	99.83	11.37

Appendices – Chapter 3

Table A: Original species list; Species with one of the following symbols were not taken into account in the analyses: E=epibenthos; H=hyperbenthos, M=meiobenthos, *= in < 0.5 % of the samples.

Species list	Species code	
Abra alba	Abraalba	
Actinaria species	Actispec	*
Ammodytes tobianus	Ammotobi	E
Amphipoda species	Amphispec	*
Ampharetidae species	Amphaspec	
Angulus fabula	Angufabu	
Angulus fabula juveniles	Angufabujuv	
Angulus pygmaeus	Angupygm	
Angulus tenuis	Angutenu	
Aonides oxycephala	Aonioxyc	*
Arenicola marina	Arenmari	
Asterias rubens	Asterube	E
Atylus falcatus	Atylfalc	
Atylus species	Atylspec	*
Atylus swammerdami	Atylswam	
Autolytus species	Autospec	
Bathyporeia elegans	Batheleg	
Bathyporeia guilliamsoniana	Bathguil	*
Bathyporeia pelagica	Bathpela	
Bathyporeia pilosa	Bathpilo	
Bathyporeia sarsi	Bathsars	
Bathyporeia species	Bathspec	*
Bodotria arenosa	Bodoaren	*
Bodotria pulchella	Bodopulc	
Buccinum undatum	Buccunda	E
Caligidae species	Calispec	M
Callionymus lyra	Calllyra	E
Capitella capitata	Capicapi	
Carcinus maenas	Carcmaen	E
Cirratulidae species	Cirrspe	
Copepoda species	Copespec	M
Corophium species	Corospec	
Corystes cassivelaunus	Corycass	E
Crangon crangon	Crancran	E
Crangon crangon juveniles	Crancranjuv	H
Cumacea species	Cumaspec	*
Cumopsis goodsir	Cumogood	
Cumopsis longipes	Cumolong	*
Decapoda juveniles	Decajuv	H
Diastylis bradyi	Diasbrad	
Diastylis lucifera	Diasluci	*
Diastylis rathkei	Diasrath	
Diogenes pugilator	Diogpugi	E
Donax vittatus	Donavitt	
Donax vittatus juveniles	Donavittjuv	
Echinocyamus pusillus	Echipusi	E
Enchytraeidae species	Enchspec	
Ensis arcuatus	Ensiarcu	*
Ensis directus	Ensidire	*
Ensis juveniles	Ensijuv	
Ensis species	Ensispec	
Eteone flava	Eteoflav	
Eteone longa	Eteolong	
Eteone species	Eteospec	*
Eumida sanguinea	Eumisang	
Eurydice affinis	Euryaffi	
Eurydice pulchra	Eurypulc	
Gammarus oceanicus	Gammocea	
Gastrossacus species	Gastspec	H
Gattyana cirrosa	Gattcirr	*
Glycera species	Glycspec	
Harmothoe nodosa	Harmnodo	
Harmothoe species	Harmspec	*
Haustorius arenarius	Hausaren	
Heteromastus filiformis	Hetefili	*
Heteromysis formosa	Heteform	H
Hydrobia ulvae	Hydrulva	E
Idotea linearis	Idotline	H
Iphinoe trispinosa	Iphitris	*
Jassa herdmani	Jassherd	
Jassa falcata	Jassfalc	
Jassa species	Jassspec	
Lanice conchilega	Laniconc	
Leucothoe incisa	Leucinci	
Liocarcinus holsatus	Liochols	E
Liocarcinus marmoreus	Liocmarm	E
Liocarcinus vernalis	Liocvern	E
Macoma balthica	Macobalt	
Macoma balthica juveniles	Macobaltjuv	
Maerella tenuimana	Maertenu	
Magelona species	Magespec	
Malmgreniella ljungmani	Malmljun	*

Table A: Original species list; Species with one of the following symbols were not taken into account in the analyses: E=epibenthos; H=hyperbenthos, M=meiobenthos, *= in < 0.5 % of the samples.

Species list	Species code	
Melita obtusata	Meliobtu	
Melita species	Melispec	*
Mesopodopsis slabberi	Mesoslab	H
Microprotopus maculatus	Micrmacu	*
Modiolula phaseolina	Modiphas	*
Kurtiella bidentata	Mysebide	
Mysida species	Mysispec	H
Mytilus edulis	Mytiedul	*
Nemertea species	Nemespec	
Nephtys assimilis	Nephassi	
Nephtys caeca	Nephcaec	
Nephtys cirrosa	Nephcirr	
Nephtys hombergii	Nephhomb	
Nephtys juveniles	Nephjuv	
Nephtys longosetosa	Nephlong	*
Nephtys species	Nephspec	*
Nereis diversicolor	Neredive	*
Nereis longissima	Nerelong	
Nereis species	Nerespec	*
Notomastus latericeus	Notolate	
Oligochaeta species	Oligspec	
Ophiura albida	Ophialbi	E
Ophiura juveniles	Ophijuv	E
Ophiura ophiura	Ophiophi	E
Orchomenella nana	Orchnana	
Owenia fusiformis	Owenfusi	
Pagurus bernhardus	Pagubernh	E
Paguridae species	Paguspec	E
Pariambus typicus	Paritypi	
Pectinaria koreni	Pectkore	
Perioculodes longimanus	Perilong	
Pholoe minuta	Pholminu	
Phyllodoce species	Phylspec	
Pisidia longicornis	Pisilong	*
Pisces species	Piscspec	E
Pleuronectes platessa	Pleuplat	*
Poecilochaetus serpens	Poecserp	*
Polydora species	Polyspec	
Pomatoschistus minutus	Pomaminu	E
Pomatoschistus lozanoi	Pomaloza	E
Pontocrates altamarinus	Pontalta	
Pontocrates arenarius	Pontaren	
Pontocrates species	Pontspec	*
Portumnus latipes	Portlati	E
Praunus neglectus	Praunegl	
Pseudocuma longicornis	Pseulong	
Pygospio elegans	Pygoeleg	
Pygospio elegans	Pygoeleg	
Schistomysis kervillei	Schikerv	H
Schistomysis spiritus	Schispir	H
Scoloplos armiger	Scolarmi	
Scolelepis squamata	Scolsqua	
Sigalion mathildae	Sigamath	
Solea solea	Solesole	E
Spiophanes bombyx	Spiobomb	
Spio species	Spiospec	
Spisula species	Spisspec	
Spisula subtruncata	Spissubt	
Talitrus saltator	Talisalt	
Urothoe brevicornis	Urotbrev	
Urothoe poseidonis	Urotpose	
Urothoe pulchella	Urotpulc	
Urothoe species	Urotspec	*

Table B1: Overview of abiotic and biotic mean values of the intertidal zone of the impact site per year and season (spring=grey; autumn=blank; nourishment period=red). The minimum and maximum values for abiotic and biotic factors in the intertidal zone of the Belgian beaches are indicated as well.

Impact site	INTERTIDAL				
	minimum	maximum	2008	2009	
Beach elevation (m versus MLW)	0.00	5.41	2.49 ± 0.24	2.28 ± 0.22	2.16 ± 0.12
Median grain size (µm)	175.00	464.00	215.89 ± 3.63	212.98 ± 3.94	236.84 ± 8.53
Silt fraction (%)	0.00	2.25	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Total organic matter (%)	0.00	1.84	0.24 ± 0.01	0.56 ± 0.02	0.51 ± 0.03
Total organic carbon (%)	0.00	19.00	1.50 ± 0.03	1.39 ± 0.06	0.19 ± 0.04
Carbonate content (%)	0.00	36.68	12.49± 0.29	11.62 ± 0.53	10.23 ± 1.12
Species richness (number of species)	0.00	19.00	3.44 ± 0.30	3.03 ± 0.28	2.96 ± 0.34
Abundance (number of individuals.m^{-2})	0.00	3988.75	65.89 ± 18.58	46.32 ± 12.57	60.46 ± 19.27
Biomass (g AFDW.m^{-2})	0.00	6.95	0.21 ± 0.04	0.33 ± 0.10	0.17 ± 0.06

Impact site	INTERTIDAL					
	2010		2011		2012	
Beach elevation (m versus MLW)	2.40 ± 0.32	2.08 ± 0.17	2.14 ± 0.28	1.91 ± 0.20	2.58 ± 0.31	1.82 ± 0.06
Median grain size (µm)	280.23 ± 8.94	168.12 ± 4.13	262.80 ± 6.84	252.39 ± 9.54	267.89 ± 5.69	247.86 ± 2.87
Silt fraction (%)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Total organic matter (%)	0.79 ± 0.04	0.83 ± 0.26	0.56 ± 0.03	0.43 ± 0.04	0.06 ± 0.10	0.88 ± 0.07
Total organic carbon (%)	0.12 ± 0.01	0.09 ± 0.05	0.14 ± 0.01	0.09 ± 0.01	0.085 ± 0.005	0.072 ± 0.002
Carbonate content (%)	9.87 ± 1.91	5.78 ± 0.28	6.62 ± 0.65	8.52 ± 1.11	6.94 ± 0.54	8.70 ± 0.25
Species richness (number of species)	2.47 ± 0.56	5.53 ± 0.50	4.27 ± 0.33	3.27 ± 0.34	3.67 ± 0.51	4.13 ± 0.12
Abundance (number of individuals.m^{-2})	36.50 ± 19.56	121.92 ± 33.68	105.59 ± 31.76	46.89 ± 13.54	39.52 ± 11.89	62.26 ± 3.15
Biomass (g AFDW.m^{-2})	0.08 ± 0.02	0.67 ± 0.12	0.46 ± 0.11	0.21 ± 0.07	0.34 ± 0.08	1.10 ± 0.10

Table B2: Overview of abiotic and biotic mean values of the shallow subtidal zone of the impact site per year and season (spring=grey; autumn=blank; nourishment period=red). The minimum and maximum values for abiotic and biotic factors in the shallow subtidal zone of the Belgian beaches are indicated as well.

Impact site	**SHALLOW SUBTIDAL**				
	minimum	maximum	2008	2009	
Beach elevation (m versus MLW)	-10.00	2.67	-2.78 ± 0.29	-1.59 ± 0.38	-1.48 ± 0.47
Median grain size (µm)	17.45	319.73	182.30 ± 2.35	188.42 ± 2.69	195.89 ± 6.62
Silt fraction (%)	0.00	89.30	1.01 ± 0.91	0.99 ± 0.49	5.51 ± 4.02
Total organic matter (%)	0.00	11.75	0.37 ± 0.05	1.45 ± 0.10	1.44 ± 0.10
Total organic carbon (%)	0.00	4.85	1.54 ± 0.03	1.31 ± 0.05	0.21 ± 0.03
Carbonate content (%)	0.00	36.10	12.84 ± 0.28	10.88 ± 0.44	11.73 ± 0.51
Species richness (number of species)	0.00	28.00	7.56 ± 0.67	5.72 ± 0.44	9.20 ± 1.13
Abundance (number of individuals.m^{-2})	0.00	1949.32	64.69 ± 10.40	50.71 ± 5.22	275.80 ± 61.06
Biomass (g AFDW.m^{-2})	0.00	246.14	2.66 ± 0.47	2.91 ± 0.82	4.40 ± 0.76

Impact site	**SHALLOW SUBTIDAL**					
	2010		2011		2012	
Beach elevation (m versus MLW)	-3.05 ± 0.43	-2.78 ± 0.48	-3.14 ± 0.43	-2.57 ± 0.36	-2.98 ± 0.43	-297 ± 0.10
Median grain size (µm)	158.35 ± 12.17	190.12 ± 2.20	177.04 ±2.36	168.94 ± 5.46	172.98 ± 1.39	174.73 ± 0.35
Silt fraction (%)	10.43 ± 5.27	5.43 ± 1.92	3.65 ± 1.47	5.07 ± 2.82	3.96 ± 1.40	2.26 ± 0.24
Total organic matter (%)	2.77 ± 0.57	2.16 ± 0.31	2.22 ± 0.31	1.64 ± 0.29	0.02 ± 0.01	1.93 ± 0.09
Total organic carbon (%)	0.51 ± 0.16	0.29 ± 0.08	0.35 ± 0.06	0.23 ± 0.05	0.22 ± 0.03	0.191 ± 0.004
Carbonate content (%)	11.62 ± 1.32	12.65 ± 1.05	12.56 ± 0.78	11.07 ± 0.57	11.60 ± 0.57	10.32 ± 0.10
Species richness (number of species)	8.53 ± 1.29	13.33 ± 1.26	10.07 ± 0.85	16.60 ± 1.32	12.80 ± 0.91	16.20 ± 0.39
Abundance (number of individuals.m^{-2})	54.94 ± 12.34	839.54 ± 201.25	298.85 ± 118.96	218.60 ± 42.55	144.58 ± 24.58	884.70 ± 37.65
Biomass (g AFDW.m^{-2})	4.85 ± 1.15	85.23 ± 16.95	14.88 ± 7.65	4.98 ± 1.64	29.47 ± 9.34	74.23 ± 2.57

Table C1: Overview of abiotic and biotic mean values of the intertidal zone of the control site per year and season (spring=grey; autumn=blank; nourishment period=red). The minimum and maximum values for abiotic and biotic factors in the intertidal zone of the Belgian beaches are indicated as well.

Control site	**INTERTIDAL**				
	minimum	maximum	2008	2009	
Beach elevation (m versus MLW)	0.00	5.41	2.65 ± 0.22	1.89 ± 0.22	1.99 ± 0.14
Median grain size (µm)	175.00	464.00	206.35 ± 3.55	197.53 ± 3.02	196.24 ± 2.83
Silt fraction (%)	0.00	2.25	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Total organic matter (%)	0.00	1.84	0.28 ± 0.01	0.65 ± 0.03	0.62 ± 0.02
Total organic carbon (%)	0.00	19.00	1.43 ± 0.05	1.34 ± 0.06	1.03 ± 0.11
Carbonate content (%)	0.00	36.68	11.90 ± 0.39	11.13 ± 0.52	12.21 ± 1.03
Species richness (number of species)	0.00	19.00	3.62 ± 0.36	4.67 ± 0.36	3.77 ± 0.36
Abundance (number of individuals.m^{-2})	0.00	3988.75	178.38 ± 51.61	67.46 ± 12.30	92.76 ± 21.28
Biomass (g AFDW.m^{-2})	0.00	6.95	0.53 ± 0.14	0.59 ± 0.10	0.19 ± 0.06

Control site	**INTERTIDAL**					
	2010		2011		2012	
Beach elevation (m versus MLW)	2.49 ± 0.26	2.04 ± 0.20	2.12 ± 0.30	2.61 ± 0.23	2.22 ± 0.36	2.17 ± 0.07
Median grain size (µm)	201.75 ± 5.32	259.69 ± 4.35	202.29 ± 5.63	206.81 ± 8.25	208.63 ± 6.97	201.97 ± 1.76
Silt fraction (%)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Total organic matter (%)	0.70 ± 0.02	0.52 ± 0.03	0.58 ± 0.04	0.53 ± 0.03	0.03 ± 0.67	0.02 ± 0.02
Total organic carbon (%)	0.27 ± 0.11	0.15 ± 0.05	0.17 ± 0.01	0.06 ± 0.01	0.081 ± 0.004	0.096 ± 0.003
Carbonate content (%)	10.36 ± 0.83	11.69 ± 1.32	11.22 ± 1.01	9.77 ± 0.59	10.82 ± 1.16	11.74 ± 0.59
Species richness (number of species)	5.40 ± 0.32	6.73 ± 0.34	6.00 ± 0.69	4.73 ± 0.47	4.73 ± 0.57	5.00 ± 0.14
Abundance (number of individuals.m^{-2})	54.24 ± 10.68	133.18 ± 49.23	136.98 ± 52.83	101.99 ± 33.75	92.17 ± 35.74	76.63 ± 7.81
Biomass (g AFDW.m^{-2})	0.67 ± 0.18	0.51 ± 0.07	0.71 ± 0.15	0.79 ± 0.16	0.54 ± 0.06	0.86 ± 0.08

Table C2: Overview of abiotic and biotic mean values of the shallow subtidal zone of the control site per year and season (spring=grey; autumn=blank; nourishment period=red). The minimum and maximum values for abiotic and biotic factors in the shallow subtidal zone of the Belgian beaches are indicated as well

Control site	**SHALLOW SUBTIDAL**				
	minimum	maximum	2008	2009	
Beach elevation (m versus MLW)	-10.00	2.67	-2.63 ± 0.38	-2.43 ± 0.53	-2.33 ± 0.55
Median grain size (µm)	17.45	319.73	186.48 ± 2.01	182.33 ± 2.38	181.50 ± 2.03
Silt fraction (%)	0.00	89.30	0.00 ± 0.00	1.51 ± 1.44	0.75 ± 0.42
Total organic matter (%)	0.00	11.75	0.38 ± 0.04	1.37 ± 0.07	1.61 ± 0.25
Total organic carbon (%)	0.00	4.85	1.32 ± 0.01	1.10 ± 0.05	0.18 ± 0.02
Carbonate content (%)	0.00	36.10	11.04 ± 0.12	9.19 ± 0.39	9.92 ± 0.28
Species richness (number of species)	0.00	28.00	7.36 ± 0.53	6.88 ± 0.45	10.48 ± 0.91
Abundance (number of individuals.m^{-2})	0.00	1949.32	67.37 ± 9.70	63.59 ± 9.34	560.49 ± 90.83
Biomass (g AFDW.m^{-2})	0.00	246.14	3.71 ± 0.73	7.34 ± 3.01	6.08 ± 0.87

Control site	**SHALLOW SUBTIDAL**					
	2010		2011		2012	
Beach elevation (m versus MLW)	-3.49 ± 0.48	-3.57 ± 0.41	-3.75 ± 0.39	-3.44 ± 0.38	-3.51 ± 0.39	-3.64 ± 0.11
Median grain size (µm)	182.32 ± 2.57	244.79 ± 4.31	137.51 ± 14.47	178.84 ± 4.30	184.06 ± 2.81	178.92 ± 0.78
Silt fraction (%)	0.67 ± 0.47	2.18 ± 0.86	22.93 ± 6.80	2.85 ± 2.26	0.63 ± 0.63	0.93 ± 0.14
Total organic matter (%)	2.10 ± 0.22	1.59 ± 0.12	4.96 ± 0.94	1.84 ± 0.35	0.013 ± 0.001	1.62 ± 0.04
Total organic carbon (%)	0.29 ± 0.07	0.18 ± 0.03	0.74 ± 0.15	0.08 ± 0.05	0.25 ± 0.05	0.211 ± 0.005
Carbonate content (%)	8.75 ± 0.52	8.93 ± 0.30	14.74 ± 1.19	9.11 ± 0.37	8.91 ± 0.30	9.50 ± 0.11
Species richness (number of species)	8.60 ± 0.86	15.20 ± 1.21	12.60 ± 0.88	17.60 ± 1.31	14.53 ± 1.37	19.13 ± 0.43
Abundance (number of individuals.m^{-2})	47.40 ± 7.64	516.61 ± 100.29	116.03 ± 19.86	174.07 ± 17.82	159.07 ± 16.66	357.29 ± 12.25
Biomass (g AFDW.m^{-2})	6.57 ± 1.18	59.56 ± 15.24	3.48 ± 6.35	15.08 ± 4.24	7.86 ± 1.33	54.18 ± 2.33

Table D: Overview of the Wilcoxon-values between impact and control site for all the abiotic and biotic factors per season, in the intertidal and shallow subtidal zone

INTERTIDAL ZONE	2008	2009		2010		2011		2012	
Beach elevation (m versus MLW)	0.445	0.577	0.191	0.792	0.648	0.903	**0.019**	0.395	0.272
Median grain size (µm)	0.055	**0.025**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.001**
Silt fraction (%)	NA	NA	NA	NA	NA	NA	NA	NA	NA
Total organic matter (%)	**0.001**	**0.049**	**0.001**	0.525	0.056	0.372	0.081	**0.019**	0.836
Total organic carbon (%)	**0.031**	0.141	**0.000**	0.946	0.073	0.053	0.126	0.628	**0.019**
Carbonate content (%)	**0.031**	0.141	0.080	**0.002**	**0.000**	**0.000**	**0.037**	**0.002**	0.202
Species richness (number of species)	0.766	**0.015**	0.114	**0.003**	0.102	0.051	**0.026**	0.480	0.215
Abundance (number of individuals.m^{-2})	0.090	0.376	**0.003**	**0.015**	0.494	1.000	0.054	0.238	1.000
Biomass (g AFDW.m^{-2})	**0.027**	**0.033**	**0.040**	**0.000**	0.619	0.267	0.003	0.063	0.436

SHALLOW SUBTIDAL ZONE	2008	2009		2010		2011		2012	
Beach elevation (m versus MLW)	0.923	0.265	0.288	0.367	0.217	0.310	0.068	0.389	0.202
Median grain size (µm)	0.171	0.271	0.378	0.126	**0.000**	**0.016**	0.389	**0.021**	0.285
Silt fraction (%)	0.162	0.655	0.687	0.162	0.233	**0.008**	0.655	**0.044**	0.338
Total organic matter (%)	0.105	0.677	0.687	0.806	0.290	**0.021**	0.384	0.322	0.648
Total organic carbon (%)	**0.000**	**0.004**	0.316	0.388	0.593	0.110	**0.001**	1.000	0.533
Carbonate content (%)	**0.000**	**0.004**	**0.005**	0.059	**0.001**	0.141	**0.003**	**0.000**	0.141
Species richness (number of species)	0.784	0.082	0.836	0.983	0.095	0.058	0.505	0.317	0.219
Abundance (number of individuals.m^{-2})	0.467	0.244	**0.023**	0.967	0.486	0.683	0.713	0.351	**0.004**
Biomass (g AFDW.m^{-2})	0.397	**0.043**	0.277	0.202	0.285	0.539	**0.045**	0.624	0.126

Table E: Multiple regression model

Residual standard error on 626 DF	23.23
Multiple R²	0.6543
Adjusted R²	0.6372
F-statistic on 31 and 626 DF	38.22
p-value	< 2.2e-16

Main Model	**Estimate**	**SE**	**t value**	**Pr(>|t|)**
(Intercept)	189.5519	2.4124	78.575	**< 2e-16**
Elevation	1.9394	0.8051	2.409	**0.01629**
Time t0	7.1229	4.0755	1.748	0.081
Time t1	3.897	4.9206	0.792	0.42868
Time t2	64.4068	4.9928	12.9	**< 2e-16**
Time t3	-10.3329	4.9959	-2.068	**0.03902**
Time t4	4.9727	4.9086	1.013	0.31142
Time t5	9.4989	4.9543	1.917	0.05565
Time t6	3.4485	4.9775	0.693	0.48868
Treatment impact	15.3546	3.3109	4.638	**4.29E-06**
Elevation*Time t0	1.9005	1.3372	1.421	0.15575
Elevation*Time t1	0.8661	1.5084	0.574	0.56605
Elevation*Time t2	0.3071	1.6059	0.191	0.8484
Elevation*Time t3	9.5229	1.5485	6.15	**1.38E-09**
Elevation*Time t4	2.1609	1.5327	1.41	0.15907
Elevation*Time t5	2.2275	1.5523	1.435	0.15179
Elevation*Time t6	1.5347	1.565	0.981	0.32715
Elevation*Treatment impact	6.0904	1.2027	5.064	**5.41E-07**
Treatment impact*Time t0	-12.3841	5.7053	-2.171	**0.03034**
Treatment impact*Time t1	16.8907	7.08	2.386	**0.01734**
Treatment impact*Time t2	-91.3098	6.9526	-13.133	**< 2e-16**
Treatment impact*Time t3	32.3702	6.9705	4.644	**4.17E-06**
Treatment impact*Time t4	6.651	6.8947	0.965	0.33509
Treatment impact*Time t5	9.8314	6.9675	1.411	0.15873
Treatment impact*Time t6	11.2989	6.9794	1.619	0.10598
Elevation*Treatment impact*Time t0	-3.7051	1.9643	-1.886	0.05972
Elevation*Treatment impact*Time t1	11.8099	2.2811	5.177	**3.04E-07**
Elevation*Treatment impact*Time t2	-11.5074	2.3882	-4.818	**1.82E-06**
Elevation*Treatment impact*Time t3	-3.4478	2.2877	-1.507	0.13229
Elevation*Treatment impact*Time t4	7.4492	2.4568	3.032	**0.00253**
Elevation*Treatment impact*Time t5	4.8311	2.2836	2.116	**0.03478**
Elevation*Treatment impact*Time t6	4.9633	2.3879	2.079	**0.03807**

Appendices – Chapter 5

Appendix A: Input data, sampling strategy and model validation

Input data

The research of the Belgian beach ecosystem started about a decade ago (1997 – present). The simulation model is based upon quantitative information on the littoral macrobenthos community along the Belgian coastline provided by two major research projects, financed by different branches of the Flemish government. Within the framework of the BEST project (financed by AMINAL-Nature; (Speybroeck et al. 2005b) an inventory of the main ecosystem components (avifauna, benthos, dry beach plants and insects) was made for eleven selected beaches along the Belgian coastline both spatially and temporally (seasonal). These eleven beaches (De Westhoek, Schipgatduinen, Zeebermduinen, Ijzermonding, Raversijde, Spinoladijk, Paelsteenpanne, de Fonteintjes, Zeebrugge, Baai van Heist, VNR “De Zwinduinen en –polders”) were considered rather natural at the time (2002-2004). Monitoring studies on beach nourishment (financed by the Flemish Coastal Waterways Division) have been carried out since 2002 (Speybroeck et al. 2003; Welvaert 2005; Van Ginderdeuren et al. 2007; Vanden Eede et al. 2008; Vanden Eede & Vincx 2010, 2011b) In total, eight intertidal beaches have been sampled abiotically and biotically (benthic components: macrobenthos, hyperbenthos and epibenthos) at different occasions and times (Oostende-Center, Oostende-East, Mariakerke, Wenduine, Bredene, Lombardsijde, Koksijde-Oostduinkerke, Nieuwpoort). These data provide a more realistic view of the current state of the Belgian beaches and as such enhance the simulating power of the model. The input data for the envelope models were derived from 300 beach samples in total, taken in the period 1997-2011 along the Belgian coast (Vanden Eede et al.; Degraer et al. 2003b). The biomass of chlorophyll a was assessed based on 72 samples from 9 locations (Speybroeck et al. 2008a).

The research of the Belgian beach ecosystem started about a decade ago (1997 – present). The simulation model is based upon quantitative information on the littoral macrobenthos community along the Belgian coastline provided by two major research projects, financed by different branches of the Flemish government. Within the framework of the BEST project (financed by AMINAL-Nature; (Speybroeck et al. 2005b) an inventory of the main ecosystem components (avifauna, benthos, dry beach plants and insects) was made for eleven selected beaches along the Belgian coastline both spatially and temporally (seasonal). These eleven beaches (De Westhoek (De Panne), Schipgatduinen, Zeebermduinen, Ijzermonding, Raversijde, Spinoladijk, Paelsteenpanne, de Fonteintjes, Zeebrugge, Baai van Heist, VNR “De Zwinduinen en –polders”) were considered rather natural at the time (2002-2004). Monitoring studies on beach nourishment (financed by the Flemish Coastal Waterways Division) have been carried out since 2002 (Speybroeck et al. 2003; Welvaert 2005; Van Ginderdeuren et al. 2007; Vanden Eede et al. 2008; Vanden Eede & Vincx 2010, 2011b). In total, eight intertidal beaches have been sampled at different occasions and times (Oostende-Center, Oostende-East, Mariakerke, Wenduine, Bredene, Lombardsijde, Koksijde-Oostduinkerke, Nieuwpoort). These data provide a more realistic view of the current state of the Belgian beaches and as such enhance the simulating power of the model. The input data for the envelope models were derived from 300 beach samples in total, taken in the period

1997 – 2011 along the Belgian coast (Degraer et al. 2003b; Vanden Eede et al. 2013, in prep.). The biomass of chlorophyll a was assessed based on 72 samples from 9 locations (Speybroeck et al. 2008a).

Abiotic variables

Median grain size of the sediment (*MGS*) was determined by laser diffraction using a Coulter LS Particle Size Analyzer (or Coulter-counter).

The total organic matter (*TOM*, in mass percentage) of the sediment was determined by drying a sediment sample at 110°C for two days, to obtain the dry weight of the sample. Afterwards the organic matter was removed by heating the sample for two hours at 450 °C, resulting in the ash weight. The difference between the former and the latter then gives the ash free dry weight that after division by the dry weight results in a measure for *TOM*.

Height of sampling stations and the entire beach profile were measured using a leveler. Afterwards, these readings were corroborated with the output of the M2-tidal reduction model (Flanders Marine Institute, www.lifewatch.be).

Macrobenthos

Macrobenthos (the infauna larger than 1 mm) was sampled by excavating a 0.1026 m^2 orthogonal frame to a depth of 15 cm. The sample was sieved alive in sea water, over a mesh size of 1 mm and afterwards fixated in 8 % formaldehyde solution. Samples were taken at the water line at equal time intervals, starting at high tide and following the receding tide until low tide. After staining the samples with Rose Bengal, the organisms were separated from the sample residue and all organisms were identified to species level.

Beach profile

The beach profile with height above low tide or elevation (h) along a transect of 400 m is used as basic input of the model because this length captures the intertidal region on Belgian beaches. MGS and TOM are estimated based on the following regressions, obtained from 23 beaches that served as input for the species envelope assessment (partly published in Degraer *et al.,* 2003). The resulting MGS and TOM estimates are non-deterministic and based on sampling from regression parameter distributions (mean ± SD).

$$MGS = 193.8\ (\pm 11.52) + 13.87\ (\pm 1.32) * h \qquad \text{(eq.1)}$$

$$TOM = 1.82\ (\pm 0.21) - 0.009\ (\pm 0.001) * MGS + 0.000016\ (\pm 0.000001) * MGS^2 \qquad \text{(eq. 2)}$$

Model validation

In order to validate the model, we sampled three transects along two beaches for the dominant macrobenthos species in an identical way as described in Degraer et al. (2003). Abundance of the dominant species, biomass (total ash free dry weight, AFDW) and species richness from the samples were subsequently compared with simulated data (average values and 95 % confidence intervals for 10 000 simulations) according to the sample location h and grain size MGS.

As evidenced from figure A, observed abundances and species richness fall within the confidence intervals of the model predictions. Note, however, that observed values are derived from samples with surface 0.1026 m², while model estimates are always per m². This discrepancy in scale is responsible for the higher observed numbers of samples with zero individuals (plotted as 0.001 on the log-scale plots in figure S1) and higher estimates of species richness by the simulation model. Total availability of chlorophyll a (mg.m-²), which is a measure for microphytobenthos, is estimated without conversion (m² scale).

Unfortunately, we lack data on higher trophic levels. This was especially the case for birds, since they are additionally impacted by other factors like proximity to resting areas and disturbance, not allowing for any validation.

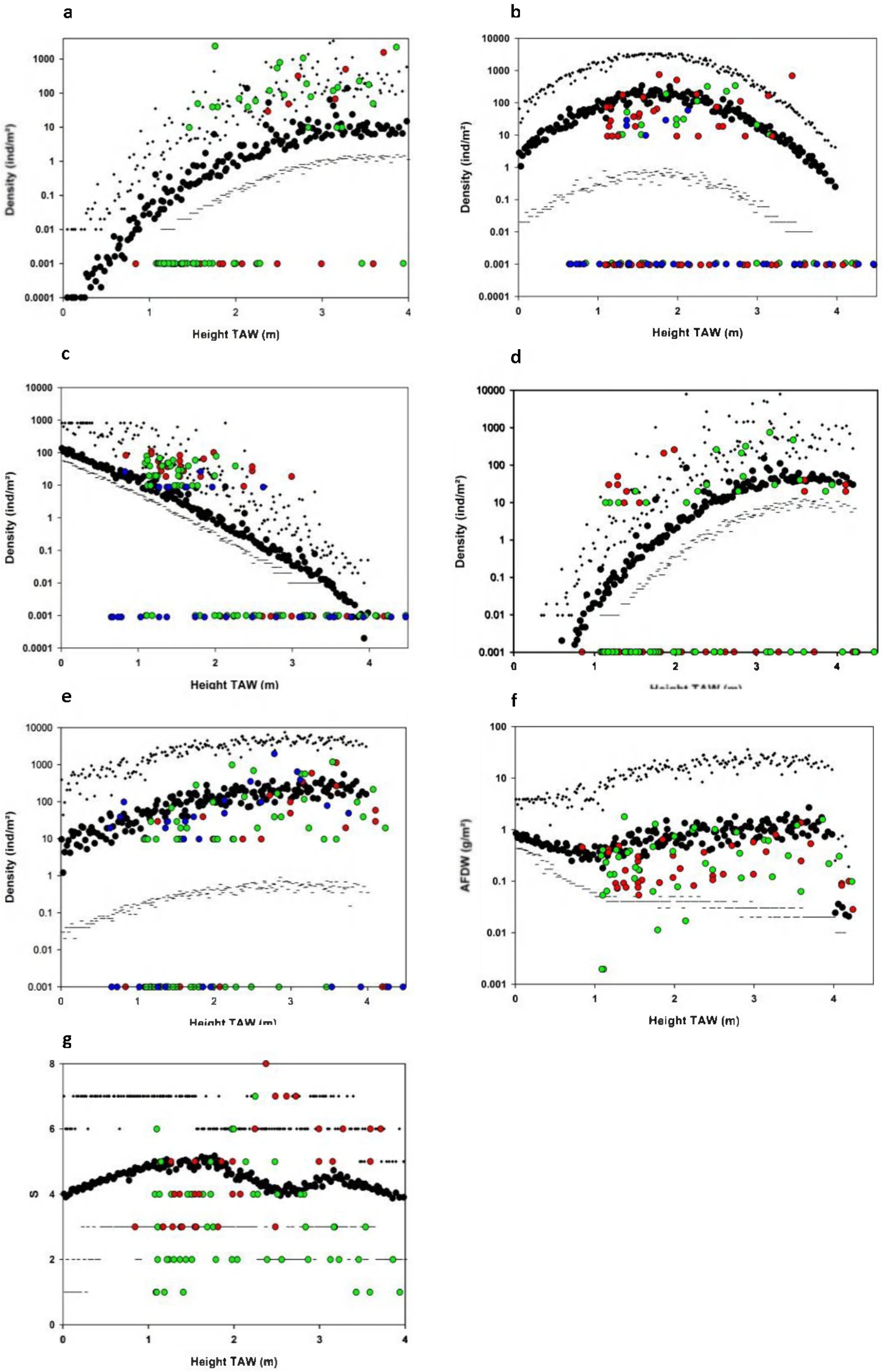


Figure A: Observed and expected abundance for the main macrobenthos species (a: *Bathyporeia pilosa*; b: *Bathyporeia sarsi*; c: *Eurydice pulchra*, d: *Nephtys cirrosa*, e: *Scolelepis squamata*), total macrobenthos biomass (f) and species richness (g). Note that for (a-f), y-values are on log scale. In figure f, observed species richness comprises values within one sample (0.10 m²), while expected values are estimates per m². Observed values are depicted in coloured circles (green: Mariakerke, red: Lombardsijde transect 1, blue: Lombardsijde transect 2), average estimated values from the model in black filled circles, lower limits in (-) and upper limits in (+).

Appendix B: Estimated regression coefficients by Bayesian modelling for the studied macrobenthos species and chorophyl concentraties

Mean estimates are provided with standard deviation between brackets. The 95 % credibility intervals are depicted between square brackets in grey tones. The Poisson error structure was found to generate the best convergence for all species, and was thus chosen because of its intrinsic simplicity relative to zero inflated Poisson and Negative Binomial regressions.

taxa	β_0	β_1 (MGS)	β_2 (MGS^2)	β_3 (h)	β_4 (h^2)	β_5 (TOM)	β_6 (TOM^2)
P. elegans	3.2 (1.43)	-3.0 (2.3)	-7.4 (3.38)	-1.13 (0.67)	-2.14 (0.6)	0.43 (0.74)	-1.06 (0.81)
	[0.49,5.9]	[-8.5,0.51]	[-14.5,-1.97]	[-2.68,0.07]	[-3.35,-1.07]	[-1.15,1.84]	[-2.68,0.47]
S. filicornis	-0.22 (2.04)	-2.82 (2.43)	-6.48(3.34)	-5.38 (1.06)	-1.69 (0.88)	0.8 (0.7)	-0.25 (0.68)
	[-4.64,3.35]	[-7.76,1.67]	[-13.39,-0.86]	[-7.65,-3.4]	[-3.85,-0.25]	[-0.52,2.34]	[-1.69,1.05]
E. longa	4.76 (0.95)	1.65 (0.75)	-1.27 (0.43)	-0.93 (0.34)	-1.28 (0.26)	1.04 (0.46)	-0.96 (0.34)
	[3.14,7.21]	[0.29,3.23]	[-2.18,-0.52]	[-1.59,-0.25]	[-1.82,-0.81]	[0.12,1.92]	[-1.66,-0.32]
C. capitata	0.93 (1.61)	-0.19 (1.31)	-0.28 (0.73)	-2.24 (0.57)	-0.14 (0.4)	0.68 (0.89)	0.15 (0.22)
	[-1.82,4.52]	[-2.54,2.22]	[-2.1,0.88]	[-3.63,-1.27]	[-0.95,0.65]	[-0.78,2.85]	[-0.26,0.62]
S. squamata	2.05 (2.28)	1.18 (0.51)	-0.46 (0.19)	0.49 (0.50)	-0.81 (0.31)	-0.50 (0.57)	0.15 (0.11)
	[-4.40,4.22]	[-0.39,1.94]	[-0.73,0.20]	[-0.90,1.2]	[-1.19,0.11]	[-1.93,0.24]	[-0.01,0.46]
N.hombergii	-13.22 (5.54)	-5.56 (5.68)	-5.89 (7.49)	-16.27 (9.2)	-6.94 (5.82)	1.27 (2.21)	0.82 (2.34)
	[-28.61,-5.17]	[-20.1,3.11]	[-27.92,2.94]	[-37.88,-4.63]	[-19.02,1.51]	[-2.85,6.02]	[-3.14,6.97]
N.cirrosa	-1.41 (0.69)	-1.03 (0.47)	-0.22 (0.31)	-3.51 (0.33)	-0.09 (0.28)	-0.29 (0.31)	-0.17 (0.19)
	[-2.76, -0.14]	[-2.15,-0.21]	[-0.85,0.36]	[-4.2,-2.89]	[-0.71,0.44]	[-0.93,0.30]	[-0.58,0.15]
B. pilosa	-1.18 (0.25)	-0.19 (0.65)	-0.15 (0.12)	3.54 (0.18)	-1.13 (0.13)	-0.15 (0.31)	0.17 (0.11)
	[-2.22,-1.30]	[-1.55,0.91]	[-0.42,0.07]	[3.05,3.87]	[-1.35,0.93]	[-0.83,0.35]	[-0.03,0.45]
E. pulchra	-0.04 (1.41)	0.52 (0.76)	-0.52 (0.25)	2.75 (0.67)	-1.61 (0.40)	-0.28 (0.46)	0.04 (0.13)
	[-4.15,1.48]	[-1.02,2.01]	[-0.98,0.02]	[0.99,3.72]	[-2.14,0.44]	[-1.16,0.62]	[-0.20,0.29]
E. affinis	0.26 (1.88)	-1.08 (0.68)	-0.25 (0.28)	-0.36 (0.39)	-1.68 (0.39)	0.24 (0.54)	-0.22 (0.28)
	[-5.48,2.48]	[-2.65,0.17]	[-0.77,0.43]	[-1.15,0.51]	[-2.31,0.43]	[-0.64,1.40]	[-0.92,0.22]
B. sarsi	4.02 (0.93)	-0.65 (0.82)	-0.63 (0.59)	-0.21 (0.48)	-2.51 (0.53)	-0.66 (0.79)	-0.15 (0.46)
	[2.37,5.75]	[-2.36,0.9]	[-1.76,0.54]	[-1.11,0.75]	[-3.66,-1.54]	[-2.52,0.65]	[-1.09,0.67]
Chla	0.1635 (0.5652)	-0.634 (0.246)	-0.097 (0.075)	-0.435 (0.096)	-0.1012 (0.108)		
	[-1.026,1.02]	[-1.149,-0.158]	[-0.0501, 0.2458]	[-0.6265,- 0.2382]	[-0.314, 0.11]		

We subsequently expected the abundance of species y in sample i to follow a Poisson distribution:
$y(i) \sim \mathrm{dpois}\,(\theta(i))$ with expected abundance in sample i: $\theta(i) = e\eta(i)$.

$\eta(i)$ is a mixed function with overall intercept $\beta0$ and both linearly and quadratically dependent on the abiotic variables MGS, TOM and h.

All independent variables were Z-transformed to guarantee standardized effect weights. We additionally incorporated variance estimates related to dependency within transects among samples (u) and the overall residual variation (v) to account for possible over-or underdispersion among samples from transects (eq. 1). As such we modelled average abundances across seasons among different beaches.

The full model formulation is:

$$\eta(i) = \beta_0 + \beta_1 \times \frac{MGS(i) - \overline{MGS}}{stdev(MGS)} + \beta_2 \times \left(\frac{MGS(i) - \overline{MGS}}{stdev(MGS)} \right)^2 + \beta_3 \times \frac{h(i) - \overline{h}}{stdev(h)} + \beta_4 \times \left(\frac{h(i) - \overline{h}}{stdev(h)} \right)^2$$

$$+ \beta_5 \times \frac{OM(i) - \overline{OM}}{stdev(OM)} + \beta_6 \times \left(\frac{OM(i) - \overline{OM}}{stdev(OM)} \right)^2 + u(beach(i)) + v(i) \qquad \text{(eq. 1)}$$

Because we had no a priori information, flat priors for regression coefficients were drawn from a normal distribution with a mean of 0 and a standard deviation SD of 106. Priors for variance components were drawn from a positively constrained uniform distribution with a mean of 1 and SD 5. Three chains were modelled for each model. To assure accurate MCMC simulations from the prior distributions, an initial “burn in” of 10 000 iterations was performed and discarded from the analysis. This was followed by 20 000 iterations for all analyses. After visual inspections for possible autocorrelation and assessing chain convergence Brooks-Gelman-Rubin diagnostics (Brooks & Gelman 1998), the mean and SD of each posterior parameter, estimated regression coefficients and variance estimates were calculated, as well as the 2.5th and 97.5th percentiles of the samples. These were used to describe the 95 % Bayesian credibility interval of the posterior distributions of model parameters. An overview of all the estimated regression coefficients can be found in the table at the beginning of this appendix.

Appendix C: Average conversion factors for converting abundance to biomass (g AFDW) (Vanden Eede, unpublished data)

	Species	**g AFDW**
Amphipoda	*Bathyporeia pilosa*	0.00025
Amphipoda	*Bathyporeia sarsi*	0.00033
Annelida (Spionidae)	*Scolelepis squamata*	0.00479
Annelida (Spionidae)	*Pygospio elegans*	0.00016
Annelida (Spionidae)	*Spio filicornis*	0.00010
Annelida (Phyllodocidae)	*Eteone longa*	0.00016
Annelida (Capitellidae)	*Capitella capitata*	0.00007
Annelida (Nephtyidae)	*Nephtys cirrosa*	0.00434
Annelida (Nephtyidae)	*Nephtys hombergii*	0.00500
Isopoda	*Eurydice pulchra*	0.00066
Isopoda	*Eurydice affinis*	0.00066

Appendices – Chapter 6

Appendix A: Basis for selection of species as Habitat Forming or Ecologically Significant, per ecosystem component with corresponding Aphia ID (WORMS). Criteria for selection are represented by the number code as stated here: (1) Top predators (for benthos only); (2) Important food source; (3) Species present in conservation lists (IUCN Red List, Bird Directive Annex I, Bern Convention, Belgian Birds Red List); (4) Species most exclusively linked to the presence of *Lanice conchilega* - This tube worm builds small reefs patches on the seabed surface attracting other species by offering refuge for predators; (5) Most important species of the *Abra alba* community - One of the ecologically most important soft-sediment (muddy fine sandy sediment) macrobenthic communities, characterized by a high diversity, abundance and biomass; (6) Coastal birds occurring in more than 1 % along the Belgian coast - According to the Birds Directive, species for which more than 1 % of their biogeographical population is located in the study area, are considered ecologically significant and should be protected, preferable through special areas of conservation.

AphiaID	***Scientifc name***	***Habitat forming spp.***	***Ecologically significant spp.***	
		Expert Judgment	Expert Judgment	Van Hoey *et al.*2005, 2007
MACROBENTHOS				
141433	*Abra alba*		**2**	**5**
129868	*Arenicola marina*	**Yes**		
123776	*Asterias rubens*		**1**	
103068	*Bathyporeia pilosa*		**2**	
103073	*Bathyporeia sarsi*		**2**	
130644	*Eumida sanguinea*		**5**	**4**
118843	*Eurydice affinis*		**1**	
118852	*Eurydice pulchra*		**1**	
102431	*Jassa falcata*	**Yes**		
131495	*Lanice conchilega*	**Yes**		
140380	*Kurtiella bidentata*		**2**	**5**
130359	*Nephtys hombergii*			**5**
130544	*Owenia fusıformis*	**Yes**		
130595	*Pectinaria koreni*	**Yes**		
101857	*Pariambus typicus*		2	**4**
131170	*Pygospio elegans*			
157566	*Scolelepis squamata*		**2**	
131187	*Spiophanes bombyx*	**Yes**		
141587	*Tellina fabula*		**2**	**5**
HYPERBENTHOS				
126925	*Pomatoschistus lozanoi*		**2**	
126926	*Pomatoschistus microps*		**2**	
126928	*Pomatoschistus minutus*		**2**	
BIRDS				
147431	*Arenaria interpres*		**6**	
137138	*Larus argentatus*		**6**	
137142	*Larus fuscus*		**6**	
137157	*Sterna albifrons*		**3 and 6**	
137162	*Sterna hirundo*		**3 and 6**	
413044	*Thalasseus sandvicensis*		**3 and 6**	

Appendix B: Algorithms used to apply the assessment questions for the different ecosystem components. If there are no data available for a certain subzone within a study area, this subzone is labeled 'NA' and is not incorporated when the algorithm is applied (adapted from Derous et al., 2007b). All calculation steps were incorporated into the R script. Key: SR= Species Richness; ESS= Ecologically Significant Species; HFS= Habitat-Forming Species.

Assessment Question (Acronym)	*Algorithm*
High Counts of Many Species (QHCMS)	**1** Determine the spp. which are regularly occurring in your study area, selecting all the species which occur in more than 10 % of your records (hence excluding rare spp. from the list). **2** Interpolate density data of spp. to the chosen subzones. **3** Create 5 density classes with values between 1 and 5 (with an equal amount of subzones in each class). **4** Assign values to data for all spp. and sum the values in every subzone. **5** Divide the resulting summed values again in 5 classes (with an equal amount of subzones in each class).
High Species Richness (QHSR)	**1** Determine the average SR for each subzone. **2** Create 5 classes for SR ranging from 1 to 5 (with an equal amount of subzones in each class.
Ecologically Significant Species (QESS)	**1** Select ESS from species list (rationale for selection *see table A1*). **2** Create 5 density classes for these spp. with values ranging from 1 to 5 (with an equal amount of subzones in each class).Class 1 holds subzones without any ESS. **3** If there are several ESS present in the study area, then create a different density class for each species separately and average the values afterwards.
Abundance of Certain Species (QACS)	**1** Determine the species which are regularly occurring in your study area by selecting all species which occur in more than 10 % of the subzones (exclude rare species from the species list). **2** Determine the mean density of every species for the whole study area (=X) **3** Calculate the mean density of every species for every subzone (=Xi). **4** Calculate the ratio Xi/X for every species in each subzone **5** Determine the 5 % subzones with the highest ratio. Calculate the percentage of the density of every species that occurs in the 5 % most important subzones (=Y) **6** Determine in how many subzones every species occurs (=Z). **7** Calculate the ratio Y/Z which is the aggregation coefficient for each species. **8** Multiply the ratio Y/Z with the ratio Xi/X and divide these values in 5 classes with values between 1 and 5 (with an equal amount of subzones in each class).
Habitat Forming Species (QHFS)	**1** Select HFS from species list (rationale for selection *see table A1*). **2** Create 5 density classes for these spp. with values ranging from 1 to 5 (with an equal amount of subzones in each class). Class 1 holds subzones without any HFS. **3** If there are several HFS present in the study area, then create a different density class for each species separately and average the values afterwards
Abundance (or Presence) of Rare Species (QARS/QPRS)	**1** Determine the spp. which occur in less than 10 % of your subzones (rare species). 2 Interpolate density (QARS) or presence (QPRS) data of spp. to the chosen subzones. 3 Create 5 density (QARS) or presence (QPRS) classes with values between 1 and 5 (with an equal amount of subzones in each class). 4 Assign values to data for all spp. and sum the values in every subzone. 5 Divide the resulting summed values again in 5 classes (with an equal amount of subzones in each class).

Table C1: Example of the proposed scoring system for a hypothetical study area with 6 subzones and 2 ecosystem components, with 1st order criteria questions only (Criteria: R= rarity; AF= Aggregation/ Fitness consequences). The individual scores for every criterion and the data availability levels are also hypothetical and only used to illustrate the scoring process. When no biological data is available for certain subzones this is indicated by *'NA'* (Adapted from Derous et al 2007b); Key: VL= Very Low; L=Low; M=Medium; H= High; VH= Very High

Ecosystem component	*Assessment Question*	*Subzones*					
		1	**2**	**3**	**4**	**5**	**6**
Birds	High counts of many species (AF)	2	5	NA	1	4	1
	High abundance of certain species (AF)	5	4	NA	3	3	2
	Number of questions (#Q) answered	2	2	0	2	2	2
	Total score	7	9	NA	4	7	3
	*Intermediate Value (*****see table A4****)*	**H**	**VH**	**NA**	**VL**	**H**	**VL**
	Birds data availability [a]	3	1	0	3	3	2
Macrobenthos	High counts of many species (AF)	3	NA	2	NA	3	3
	High abundance of certain species (AF)	3	NA	4	NA	4	4
	Presence of rare species (R)	1	NA	5	NA	4	2
	Ecologically Significant Species (R, AF)	2	NA	1	NA	5	1
	Number of questions (#Q) answered	5	0	5	0	5	5
	Total score	9	0	12	0	16	10
	*Intermediate Value (*****see table A5****)*	**VL**	**NA**	**M**	**NA**	**VH**	**VL**
	Macrobenthos data availability [a]	3	1	3	1	3	3
	Intermediate Value for Birds	**H**	**VH**	**NA**	**VL**	**H**	**VL**
	Intermediate Value for Macrobenthos	**VL**	**NA**	**M**	**NA**	**VH**	**VL**
	Average Total Numeric Value	3	5	3	1	5	1
	*Average Value (*****see table A6****)*	**L**	**VH**	**M**	**VL**	**VH**	**VL**
	Total Average Data Availability	**H**	**L**	**L**	**M**	**H**	**M**
	*Information Reliability (*****see Table 7****)*	**H**	**L**	**M**	**L**	**H**	**H**

[a] Data availability is given by the number of observations (samples) taken per subzone, for each ecosystem component. The values are then categorized in 3 classes using the same method shown for Information Reliability (table A7)

Table C2: Classification of Intermediate Value for Birds into 5 classes. Both the raw equations and the calculations for hypothetical example data (table A3) are given

Quantile intervals (x) given by: $\mathrm{x} = \frac{Max\ (Total\ score) - Min\ (Total\ score)}{5}$

For this example: $x = \frac{9-3}{5} = 1.2$

Intervals for Intermediate Values (Table 3)

Interval range (Equations)		Interval range (Example)		Final Classification
[*Min;*	*Min+x*[	[**3**(*Min*);	3+1.2=**4.2** [	1= Very Low (**VL**)
[*Min+x;*	*Min + 2x*[	[**4.2**;	**5.4**[	2= Low (**L**)
[*Min+2x;*	*Min+3x* [	[**5.4**;	**6.8**[	4= Medium (**M**)
[*Min+3x;*	*Min+4x[*	[**6.8**;	**8**[	3= High (**H**)
[*Min+4x;*	*Max*]	[**8**;	**9**(*Max)*]	5= Very High (**VH**)

Table C3: Classification of Intermediate Value for Macrobenthos into 5 classes. Both the raw equations and the calculations for hypothetical example (table A3) are given

Quantile intervals (x) given by: $\mathrm{x} = \frac{Max\ (Total\ score) - Min\ (Total\ score)}{5}$

For this example: $x = \frac{16-9}{5} = 1.4$

Intervals for Intermediate Values (Table 3)

Interval range (Equations)		Interval range (Example)		Final Classification
[*Min;*	*Min+x*[	[9 *(Min)*;	9+1.4= **10.4**[	1= Very Low (**VL**)
[*Min+x;*	*Min + 2x*[	[**10.4**;	**11.8** [	2= Low (**L**)
[*Min+2x;*	*Min+3x* [	[**11.8**;	**13.2**[	4= Medium (**M**)
[*Min+3x;*	*Min+4x[*	[**13.2**;	**14.6**[	3= High (**H**)
[*Min+4x;*	*Max*]	[**14.6**;	**16** *(Max)*]	5= Very High (**VH**)

Table C4: Determination of the Total Value using the numerical equivalents of the Intermediate Values

Interval range		**Final Classification**
Min.	Max.	
[1;	1.8[	1= Very Low (**VL**)
[1.8;	2.6 [	2= Low (L)
[2.6;	3.4[	4= Medium (**M**)
[3.4;	4.2[	3= High (H)
[4.2;	5]	5= Very High (**VH**)

Table C5: Determination of information reliability per subzone and classification into 3 classes

Information Reliability is given by:

$$\frac{\#Q\ (answered\ per\ subzone)}{\#Q\ (total\ maximum\ answered)}$$

Interval range		**Final Classification**
Min.	Max.	
[0;	**0.33**[	1= Low (L)
[**0.33**;	**0.66**[	2= Medium (**M**)
[**0.66**;	1]	3= High (H)

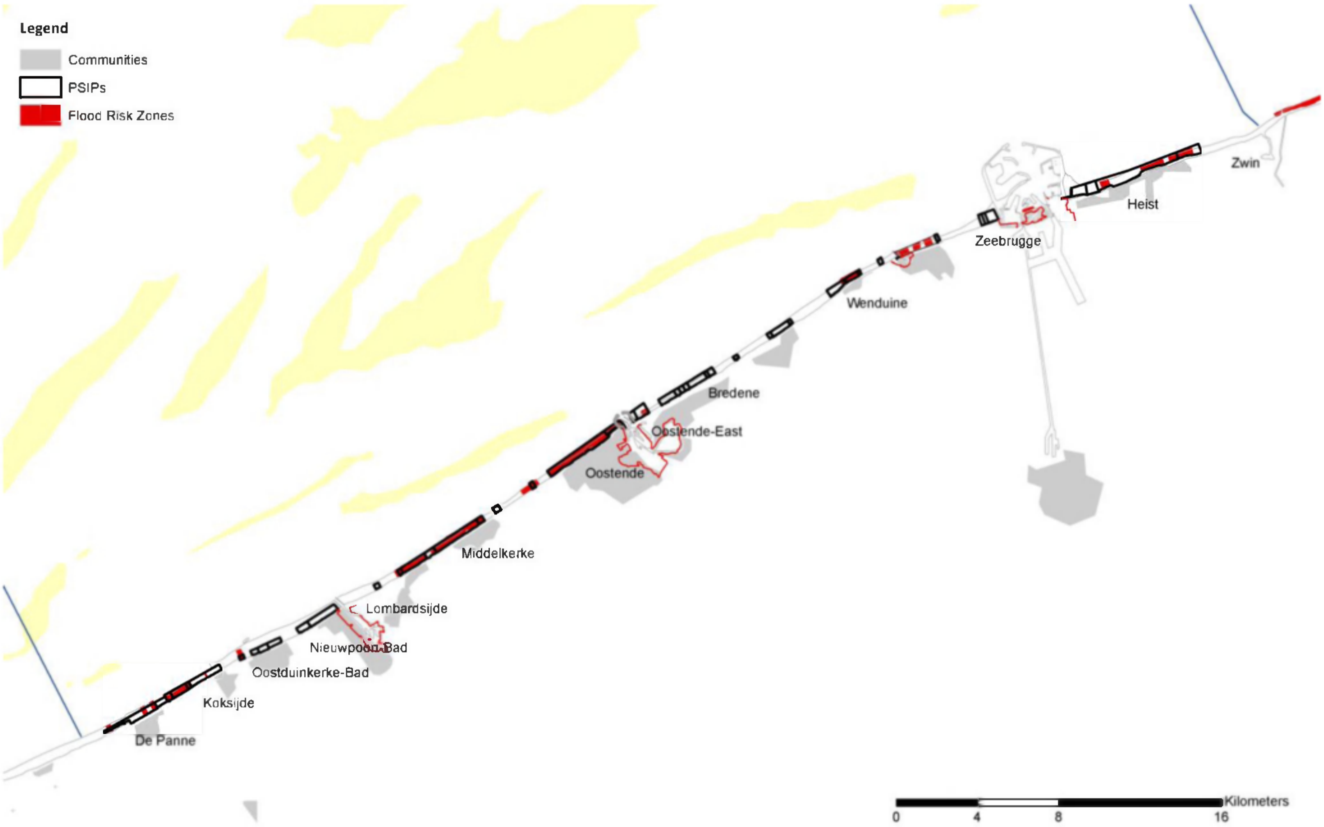


Figure D1: Areas of high coastal flood risk (in red) and the Provincial Spatial Implementation Plans (PSIPs) at the Belgian coast.

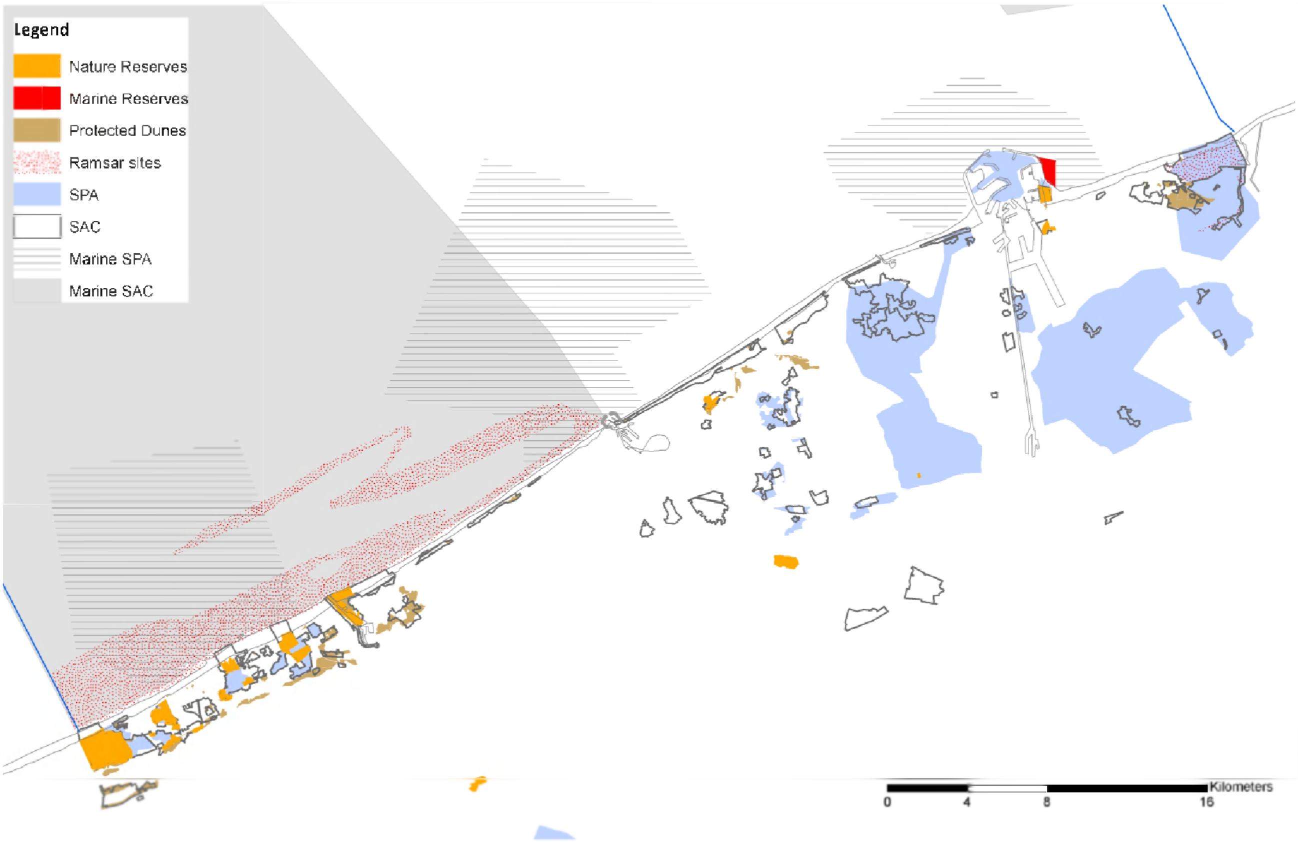


Figure D2: Map displaying protected areas under national and international obligations at the Belgian coast (SAC=Special Area of Conservation; SPA=Special Protection Area)

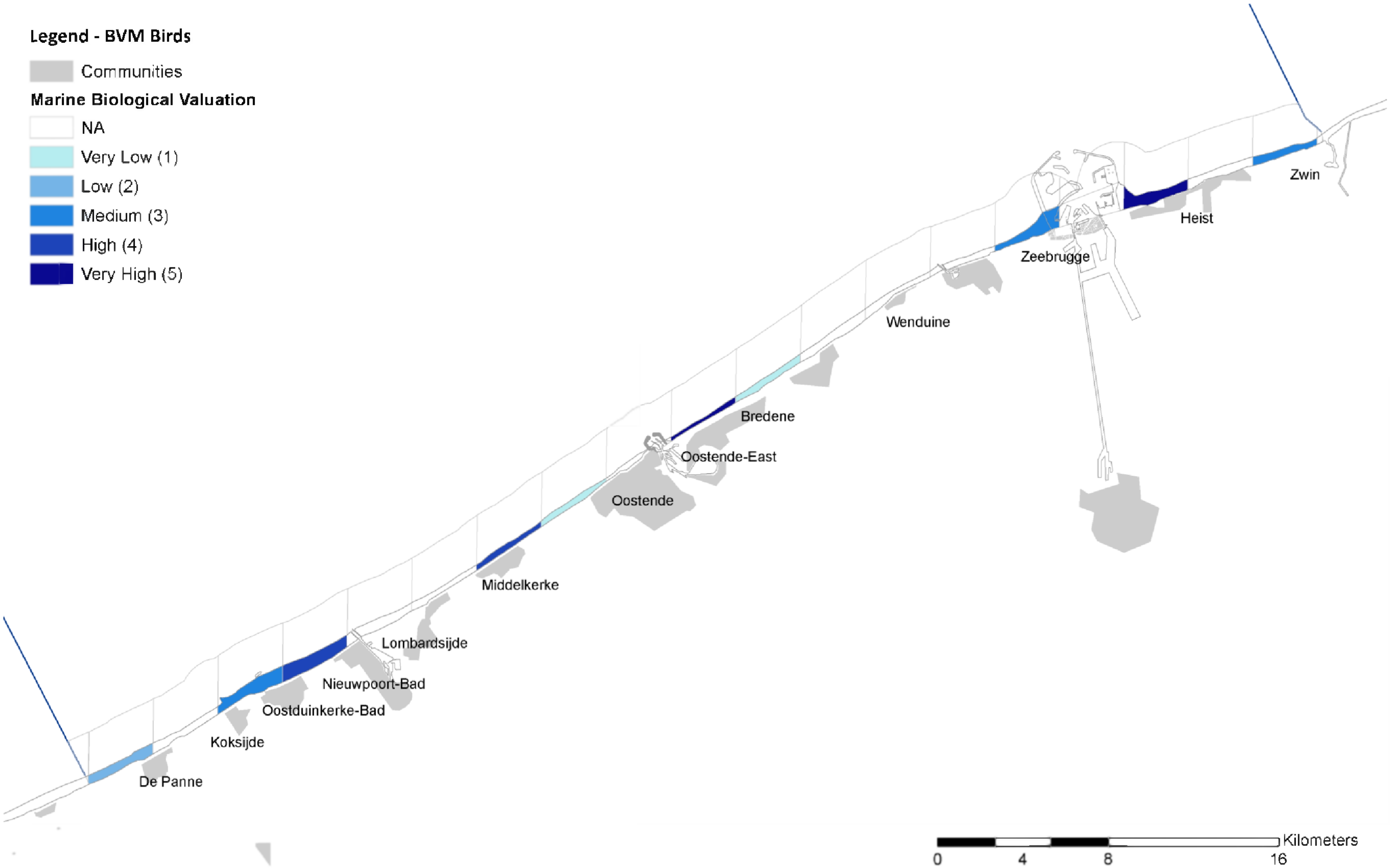


Figure E1: Intermediate Valuation Map showing the Biological Valuation results for the birds component. Note that blanks indicate subzones where no data on this component was available to perform a valuation. The highest biological value is observed for the subzone comprising the area of Baai van Heist.

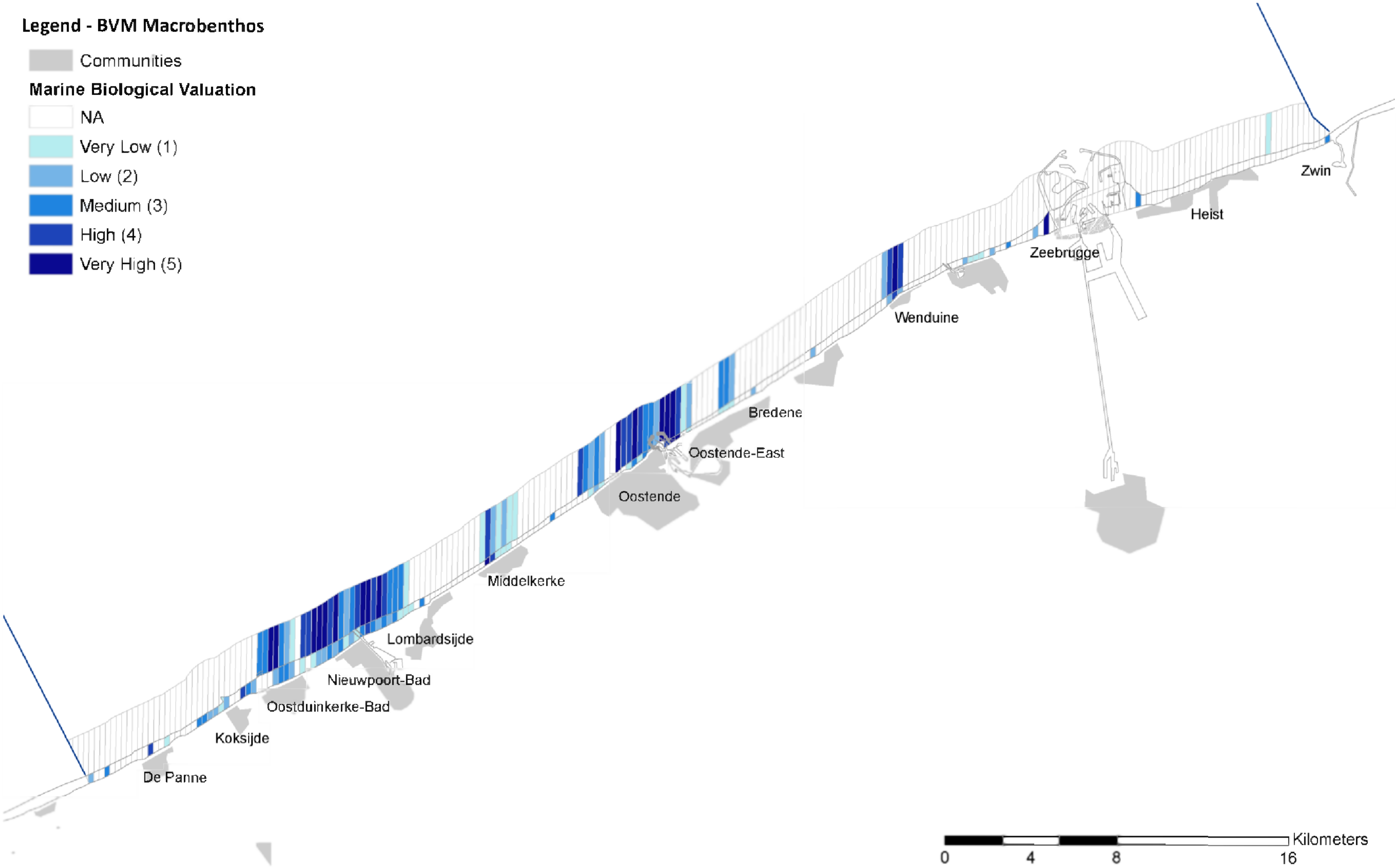


Figure E2: Intermediate Valuation Map showing the Biological Valuation results for the macrobenthos component. Note that blanks indicate subzones where no data on this component was available to perform a valuation.

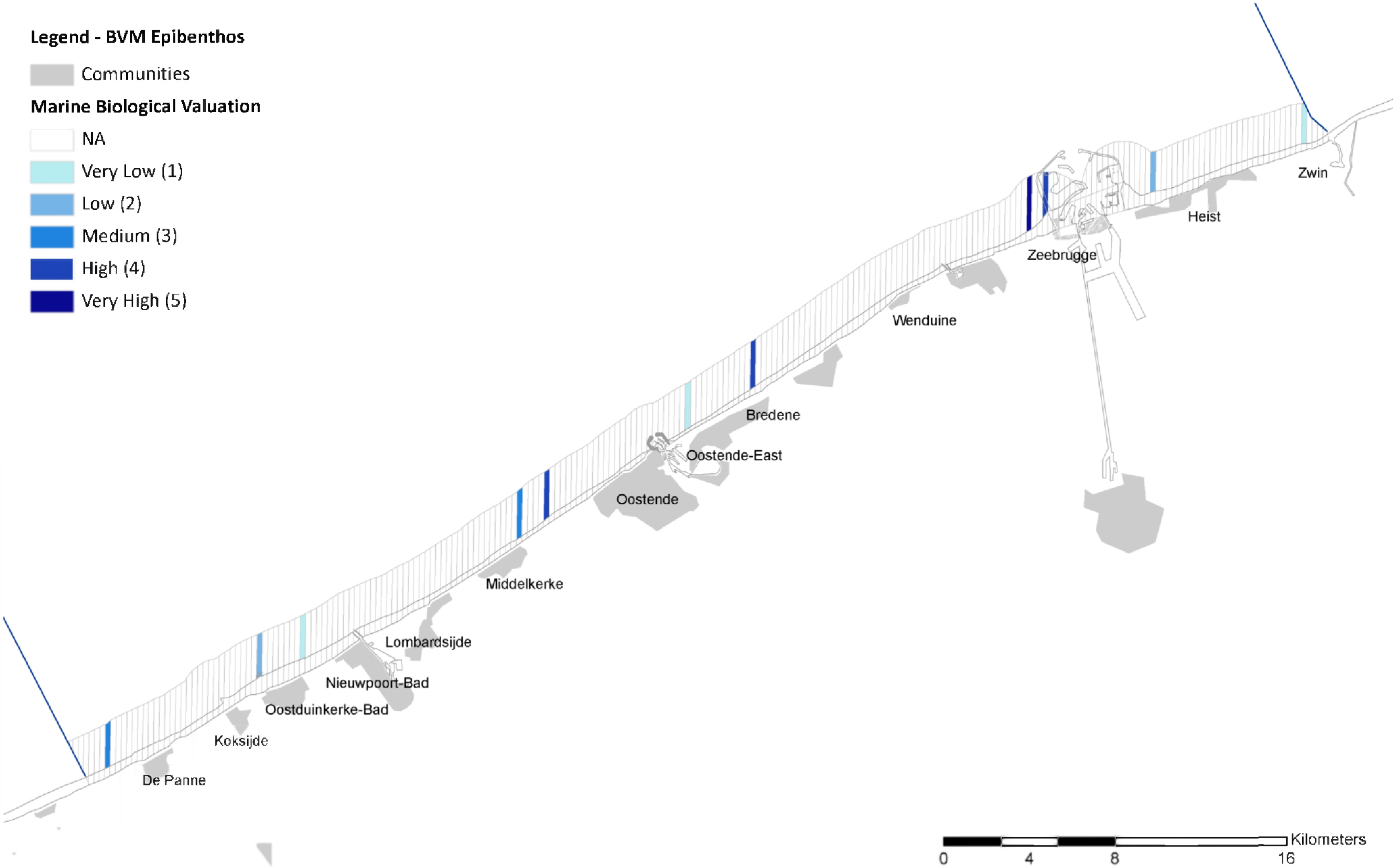


Figure E3: Intermediate Valuation Map showing the Biological Valuation results for the epibenthos component. Note that blanks indicate subzones where no data on this component was available to perform a valuation.

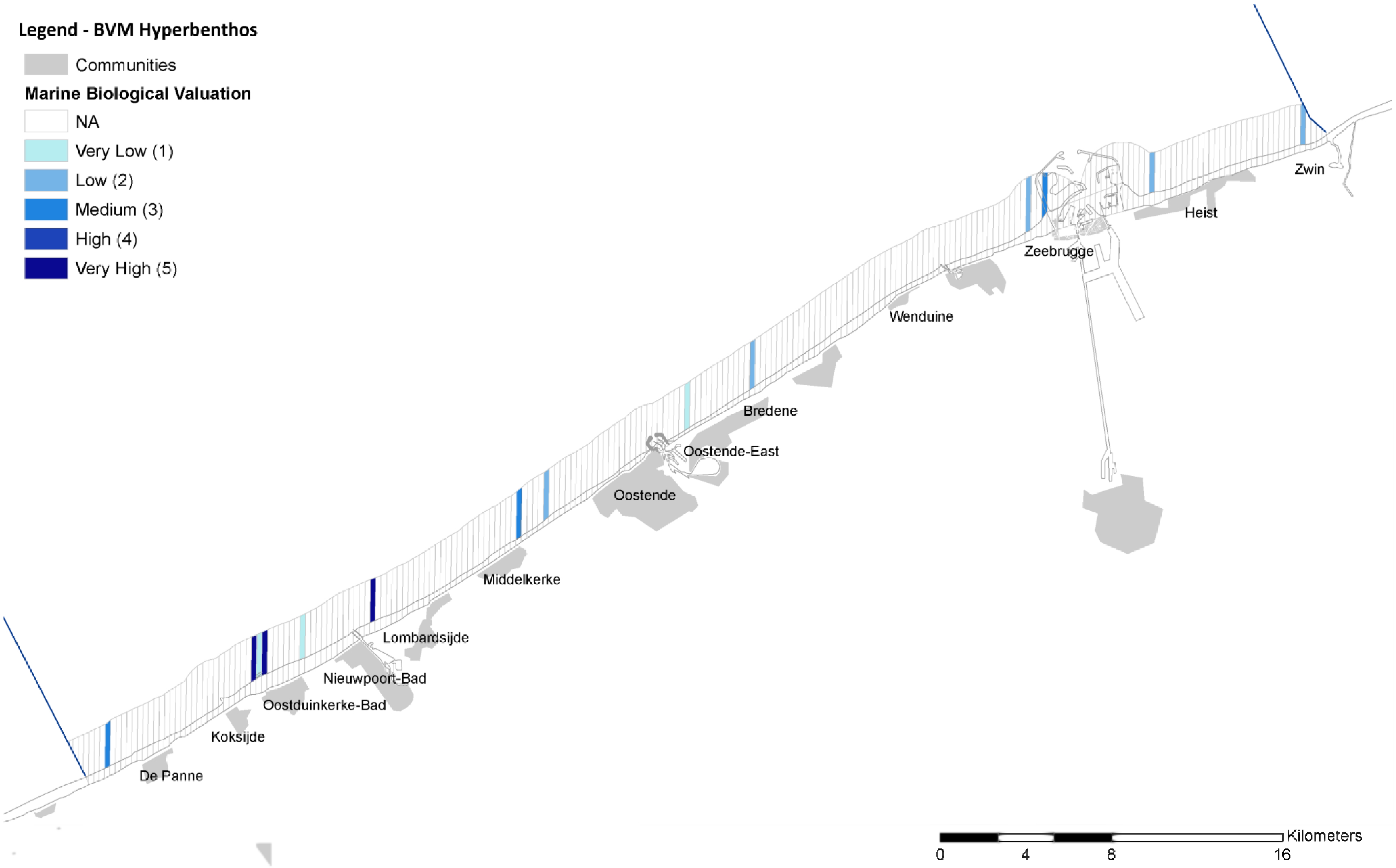


Figure E4: Intermediate Valuation Map showing the Biological Valuation results for the hyperbenthos component. Note that blanks indicate subzones where no data on this component was available to perform a valuation.

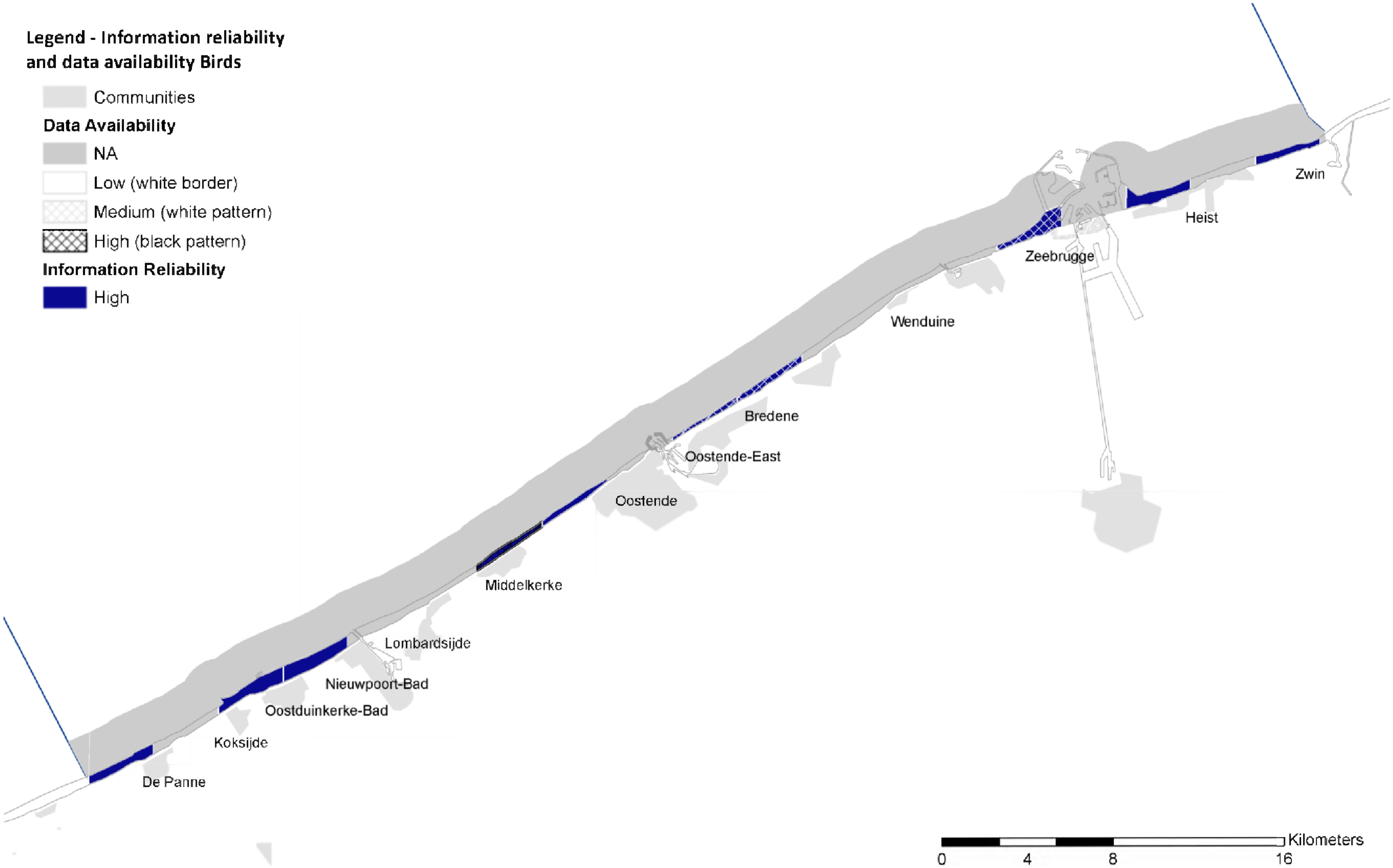


Figure F1: Map showing the information reliability and data availability for the birds component. Note that dark grey areas indicate subzones where no data on this component was available to perform a valuation. Information reliability was high for all subzones valued, meaning that all assessment questions could be answered in all subzones containing data. Low data availability was not found.

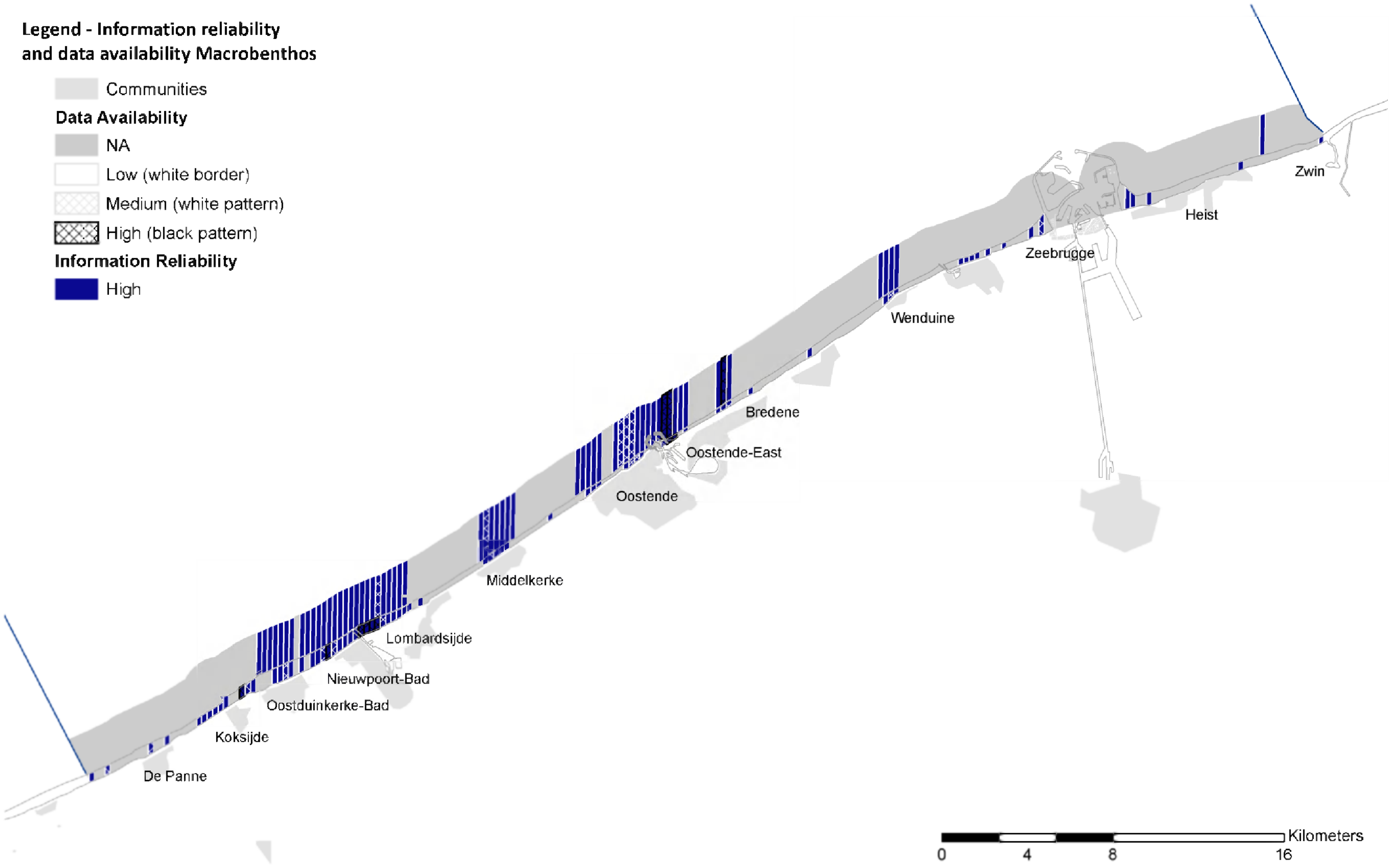


Figure F2: Map showing the information reliability and data availability for the macrobenthos component. Note that dark grey areas indicate subzones where no data on this component was available to perform a valuation. Information reliability was high for all subzones valued, meaning that all assessment questions could be answered in all subzones containing data.

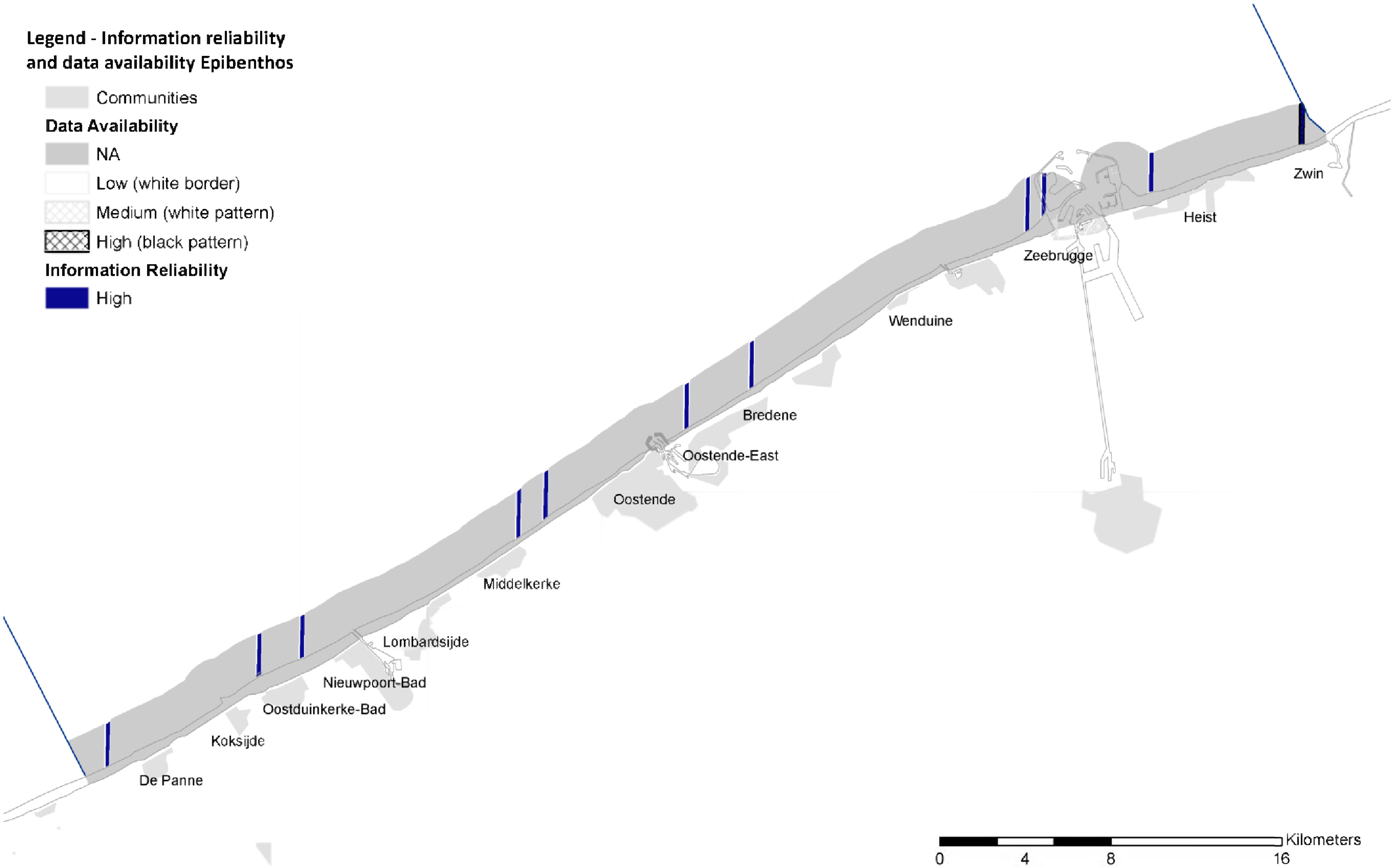


Figure F3: Map showing the information reliability and data availability for the epibenthos component. Note that dark grey areas indicate subzones where no data on this component was available to perform a valuation. Information reliability was high for all subzones valued, meaning that all assessment questions could be answered in all subzones containing data.

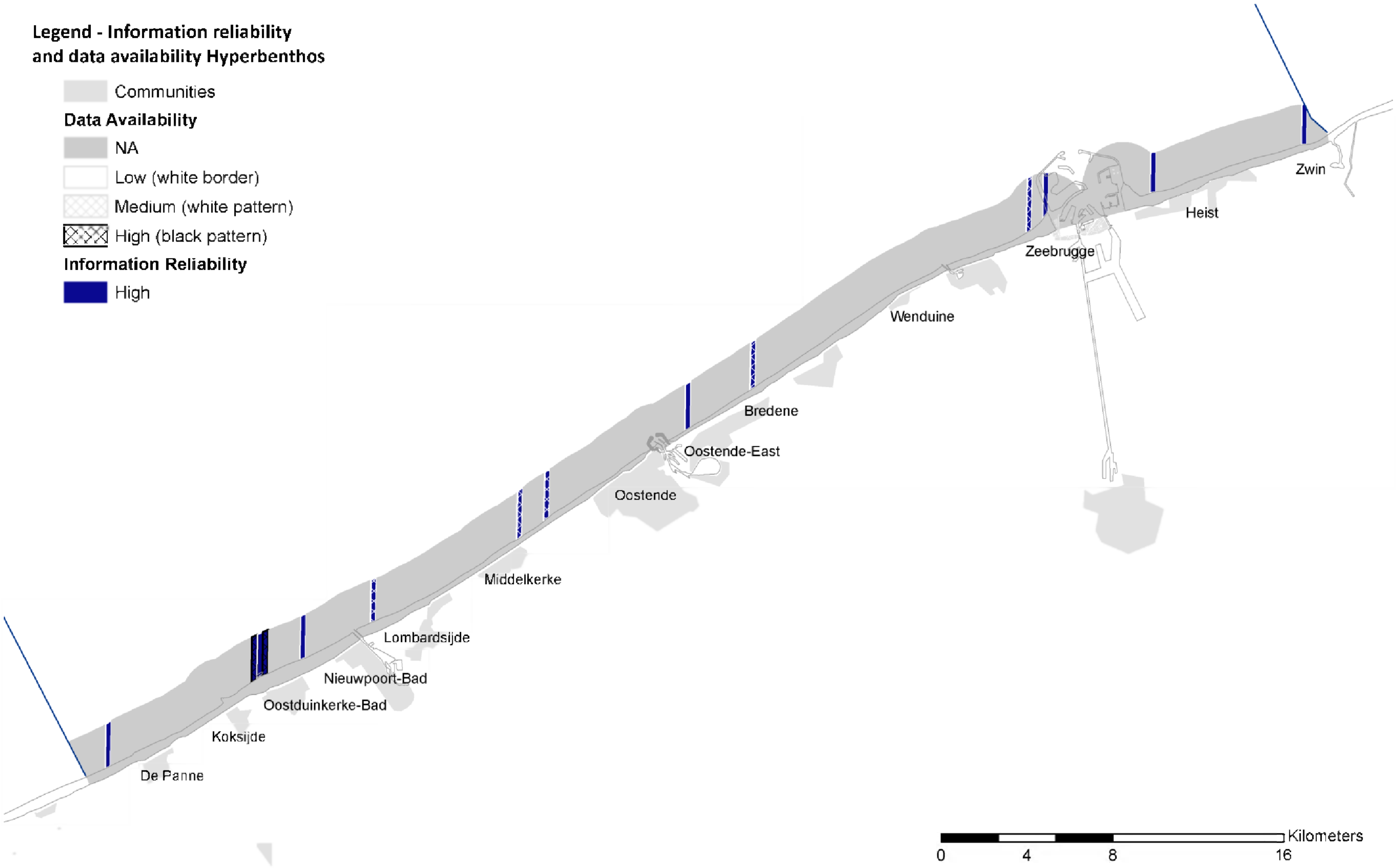


Figure F4: Map showing the information reliability and data availability for the hyperbenthos component. Note that dark grey areas indicate subzones where no data on this component was available to perform a valuation. Information reliability was high for all subzones valued, meaning that all assessment questions could be answered in all subzones containing data.

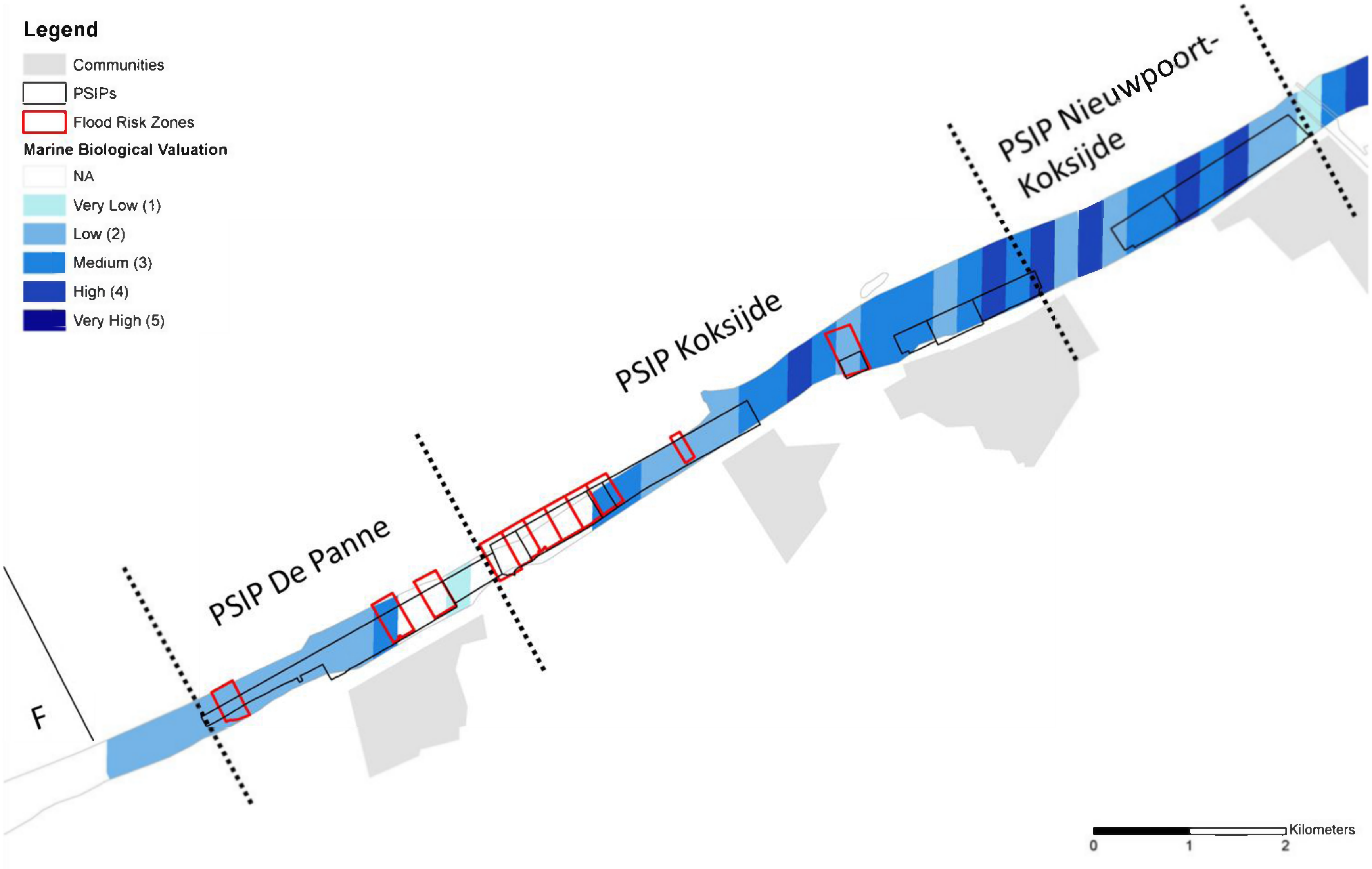


Figure G1: Detailed map with Marine Biological Valuation scores of intertidal subzones inside the PSIPs of De Panne, Koksijde and Nieuwpoort-Koksijde. Red indicates areas under coastal flood risk. The dashed lines mark the boundaries of each PSIP while F indicates the French territory.

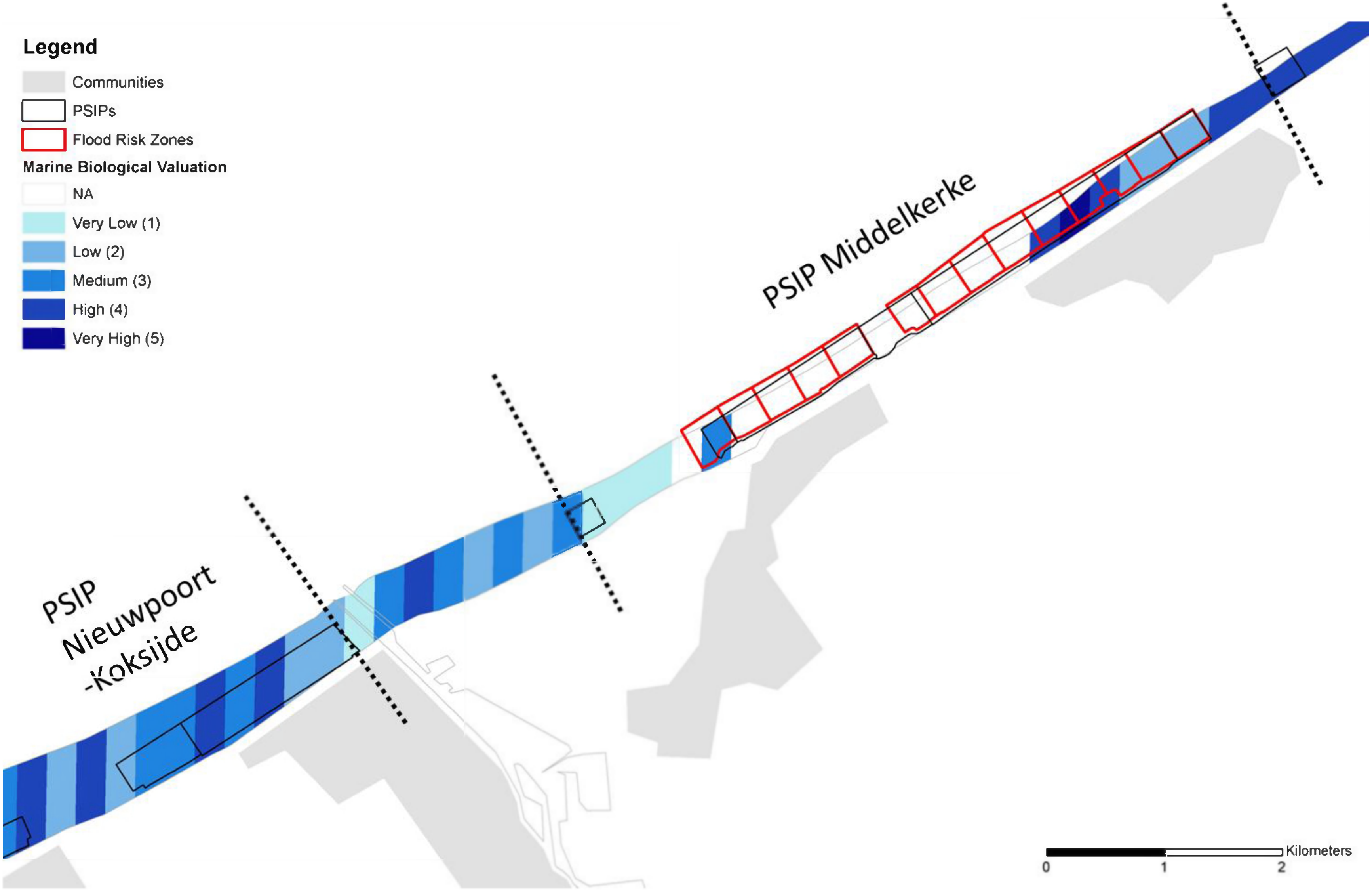


Figure G2: Detailed map with Marine Biological Valuation scores of intertidal subzones inside the PSIPs of Nieuwpoort-Koksijde and Middelkerke. Red indicates areas under coastal flood risk. The dashed lines mark the boundaries of each PSIP.

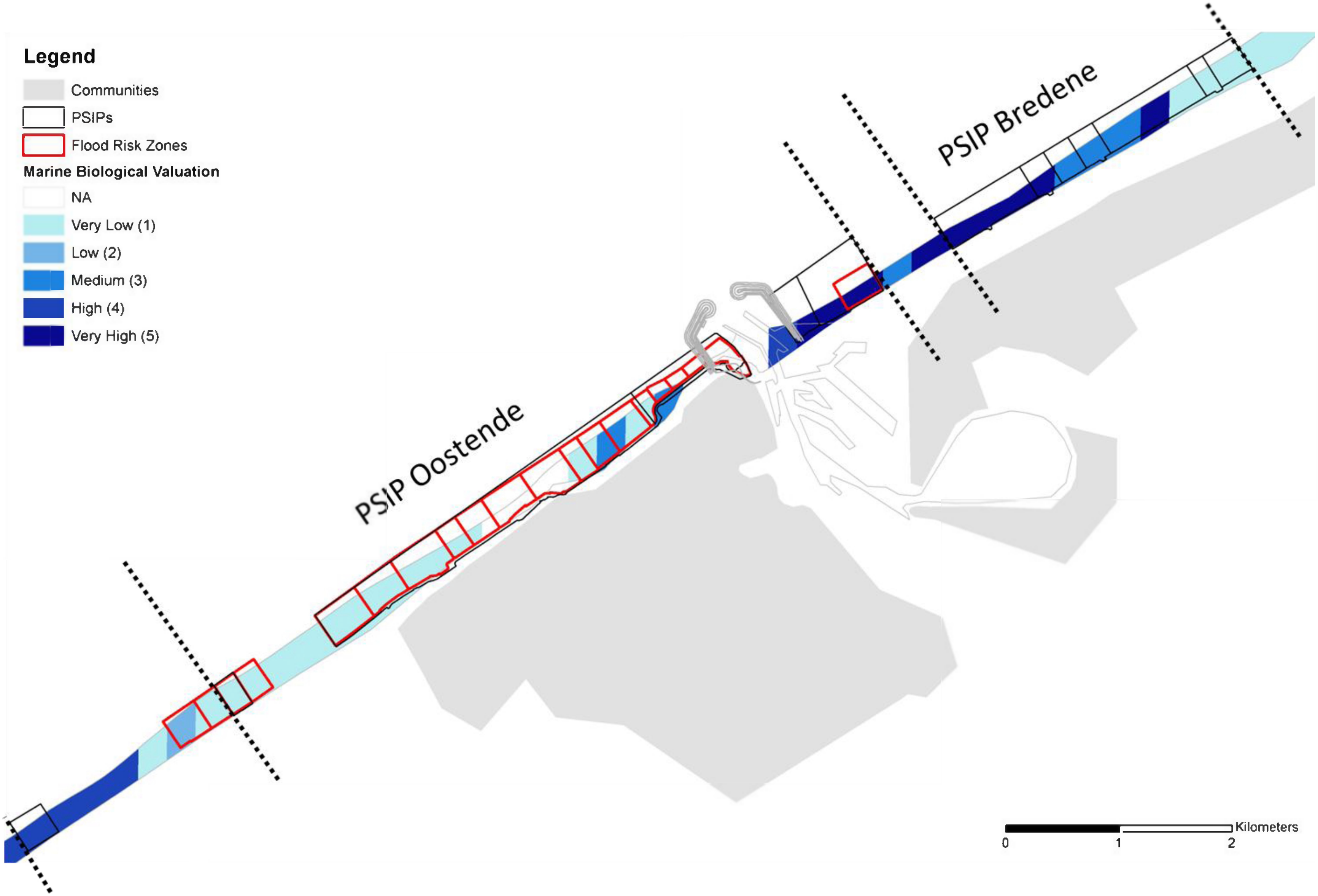


Figure G3: Detailed map with Marine Biological Valuation scores of intertidal subzones inside the PSIPs of Middelkerke, Oostende and Bredene. Red indicates areas under coastal flood risk. The dashed lines mark the boundaries of each PSIP.

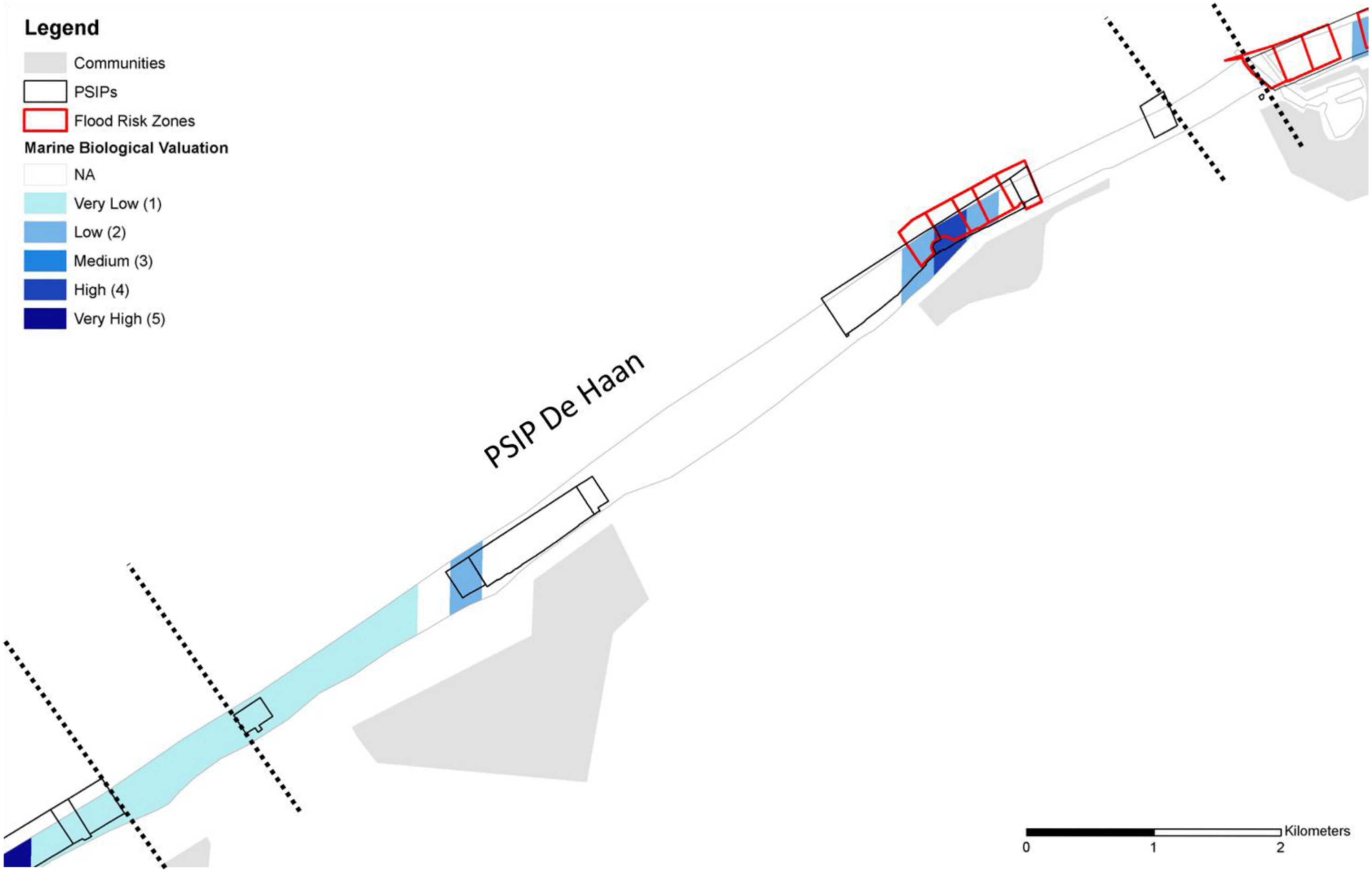


Figure G4: Detailed map with Marine Biological Valuation scores of intertidal subzones inside the PSIP of De Haan. Red indicates areas under coastal flood risk. The dashed lines mark the boundaries of each PSIP.

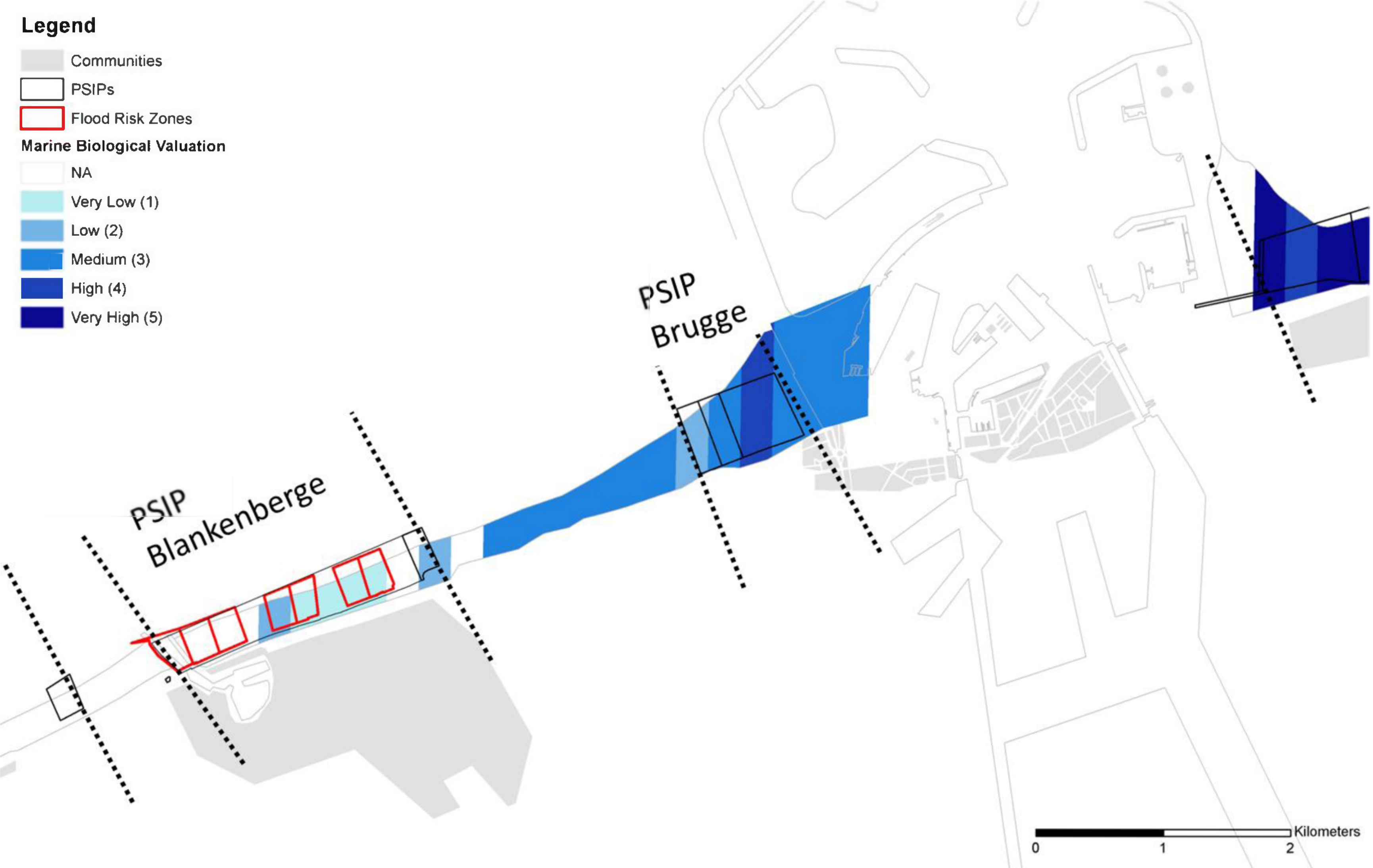


Figure G5: Detailed map with Marine Biological Valuation scores of intertidal subzones inside the PSIPs of Blankenberge and Brugge. Red indicates areas under coastal flood risk. The dashed lines mark the boundaries of each PSIP.

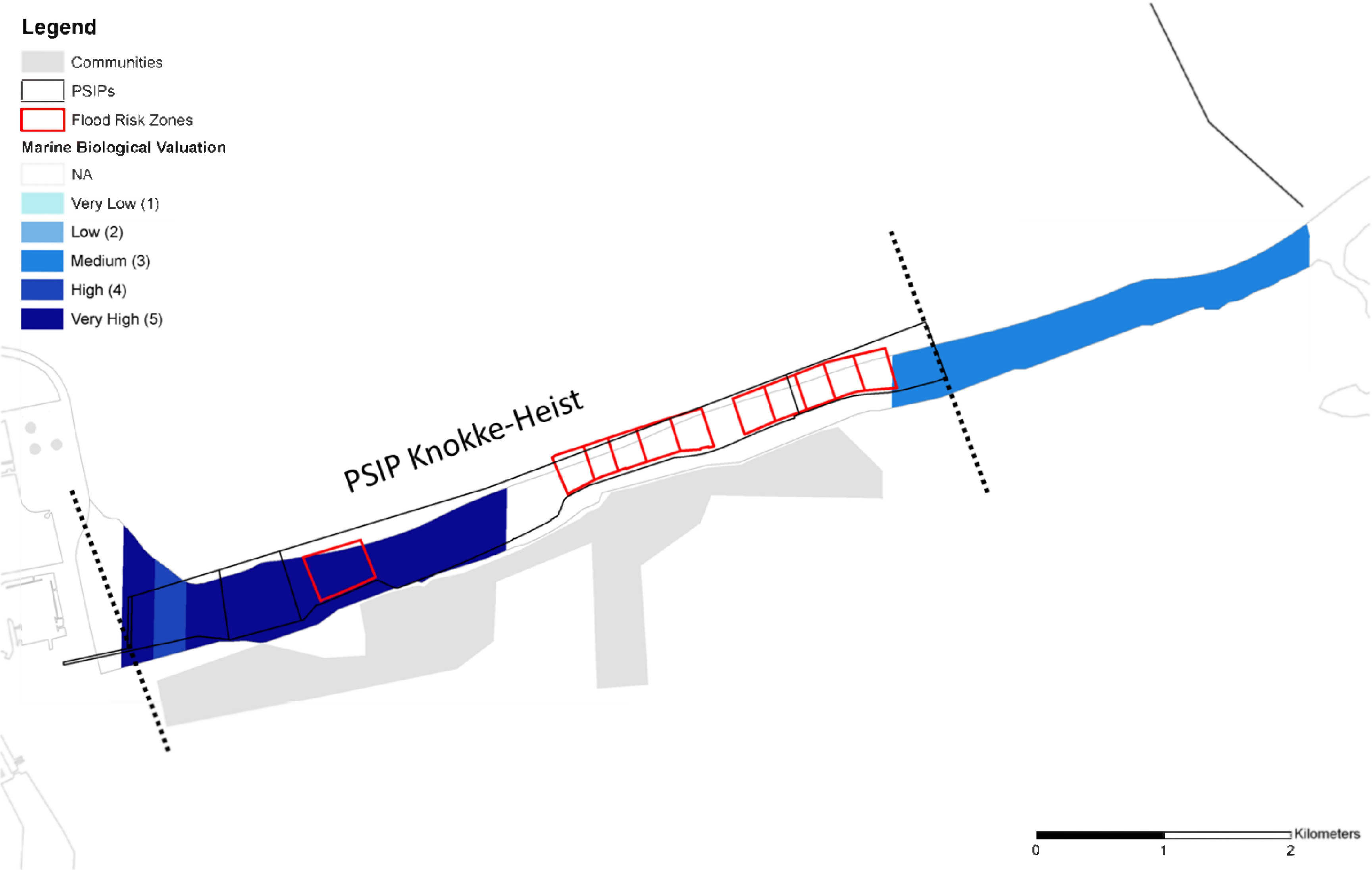


Figure G6: Detailed map with Marine Biological Valuation scores of intertidal subzones inside the PSIP of Knokke-Heist. Red indicates areas under coastal flood risk. The dashed lines mark the boundaries of the PSIP while NL indicates the Dutch territory.

Appendices – Chapter 7

Based on the data gathered during the period 1997 – 2011 in the intertidal and shallow subtidal zones, 16 beach fact sheets have been created for the beaches depicted in figure A.

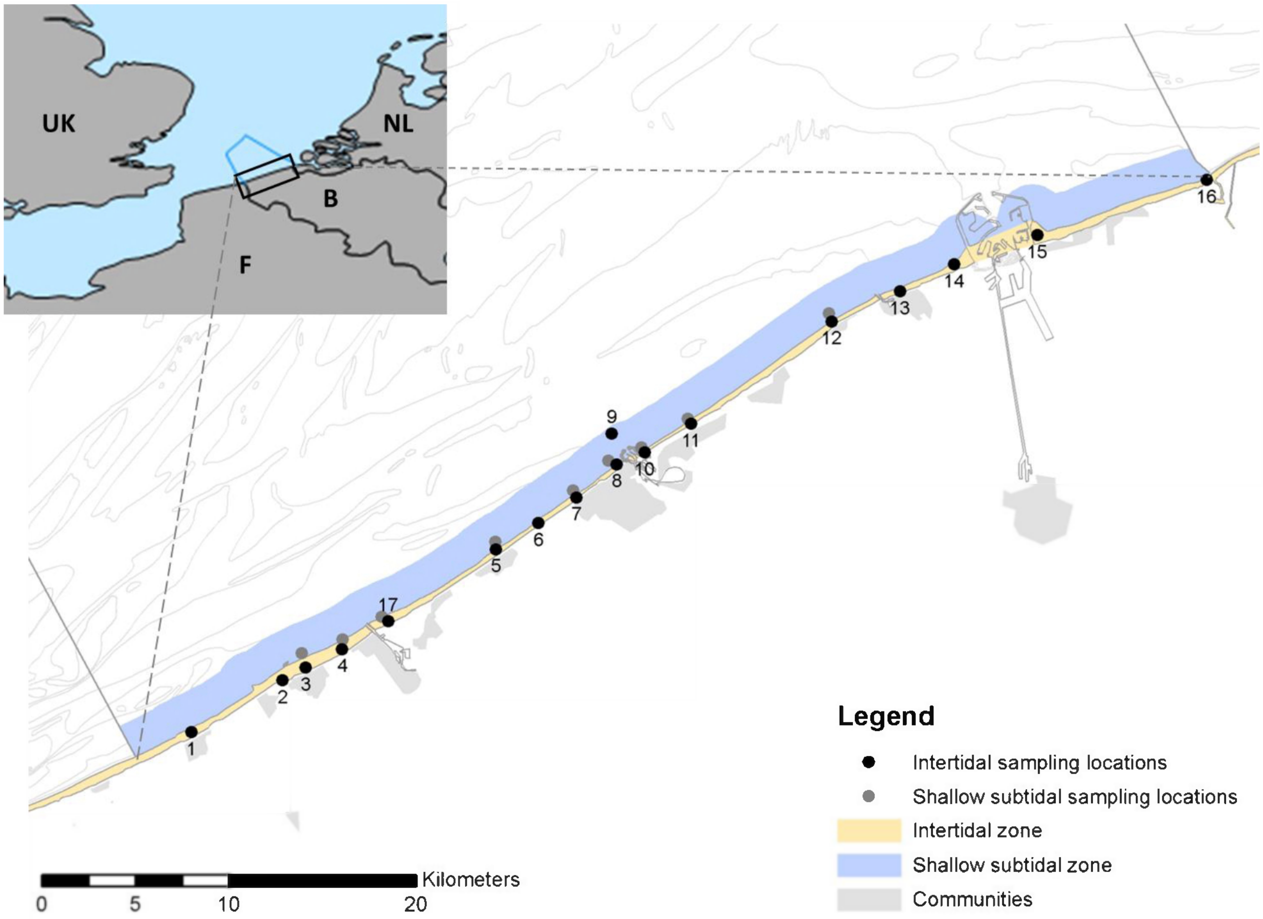


Figure A: The Belgian coastal zone, with a distinction between the intertidal (light brown; between mean high water level (MHW) and mean low water level (MLW)) and shallow subtidal zone (blue), showing the intertidal (black dots) and shallow subtidal (grey dots) sampling locations: (1) De Panne, (2) Schipgatduinen, (3) Koksijde-Oostduinkerke, (4) Nieuwpoort, (5) Middelkerke, (6) Raversijde, (7) Mariakerke, (8) Oostende-Center, (9) Oostende-Fairway, (10) Oostende-East, (11) Bredene, (12) Wenduine, (13) Blankenberge, (14) Fonteintjes, (15) Heist and (16) Zwinduinen en Polders, (17) Lombardsijde

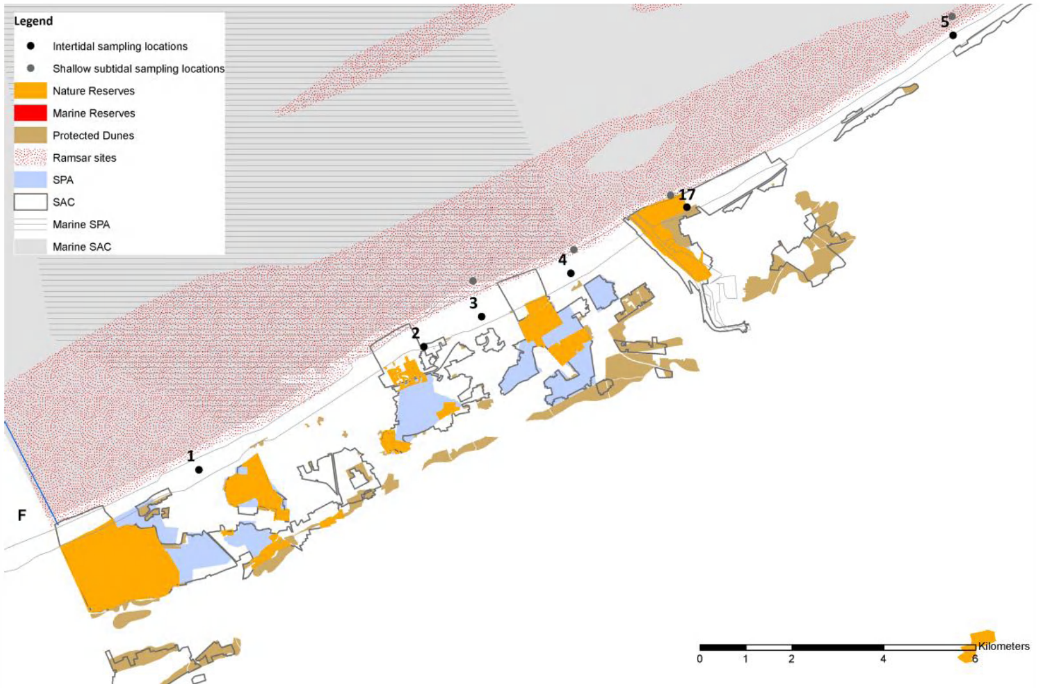
Appendices – Chapter 7

De Panne

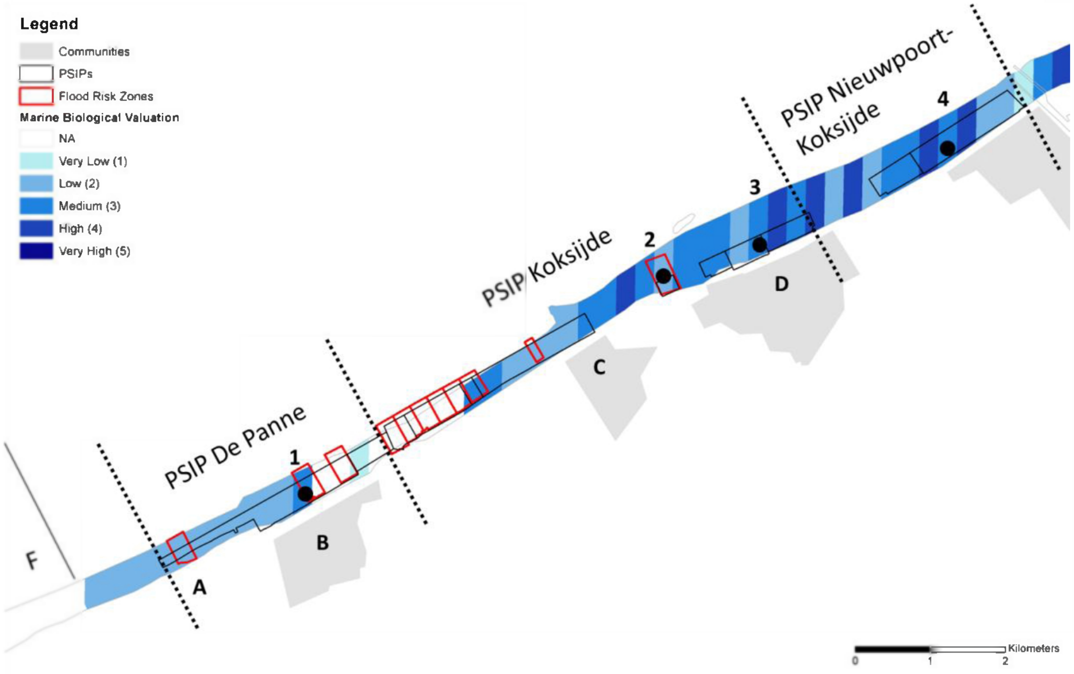
The beach of De Panne is situated at the western part of the Belgian coast (location 1 in all figures). Between this beach and the French border lies ‘De Westhoek’ which is part of the 640 ha nature reserve, SPA (Special Protection Area) and SAC (Special Area of Conservation) ‘Duinen en Bossen van de Panne’ (a). The Provincial Spatial Implementation Plan (PSIP) of De Panne covers the intertidal zone of the beach of De Panne (b).

According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of De Panne has a low to medium biological value (b). Because of the high coastal flood risk in the PSIP of De Panne, the beach at location A (b) received a dune nourishment (22 000 m³ sand) in October 2011 while a beach nourishment (40 000 m³ sand – location B in b) was performed in front of the entire municipality of De Panne at the same time.

All the following findings (c, e and f) are based on 28 intertidal samples taken in autumn 1997, right in front of the municipality of De Panne (Speybroeck et al. 2005). In total, 17 species were identified (d). All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches (c). The median grain size decreased from MHW (250 µm) to MLW (170 µm) (e) and *Bathyporeia pilosa* was



a



b

clearly the most dominant species in the intertidal zone of De Panne.

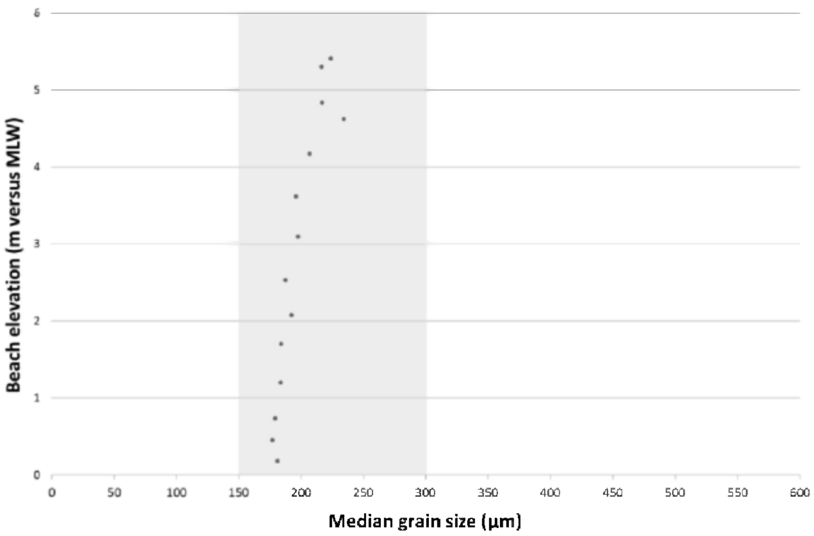
c

De Panne	INTERTIDAL			SHALLOW SUBTIDAL		
	1997	minimum	maximum	1997	minimum	maximum
Beach elevation (m versus MLW)	2.85 ± 0.34	0.00	5.41	-	-10.00	2.67
Median grain size (µm)	198.51 ± 3.40	175.00	464.00	-	0.00	319.73
Silt fraction (%)	0.00 ± 0.00	0.00	2.25	-	0.00	89.30
Total organic matter (%)	0.61 ± 0.02	0.00	1.84	-	0.00	11.75
Total organic carbon (%)	0.00 ± 0.00	0.00	19.00	-	0.00	4.85
Carbonate content (%)	0.58 ± 0.09	0.00	36.68	-	0.00	36.10
Species richness (number of species)	4.96 ± 0.36	0.00	19.00	-	0.00	28.00
Abundance (number of individuals.m^{-2})	455.70 ± 162.21	0.00	3988.75	-	0.00	1949.32
Biomass (g AFDW.m^{-2})	0.00 ± 0.00	0.00	6.95	-	0.00	246.14

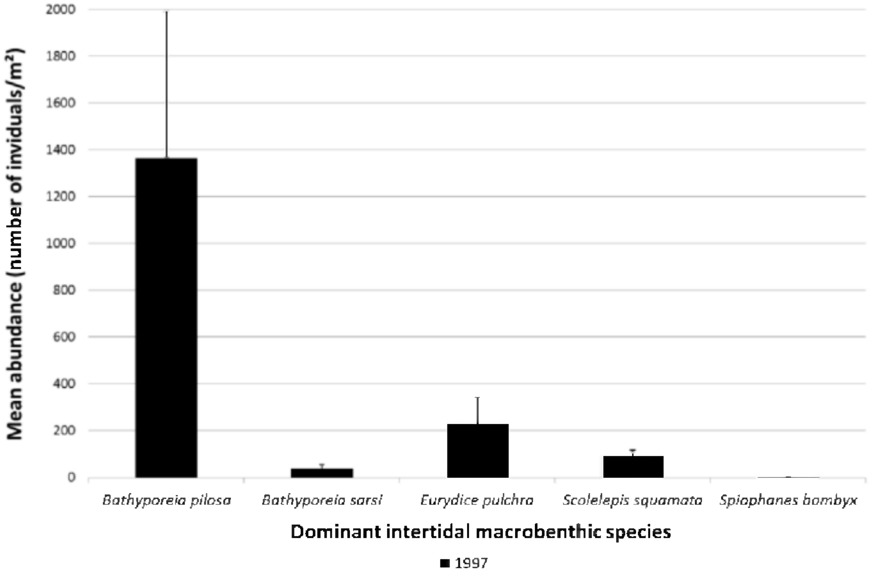
d

Species list De Panne	
Bathyporeia pilosa	*Nephtys cirrosa*
Bathyporeia sarsi	*Nephtys hombergii*
Capitella capitata	*Pontocrates altamarinus*
Cumopsis goodsir	*Pygospio elegans*
Eteone longa	*Scolelepis squamata*
Eurydice affinis	*Spiophanes bombyx*
Eurydice pulchra	*Spio species*
Macoma balthica	*Urothoe poseidonis*
Magelona species	

e



f

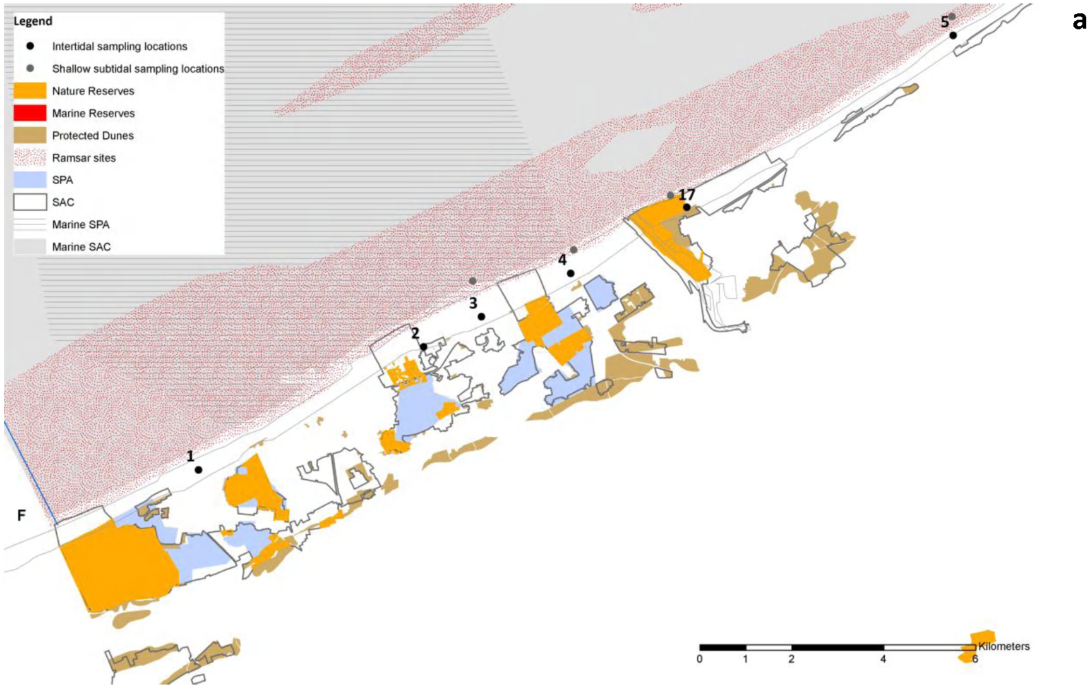


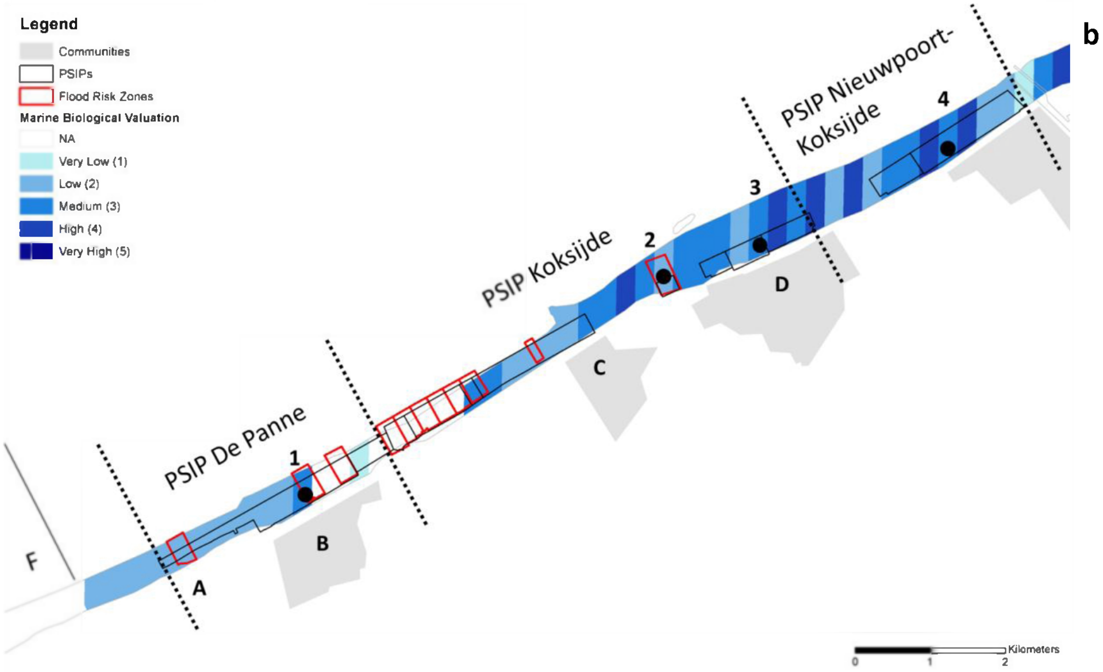
Schipgatduinen

The beach of Schipgatduinen is situated in a 45 ha nature reserve at the western part of the Belgian coast (location 2 in all figures). This dune area is also a SAC (Special Area of Conservation) (a). The Provincial Spatial Implementation Plan (PSIP) of Koksijde covers the intertidal zone of this beach (b).

According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Schipgatduinen has a low biological value (b). Due to the coastal flood risk at the beach of Schipgatduinen, the dune passage (location D in b) was elevated by 1 meter by means of a dune nourishment (3 000 m³), between January and April 2013.

All the following findings (c, e and f) are based on 22 intertidal samples taken in autumn 1997 (Speybroeck et al. 2005). In total, 17 species were identified (d). All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches (c). The median grain size decreased from MHW (464 µm) to MLW (182.90 µm) (e) but the 4 highest samples (taken closest to the MHW) contained much coarser sediment (larger than 300 µm) than usually found on Belgian beaches. *Scolelepis squamata* was clearly the most dominant species in the intertidal zone of Schipgatduinen.





c

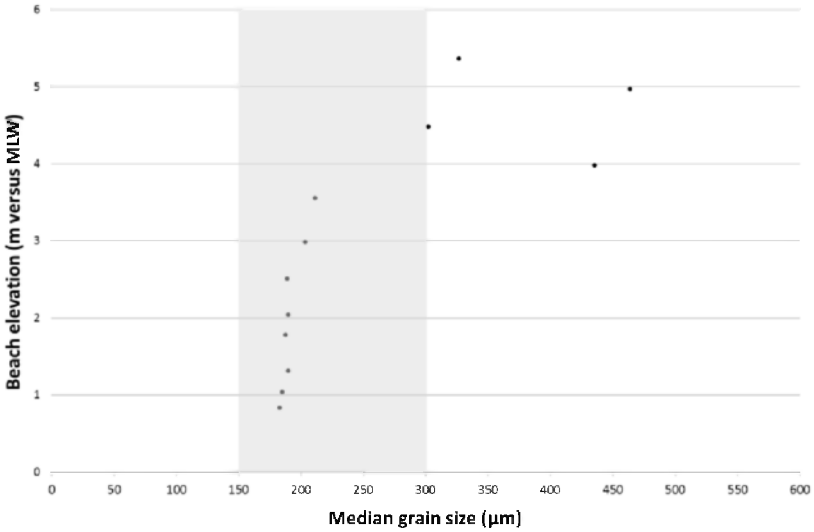
Schipgatduinen	INTERTIDAL			SHALLOW SUBTIDAL		
	1997	minimum	maximum	1997	minimum	maximum
Beach elevation (m versus MLW)	2.70 ± 0.30	0.00	5.41	-	-10.00	2.67
Median grain size (µm)	242.96 ± 19.70	175.00	464.00	-	0.00	319.73
Silt fraction (%)	0.00 ± 0.00	0.00	2.25	-	0.00	89.30
Total organic matter (%)	0.74 ± 0.03	0.00	1.84	-	0.00	11.75
Total organic carbon (%)	0.00 ± 0.00	0.00	19.00	-	0.00	4.85
Carbonate content (%)	0.47 ± 0.16	0.00	36.68	-	0.00	36.10
Species richness (number of species)	4.64 ± 0.35	0.00	19.00	-	0.00	28.00
Abundance (number of individuals.m^{-2})	63.21 ± 13.32	0.00	3988.75	-	0.00	1949.32
Biomass (g AFDW.m^{-2})	0.00 ± 0.00	0.00	6.95	-	0.00	246.14

d

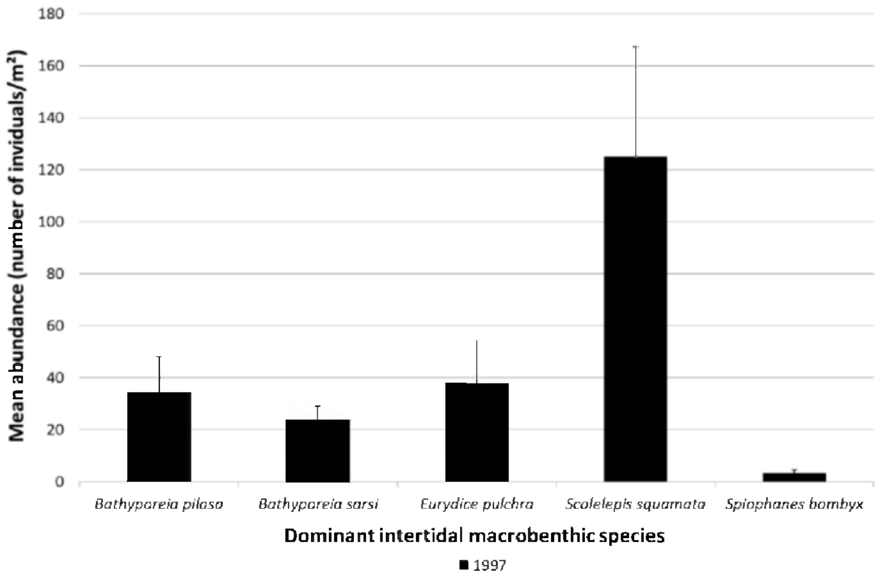
Species list Schipgatduinen

Bathyporeia pelagica
Bathyporeia pilosa
Bathyporeia sarsi
Capitella capitata
Cumopsis goodsir
Eteone longa
Eurydice affinis
Eurydice pulchra
Haustorius arenarius
Nephtys cirrosa
Nephtys hombergii
Pontocrates altamarinus
Pygospio elegans
Scolelepis squamata
Spiophanes bombyx
Spio species
Urothoe poseidonis

e



f



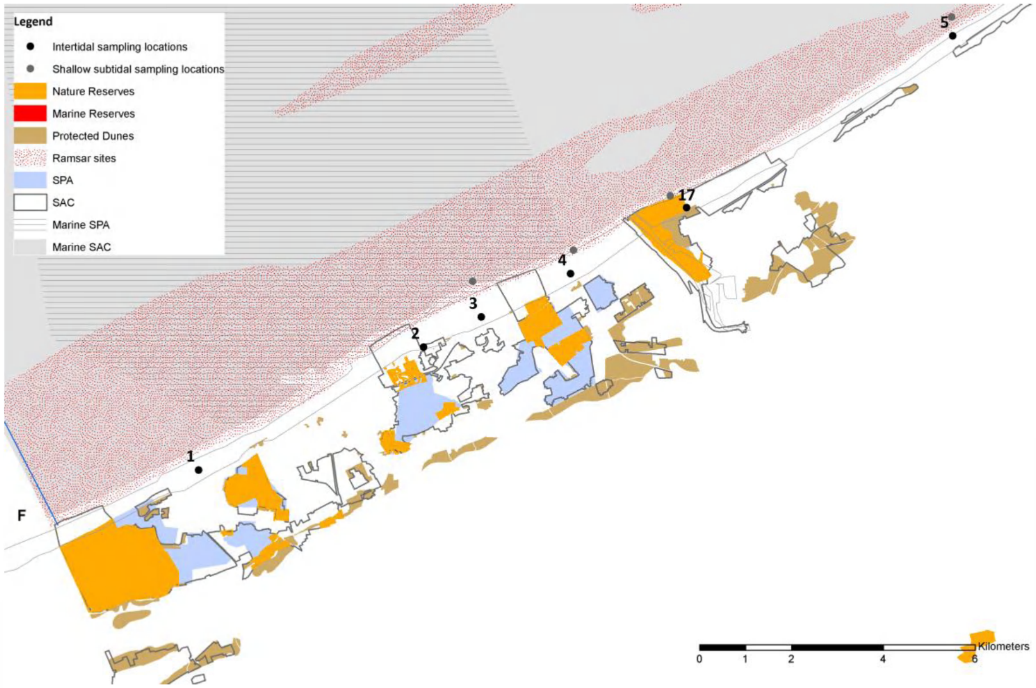
Koksijde-Oostduinkerke

The beach of Koksijde-Oostduinkerke is situated in front of the municipality of Oostduinkerke at the western part of the Belgian coast (location 3 in all figures). Although this beach is surrounded by nature reserves, SPAs (Special Protection Areas) and SACs (Special Areas of Conservation), it does not hold any conservation status (a). The Provincial Spatial Implementation Plan (PSIP) of Koksijde covers the intertidal zone of this beach (b).

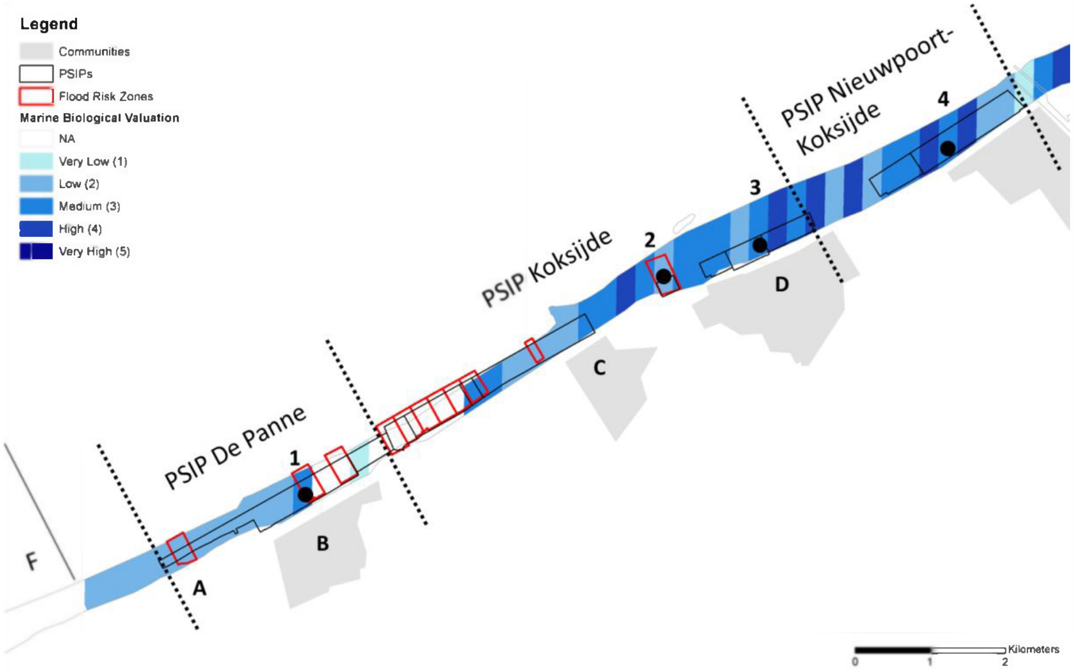
According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Koksijde-Oostduinkerke has a low to medium biological value (b). Due to the coastal flood risk in the PSIP of Koksijde, the beach in front of the entire municipality of Koksijde (location C in b) and the beaches in all the coastal flood risk priority areas west of Koksijde received a beach nourishment (140 000 m³) in October 2011. No coastal defence activities were planned at the beach of Koksijde-Oostduinkerke.

All the following findings (c, d, f, g, h and i) are based on 41 intertidal and 40 shallow subtidal samples taken during summer 2004 and autumn 2006 (Speybroeck et al. 2003; Welvaert et al. 2005):

Koksijde-Oostduinkerke	2004	2006
Intertidal zone	15	26
Shallow subtidal zone	15	25

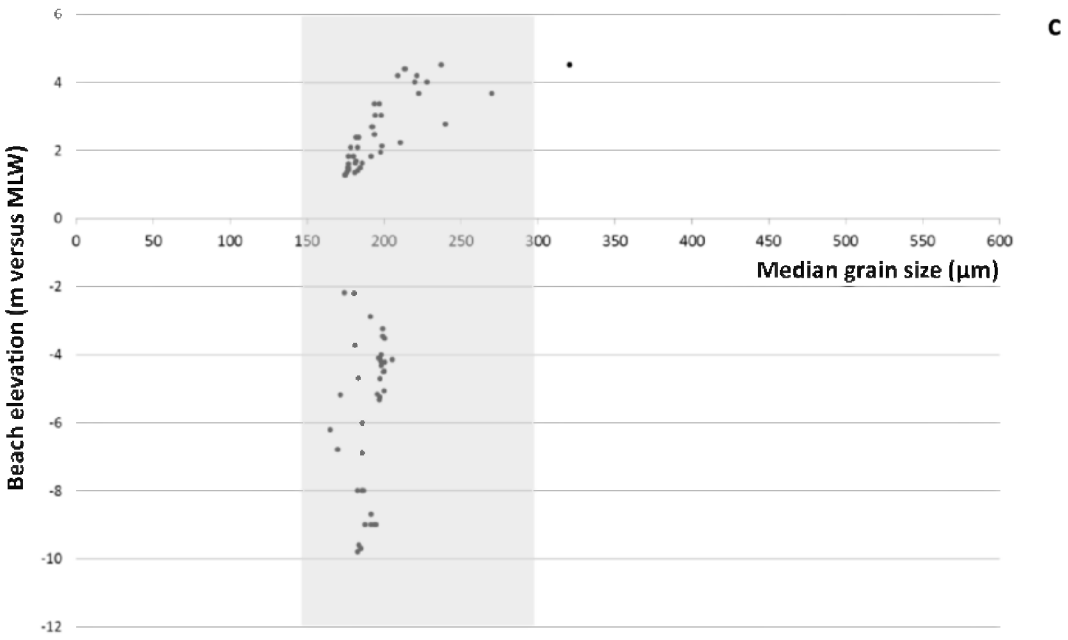


a



b

In total, 27 intertidal and 46 shallow subtidal species were identified (e). All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches (d). The median grain size decreased from MHW (320.54 µm, only value higher than 300 µm) to MLW (175 µm) while the shallow subtidal values fluctuated between 150 and 200 µm (c). The intertidal median grain size and abundance were always higher than the shallow subtidal values, both in 2004 and 2006 (f and g). Moreover, the values in 2006 were higher than the values in 2004.



In the intertidal zone, *Bathyporeia pilosa* was clearly the most dominant species in 2006. However, *Scolelepis squamata* was the only dominant intertidal species appearing in both studied years. In the shallow subtidal zone of Koksijde-Oostduinkerke, *Spio* species dominated in 2004 although *Lanice conchilega* is the only shallow subtidal dominant species appearing in both studied years. *Cirratulidae* species, *Ensis* juveniles and *Macoma balthica* juveniles were not present in the shallow subtidal samples of Koksijde-Oostduinkerke.

d

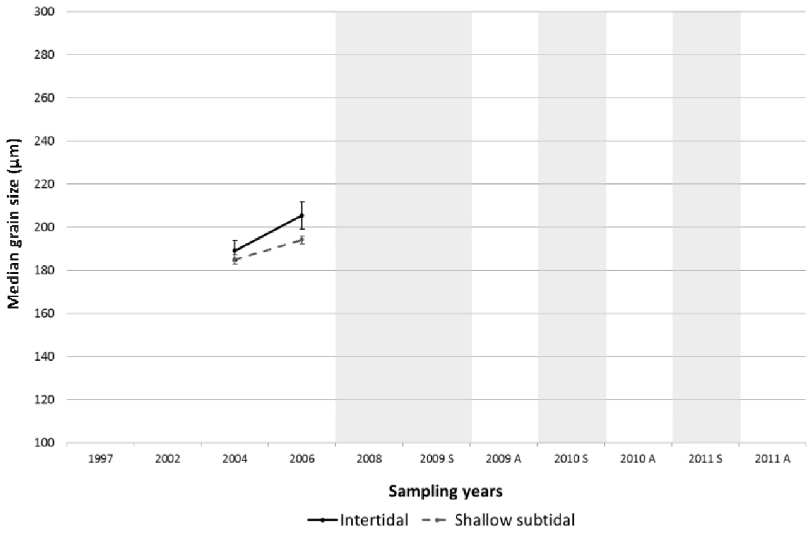
Koksijde-Oostduinkerke	**INTERTIDAL**				**SHALLOW SUBTIDAL**			
	2004	2006	minimum	maximum	2004	2006	minimum	maximum
Beach elevation (m versus MLW)	1.75 ± 0.12	3.02 ± 0.21	0.00	5.41	-8.25 ± 0.32	-4.12 ± 0.17	-10.00	2.67
Median grain size (µm)	189.13 ± 4.60	205.41 ± 6.38	175.00	464.00	185.07 ± 2.10	194.18 ± 1.79	0.00	319.73
Silt fraction (%)	0.007 ± 0.007	0.001 ± 0.00	0.00	2.25	0.19 ± 0.13	3.80 ± 1.36	0.00	89.30
Total organic matter (%)	0.15 ± 0.01	0.00 ± 0.00	0.00	1.84	0.24 ± 0.04	0.00 ± 0.00	0.00	11.75
Total organic carbon (%)	0.15 ± 0.02	1.43 ± 0.08	0.00	19.00	0.20 ± 0.03	1.25 ± 0.03	0.00	4.85
Carbonate content (%)	12.37 ± 0.77	12.54 ± 0.51	0.00	36.68	10.02 ± 0.40	10.92 ± 0.21	0.00	36.10
Species richness (number of species)	4.67 ± 0.29	3.73 ± 0.26	0.00	19.00	6.60 ± 0.70	8.13 ± 0.53	0.00	28.00
Abundance (number of individuals.m^{-2})	49.09 ± 10.58	146.92 ± 31.04	0.00	3988.75	25.76 ± 2.21	58.91 ± 8.75	0.00	1949.32
Biomass (g AFDW.m^{-2})	0.53 ± 0.15	0.63 ± 0.15	0.00	6.95	1.76 ± 0.36	8.72 ± 4.43	0.00	246.14

e

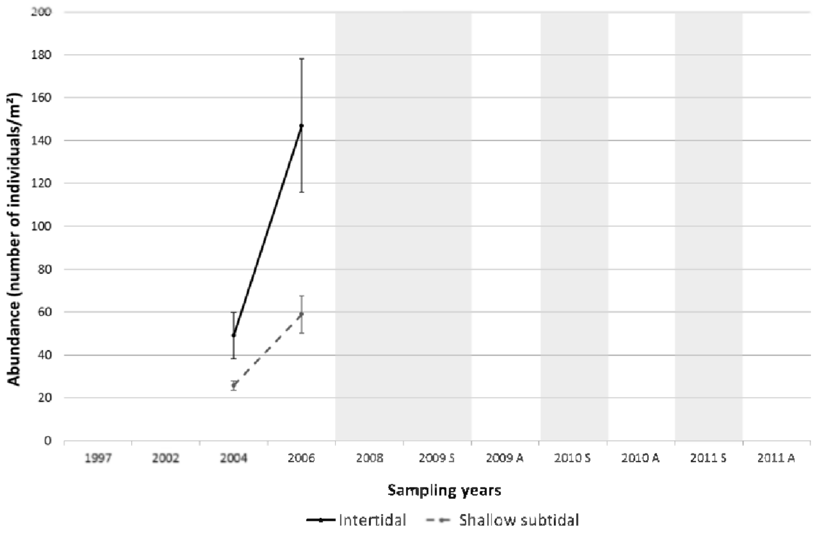
Species list Koksijde-Oostduinkerke

Intertidal		Shallow subtidal		
Abra alba	*Macoma balthica*	*Abra alba*	*Eteone longa*	*Nephtys species*
Bathyporeia elegans	*Magelona species*	*Angulus fabula*	*Gammarus species*	*Oligochaeta species*
Bathyporeia pilosa	*Nemertea species*	*Angulus tenuis*	*Glycera species*	*Owenia fusiformis*
Bathyporeia sarsi	*Nephtys assimilis*	*Aphelochaeta marioni*	*Haustorius arenarius*	*Pariambus typicus*
Cerastoderma edule	*Nephtys cirrosa*	*Atylus falcatus*	*Jassa falcata*	*Pontocrates altamarinus*
Corophium species	*Nephtys hombergii*	*Atylus swammerdami*	*Lagis koreni*	*Pontocrates arenarius*
Cumopsis goodsir	*Nephtys juveniles*	*Autolytus species*	*Lanice conchilega*	*Sagitta species*
Donax vittatus	*Pontocrates altamarinus*	*Bathyporeia elegans*	*Macoma balthica*	*Scoloplos armiger*
Eteone flava	*Scoloplos armiger*	*Bathyporeia pelagica*	*Magelona species*	*Scolelepis squamata*
Eteone longa	*Scolelepis squamata*	*Bathyporeia sarsi*	*Microprotopus maculatus*	*Sigalion mathildae*
Eurydice affinis	*Spiophanes bombyx*	*Capitella capitata*	*Mysella bidentata*	*Spiophanes bombyx*
Eurydice pulchra	*Spio species*	*Cumopsis goodsir*	*Nemertea species*	*Spio species*
Gammarus species	*Urothoe poseidonis*	*Diogenes pugilator*	*Nephtys assimilis*	*Spisula subtruncata*
Lanice conchilega		*Donax vittatus*	*Nephtys cirrosa*	*Urothoe poseidonis*
		Ensis directus	*Nephtys hombergii*	
		Eteone flava	*Nephtys juveniles*	

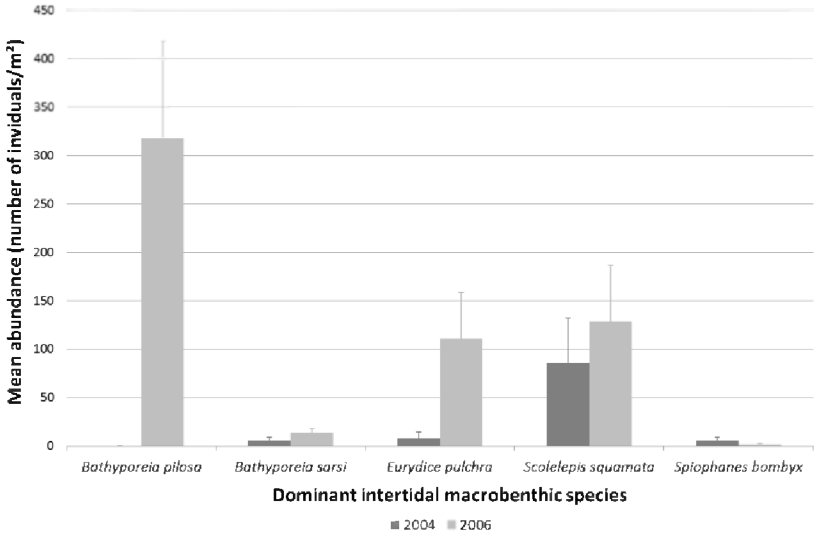
f


300
280
260
240
220
200
180
160
140
120
100
Median grain size (µm)
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal

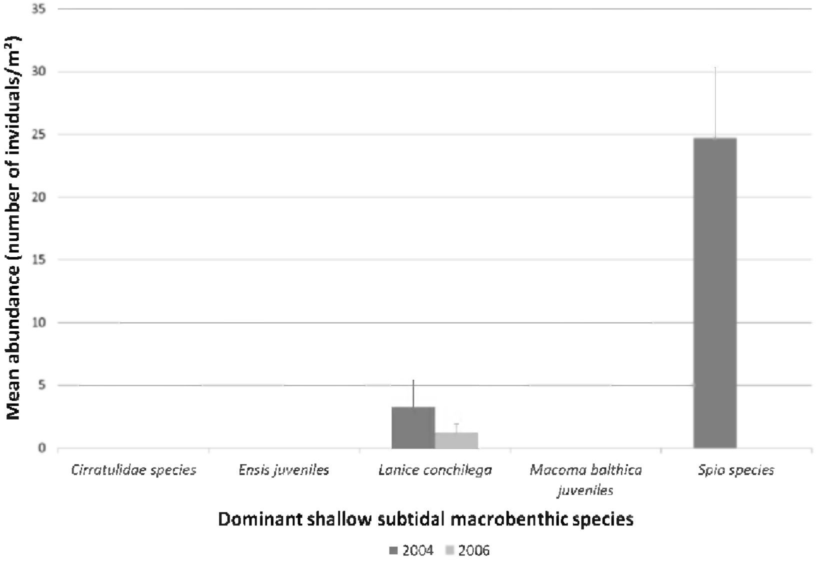
g


200
180
160
140
120
100
80
60
40
20
0
Abundance (number of individuals/m²)
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal

h


450
400
350
300
250
200
150
100
50
0
Mean abundance (number of invividuals/m²)
Bathyporeia pilosa
Bathyporeia sarsi
Eurydice pulchra
Scolelepis squamata
Spiophanes bombyx
Dominant intertidal macrobenthic species
2004
2006

i


35
30
25
20
15
10
5
0
Mean abundance (number of invividuals/m²)
Cirratulidae species
Ensis juveniles
Lanice conchilega
Macoma balthica juveniles
Spio species
Dominant shallow subtidal macrobenthic species
2004
2006

Nieuwpoort

The beach of Nieuwpoort is situated in front of the municipality of Nieuwpoort at the western side of the fairway to Nieuwpoort (location 4 in all figures). It does not hold any conservation status (a). The Provincial Spatial Implementation Plan (PSIP) of Nieuwpoort-Koksijde covers the intertidal zone of this beach (b).

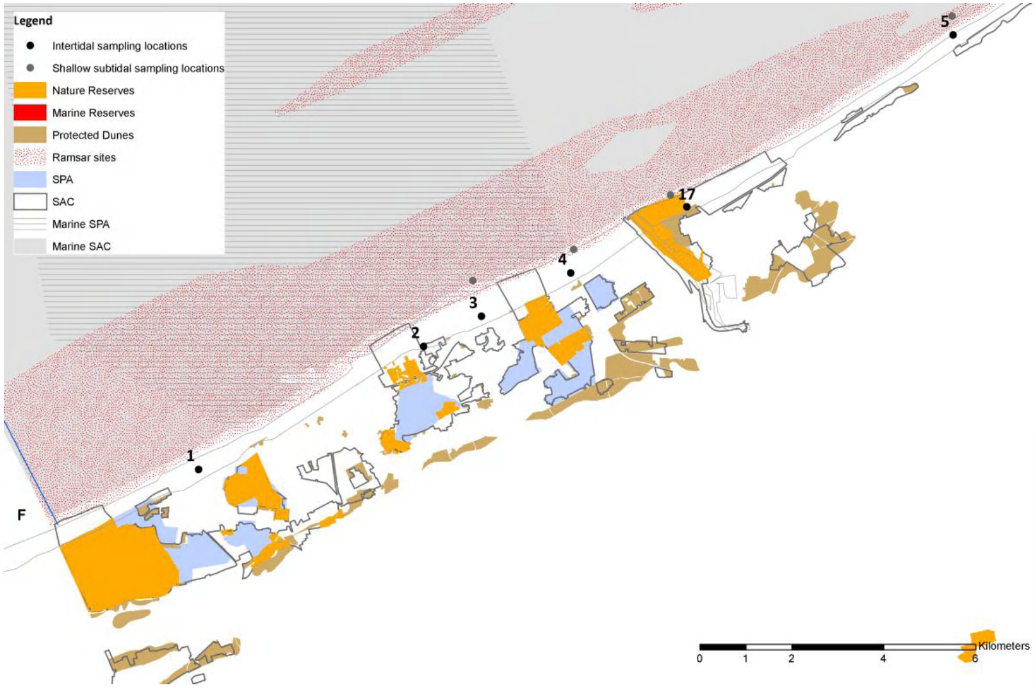
According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Nieuwpoort has a medium to high biological value (b). There is no coastal flood risk at the beach of Nieuwpoort and only maintenance beach nourishments high up on the beach are regularly planned to maintain the width of the touristic dry beach zone.

All the following findings (c, d, f, g, h and i) are based on 138 intertidal and 135 shallow subtidal samples taken in 2008 and in spring (S) and autumn (A) of 2009, 2010 and 2011 (Vanden Eede et al. 2008; 2010; 2011 and 2013):

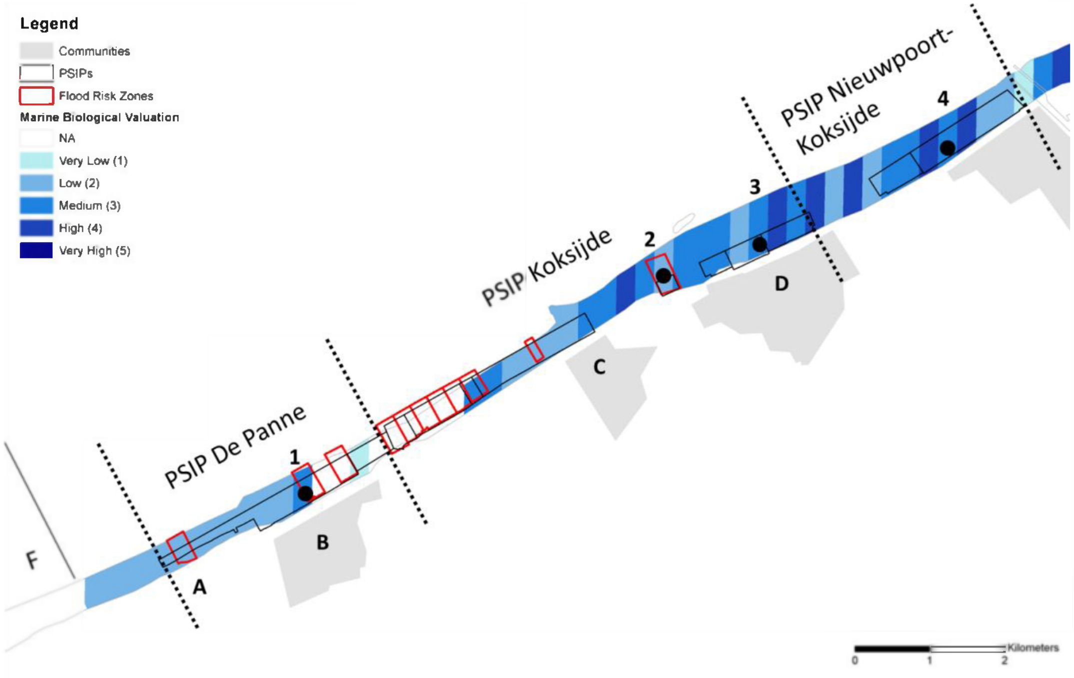
Nieuwpoort	**2008**	**2009**		**2010**		**2011**	
	S	S	A	S	A	S	A
Intertidal zone	26	26	26	15	15	15	15
Shallow subtidal zone	25	25	25	15	15	15	15

In total, 43 intertidal and 65 shallow subtidal species were identified (e). All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches (d). The median grain size

a

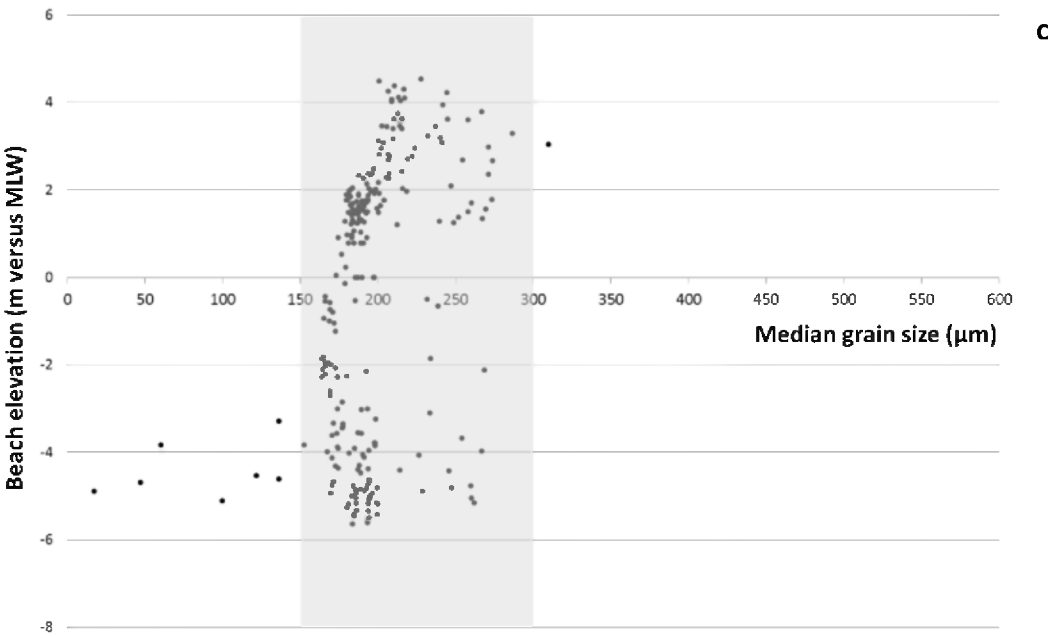


b



decreased from MHW (309.75 µm, only value higher than 300 µm) to MLW (173.21 µm) while the shallow subtidal values fluctuated between 150 and 250 µm (c). Seven very low values (< 150 µm) were detected in the shallow subtidal zone during autumn 2011. The intertidal median grain size was always higher than the shallow subtidal values. In autumn, the shallow subtidal abundance was always much higher than the intertidal abundance while in spring the values were quite similar. 2008 and autumn 2011 were the only exceptions. In these years, the intertidal values were higher than the shallow subtidal ones (f and g).

In the intertidal zone, *Bathyporeia pilosa* and *Bathyporeia sarsi* were clearly the most dominant species in all years (h), followed by *Scolelepis squamata*. *Spiophanes bombyx* was bearly found in the intertidal zone of Nieuwpoort. In the shallow subtidal zone, *Ensis* juveniles showed extreme peak values in autumn 2009 (6107.60 ± 978.77 individuals.m^{-2}) and autumn 2010 (5630.93 ± 1208.57 individuals.m^{-2}). These values are not shown in figure i to allow for a better interpretation of the remaining abundance values. In autumn 2010, *Cirratulidae* species and *Macoma balthica* juveniles were the most dominant species while *Spio* species dominated in autumn 2011. Only *Lanice conchilega* was found in 2008 but in very low abundance (1.17 ± 1.15 individual.m^{-2}).



d

Nieuwpoort	INTERTIDAL								
	2008	2009		2010		2011		minimum	maximum
Beach elevation (m versus MLW)	2.65 ± 0.22	1.89 ± 0.22	1.99 ± 0.14	2.49 ± 0.26	2.04 ± 0.20	2.12 ± 0.30	2.61 ± 0.23	0.00	5.41
Median grain size (µm)	206.35 ± 3.55	197.53 ± 3.02	196.24 ± 2.83	201.75 ± 5.32	259.69 ± 4.35	202.29 ± 5.63	206.81 ± 8.25	175.00	464.00
Silt fraction (%)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00	2.25
Total organic matter (%)	0.28 ± 0.01	0.65 ± 0.03	0.62 ± 0.02	0.70 ± 0.02	0.52 ± 0.03	0.58 ± 0.04	0.53 ± 0.03	0.00	1.84
Total organic carbon (%)	1.43 ± 0.05	1.34 ± 0.06	1.03 ± 0.11	0.27 ± 0.11	0.15 ± 0.05	0.17 ± 0.01	0.06 ± 0.01	0.00	19.00
Carbonate content (%)	11.90 ± 0.39	11.13 ± 0.52	12.21 ± 1.03	10.36 ± 0.83	11.69 ± 1.32	11.22 ± 1.01	9.77 ± 0.59	0.00	36.68
Species richness (number of species)	3.62 ± 0.36	4.67 ± 0.36	3.77 ± 0.36	5.40 ± 0.32	6.73 ± 0.34	6.00 ± 0.69	4.73 ± 0.47	0.00	19.00
Abundance (number of individuals.m^{-2})	178.38 ± 51.61	67.46 ± 12.30	92.76 ± 21.28	54.24 ± 10.68	133.18 ± 49.23	136.98 ± 52.83	101.99 ± 33.75	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.53 ± 0.14	0.59 ± 0.10	0.19 ± 0.06	0.67 ± 0.18	0.51 ± 0.07	0.71 ± 0.15	0.79 ± 0.16	0.00	6.95

	SHALLOW SUBTIDAL								
	2008	2009		2010		2011		minimum	maximum
Beach elevation (m versus MLW)	-2.63 ± 0.38	-2.43 ± 0.53	-2.33 ± 0.55	-3.49 ± 0.48	-3.57 ± 0.41	-3.75 ± 0.39	-3.44 ± 0.38	0.00	5.41
Median grain size (µm)	186.48 ± 2.01	182.33 ± 2.38	181.50 ± 2.03	182.32 ± 2.57	244.79 ± 4.31	137.51 ± 14.47	178.84 ± 4.30	175.00	464.00
Silt fraction (%)	0.00 ± 0.00	1.51 ± 1.44	0.75 ± 0.42	0.67 ± 0.47	2.18 ± 0.86	22.93 ± 6.80	2.85 ± 2.26	0.00	2.25
Total organic matter (%)	0.38 ± 0.04	1.37 ± 0.07	1.61 ± 0.25	2.10 ± 0.22	1.59 ± 0.12	4.96 ± 0.94	1.84 ± 0.35	0.00	1.84
Total organic carbon (%)	1.32 ± 0.01	1.10 ± 0.05	0.18 ± 0.02	0.29 ± 0.07	0.18 ± 0.03	0.74 ± 0.15	0.08 ± 0.05	0.00	19.00
Carbonate content (%)	11.04 ± 0.12	9.19 ± 0.39	9.92 ± 0.28	8.75 ± 0.52	8.93 ± 0.30	14.74 ± 1.19	9.11 ± 0.37	0.00	36.68
Species richness (number of species)	7.36 ± 0.53	6.88 ± 0.45	10.48 ± 0.91	8.60 ± 0.86	15.20 ± 1.21	12.60 ± 0.88	17.60 ± 1.31	0.00	19.00
Abundance (number of individuals.m^{-2})	67.37 ± 9.70	63.59 ± 9.34	560.49 ± 90.83	47.40 ± 7.64	516.61 ± 100.29	116.03 ± 19.86	174.07 ± 17.82	0.00	3988.75
Biomass (g AFDW.m^{-2})	3.71 ± 0.73	7.34 ± 3.01	6.08 ± 0.87	6.57 ± 1.18	59.56 ± 15.24	3.48 ± 6.35	15.08 ± 4.24	0.00	6.95

e

Species list Nieuwpoort

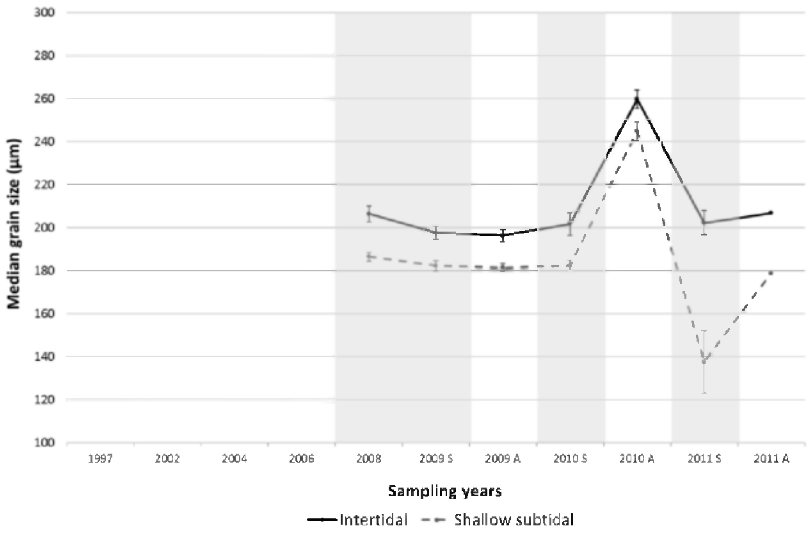
Intertidal

Abra alba
Arenicola marina
Bathyporeia pelagica
Bathyporeia pilosa
Bathyporeia sarsi
Capitella capitata
Cirratulidae species
Corophium species
Cumopsis goodsir
Diogenes pugilator
Donax vittatus
Ensis juveniles
Eteone flava
Eteone longa
Eurydice affinis
Eurydice pulchra
Glycera species
Heteromastus filıformis
Jassa herdmani
Lanice conchilega
Macoma balthica
Macoma balthica juveniles
Magelona species
Nemertea species
Nephtys caeca
Nephtys cirrosa
Nephtys hombergii
Nephtys juveniles
Oligochaeta species
Owenia fusiformis
Pariambus typicus
Phyllodoce species
Polydora species
Pseudocuma longicornis
Pygospio elegans
Scolelepis squamata
Spiophanes bombyx
Spionidae species
Spio species
Talitrus saltator
Urothoe brevicornis
Urothoe poseidonis
Urothoe pulchella

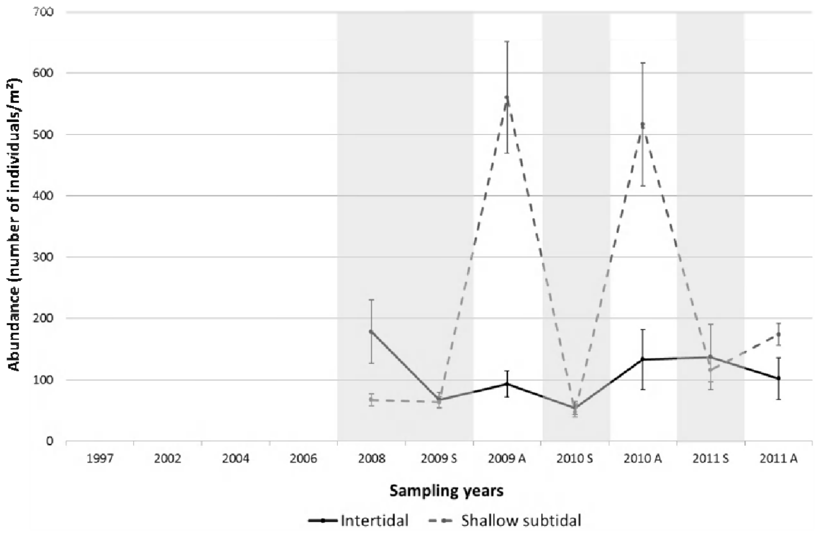
Shallow subtidal

Abra alba
Ampharetidae species
Angulus fabula
Angulus tenuis
Arenicola marina
Atylus swammerdami
Autolytus species
Bathyporeia pelagica
Bathyporeia pilosa
Bathyporeia sarsi
Bodotria pulchella
Capitella capitata
Cirratulidae species
Corophium species
Cumopsis goodsir
Diastylis bradyi
Diastylis rathkei
Diogenes pugilator
Donax vittatus
Donax vittatus juveniles
Ensis directus
Ensis juveniles
Ensis species
Eteone flava
Eteone longa
Eumida sanguinea
Eumida juveniles
Eurydice pulchra
Gammarus species
Glycera species
Haustorius arenarius
Heteromastus filıformis
Iphinoe trispinosa
Jassa falcata
Jassa species
Lagis koreni
Lanice conchilega
Macoma balthica
Macoma balthica juveniles
Magelona species
Nemertea species
Nephtys assimilis
Nephtys caeca
Nephtys cirrosa
Nephtys hombergii
Nereis longissima
Oligochaeta species
Owenia fusıformis
Pariambus typicus
Pholoe minuta
Phyllodoce species
Polydora species
Pontocrates altamarinus
Pontocrates arenarius
Pseudocuma longicorne
Pygospio elegans
Scoloplos armiger
Sigalion mathildae
Spiophanes bombyx
Spionidae species
Spio species
Spisula subtruncata
Urothoe brevicornis
Urothoe poseidonis
Urothoe pulchella

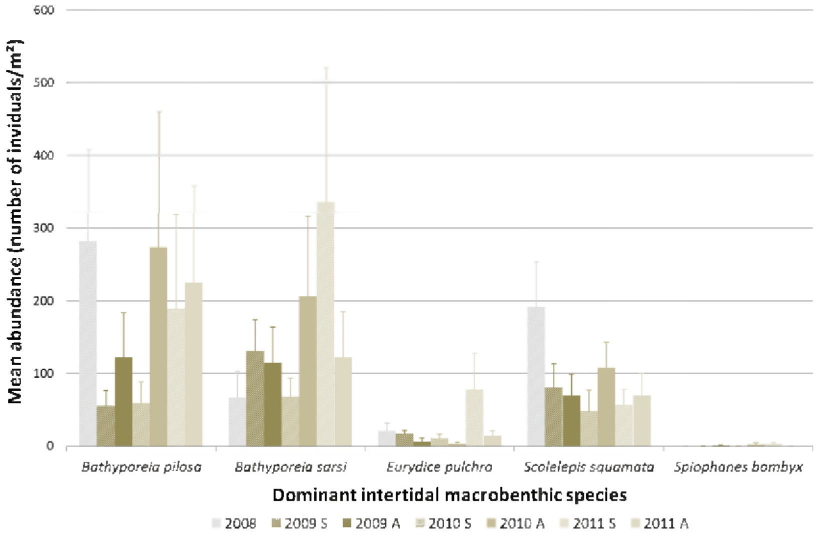
f


300
280
260
240
220
200
180
160
140
120
100
Median grain size (µm)
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal

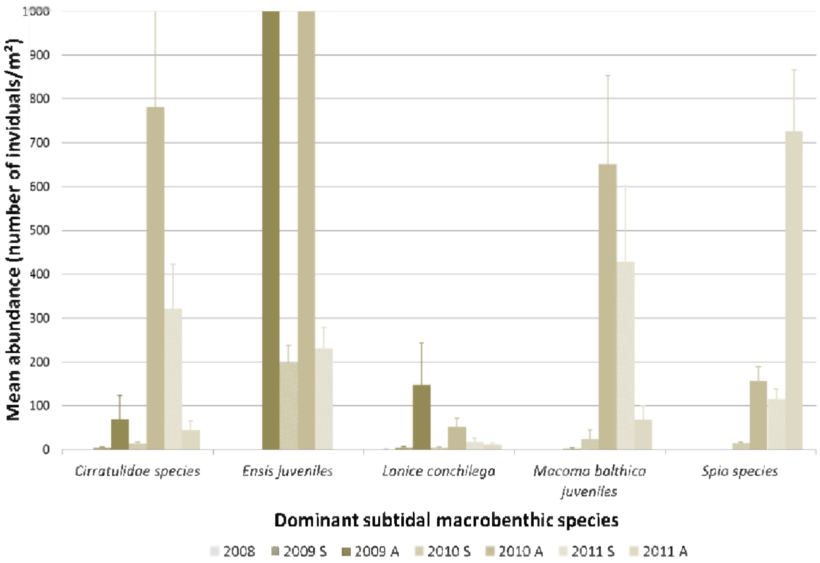
g


700
600
500
400
300
200
100
0
Abundance (number of individuals/m²)
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal

h


600
500
400
300
200
100
0
Mean abundance (number of invividuals/m²)
Bathyporeia pilosa
Bathyporeia sarsi
Eurydice pulchra
Scolelepis squamata
Spiophanes bombyx
Dominant intertidal macrobenthic species
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A

i


1000
900
800
700
600
500
400
300
200
100
0
Mean abundance (number of invividuals/m²)
Cirratulidae species
Ensis juveniles
Lanice conchilega
Macoma balthica juveniles
Spio species
Dominant subtidal macrobenthic species
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A

Middelkerke

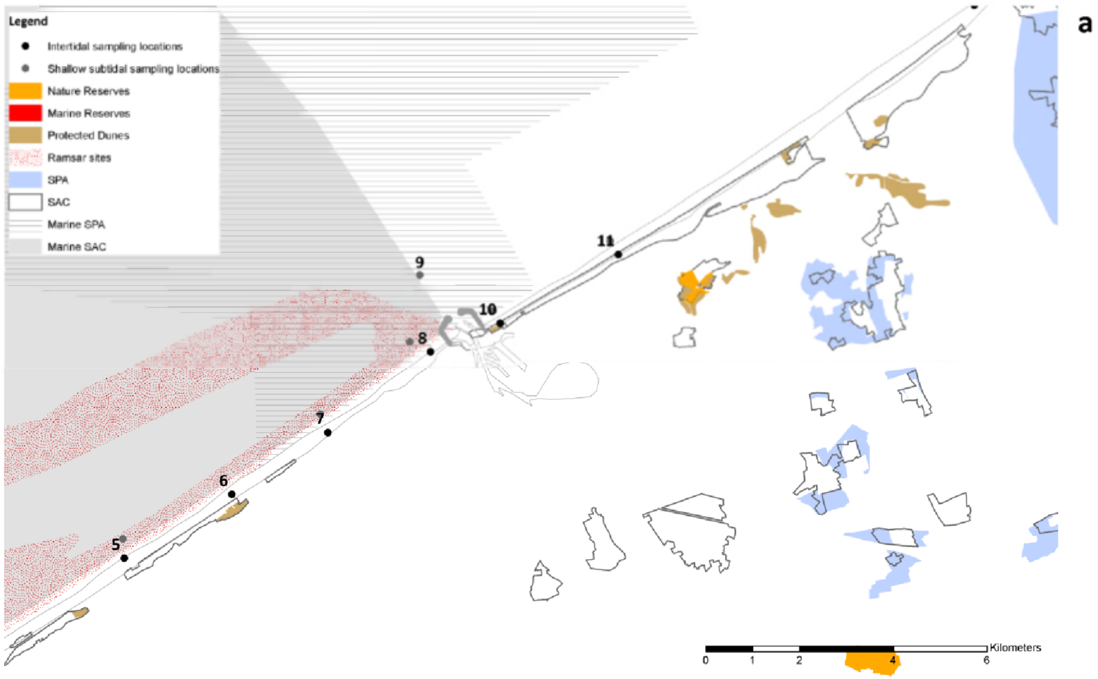
The beach of Middelkerke is situated in front of the municipality of Middelkerke (location 5 in all figures). It does not hold any conservation status but it is located in front of a Special Area of Conservation (SAC) (a). The Provincial Spatial Implementation Plan (PSIP) of Middelkerke covers the intertidal zone of this beach (b).

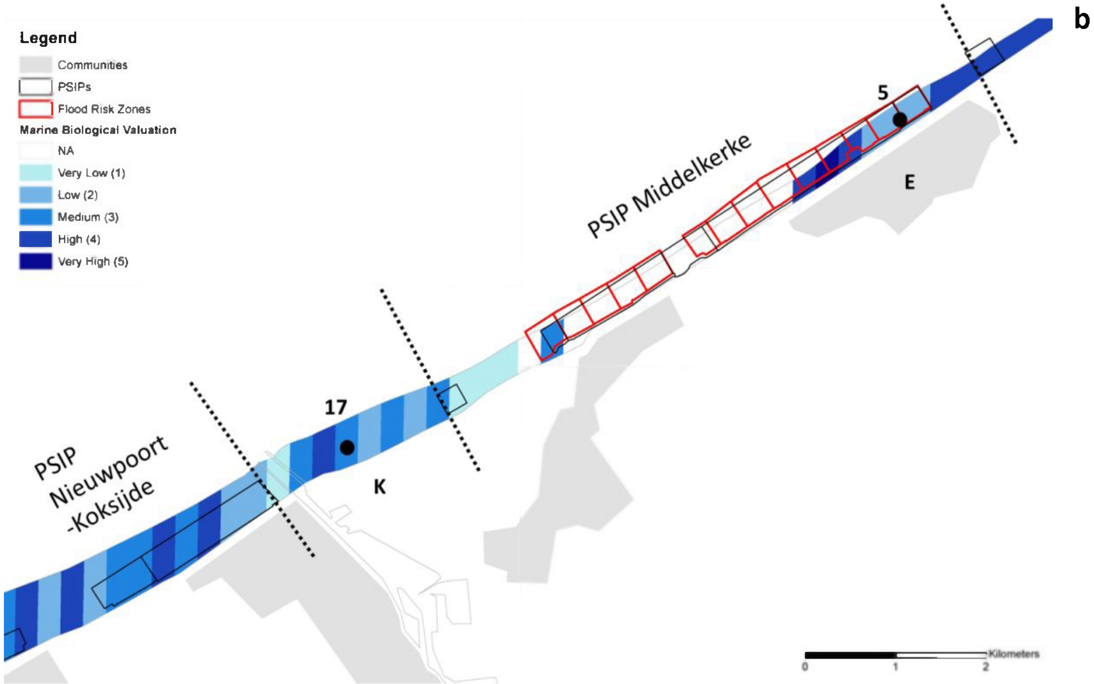
According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Middelkerke has a low biological value (b). Due to the coastal flood risk in the PSIP of Middelkerke, all flood risk priority areas will receive a huge beach nourishment (1 700 000 m³ - location E in b) after October 2013. In front of the casino of Middelkerke, a storm wall and a wave dampening extension will be built to further heighten the coastal defence level.

All the following findings (c, d, f, g, h and i) are based on 59 intertidal and 58 shallow subtidal samples taken in 2002, 2004 and 2006 (Speybroeck et al. 2003; Welvaert 2005; Van Ginderdeuren et al. 2007):

Middelkerke	2002	2004	2006
Intertidal zone	18	15	26
Shallow subtidal zone	18	15	25

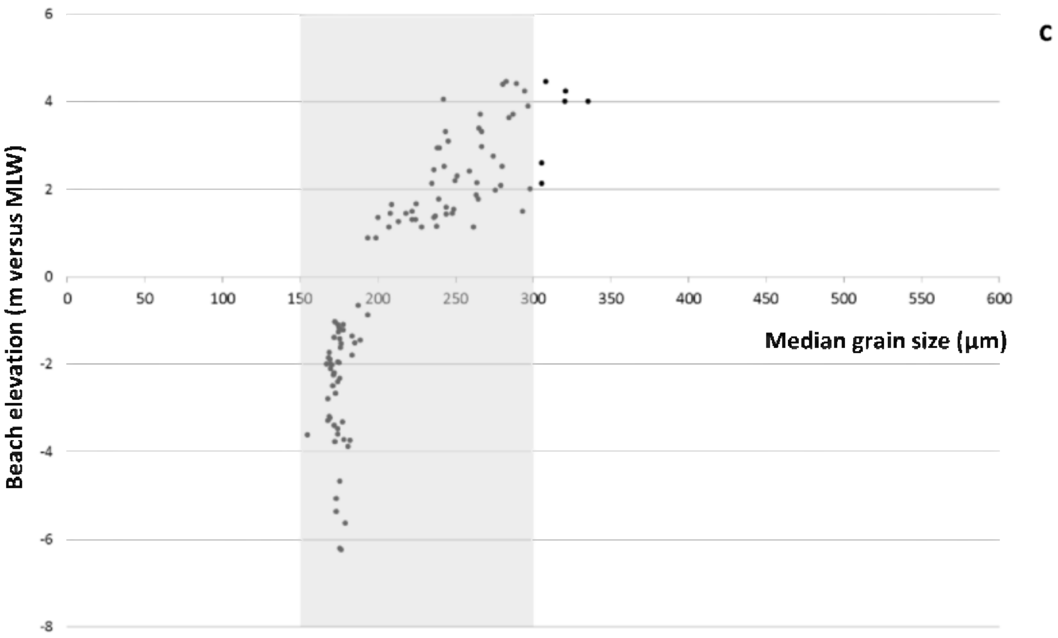
In total, 44 intertidal and 49 shallow subtidal species were identified (e). All mean measured abiotic and biotic





variables fell between the maximum and minimum values of the Belgian beaches (d). The median grain size decreased from MHW (335.58 µm) to MLW (193.55 µm) while the shallow subtidal values fluctuated between 150 and 200 µm (c). Six intertidal samples had a median grain size higher than 300 µm (1 in 2002 and 5 in 2003). The intertidal median grain size and abundance were always higher than the shallow subtidal values. In 2004, the abundance value was half of the 2002 value while the value in 2006 was tripled compared to the 2004 value (f and g).

In the intertidal zone, *Scolelepis squamata* was clearly the most dominant species in all years. *Spiophanes bombyx* was the only dominant species found in 2004 while *Bathyporeia pilosa* was bearly found in the intertidal zone of Middelkerke. In the shallow subtidal zone, *Lanice conchilega* was the only dominant species appearing in all three studied years. It was clearly the most dominant species in 2002 and 2004 but in 2006, *Spio* species became the most dominant shallow subtidal species. *Ensis* juveniles and *Macoma balthica* juveniles were not present in the shallow subtidal samples of Middelkerke.



d

Middelkerke	**INTERTIDAL**				
	2002	2004	2006	minimum	maximum
Beach elevation (m versus MLW)	2.38 ± 0.23	1.70 ± 0.10	2.84 ± 0.25	0.00	5.41
Median grain size (µm)	254.53 ± 6.15	245.07 ± 7.60	263.73 ± 7.61	175.00	464.00
Silt fraction (%)	0.03 ± 0.03	0.00 ± 0.00	0.00 ± 0.00	0.00	2.25
Total organic matter (%)	0.43 ± 0.02	0.10 ± 0.01	0.00 ± 0.00	0.00	1.84
Total organic carbon (%)	0.00 ± 0.00	0.11 ± 0.01	1.47 ± 0.14	0.00	19.00
Carbonate content (%)	0.00 ± 0.00	10.70 ± 0.89	13.10 ± 0.69	0.00	36.68
Species richness (number of species)	3.61 ± 0.20	4.67 ± 1.25	3.58 ± 0.24	0.00	19.00
Abundance (number of individuals.m^{-2})	50.09 ± 6.50	25.80 ± 6.08	68.58 ± 20.03	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.00 ± 0.00	0.71 ± 0.23	0.50 ± 0.15	0.00	6.95
	SHALLOW SUBTIDAL				
	2002	2004	2006	minimum	maximum
Beach elevation (m versus MLW)	-3.41 ± 0.47	-2.74 ± 0.16	-2.08 ± 0.19	0.00	5.41
Median grain size (µm)	176.81 ± 2.25	170.50 ± 0.65	175.17 ± 0.78	175.00	464.00
Silt fraction (%)	3.89 ± 1.30	0.08 ± 0.01	1.10 ± 0.35	0.00	2.25
Total organic matter (%)	0.95 ± 0.28	0.13 ± 0.004	0.00 ± 0.00	0.00	1.84
Total organic carbon (%)	0.00 ± 0.00	0.11 ± 0.01	1.37 ± 0.05	0.00	19.00
Carbonate content (%)	0.00 ± 0.01	9.57 ± 0.35	11.99 ± 0.16	0.00	36.68
Species richness (number of species)	8.73 ± 0.65	2.50 ± 0.67	6.56 ± 0.49	0.00	19.00
Abundance (number of individuals.m^{-2})	42.20 ± 4.62	12.55 ± 0.99	44.60 ± 2.59	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.00 ± 0.00	0.55 ± 0.08	7.04 ± 0.97	0.00	6.95

e

Species list Middelkerke

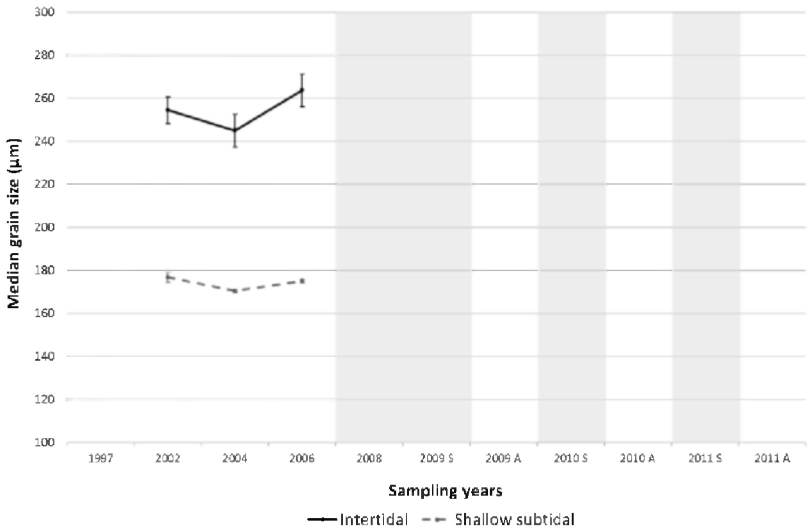
Intertidal

Abra alba
Angulus fabula
Aphelochaeta marioni
Bathyporeia pelagica
Bathyporeia pilosa
Bathyporeia sarsi
Capitella capitata
Chaetozone setosa
Corophium species
Cumopsis goodsir
Diogenes pugilator
Donax vittatus
Ensis directus
Eteone longa
Eumida juveniles
Eurydice affinis
Eurydice pulchra
Gammarus species
Glycera species
Heteromastus filiformis
Lagis koreni
Lanice conchilega
Macoma balthica
Magelona species
Mysella bidentata
Mytilus edulis
Nemertea species
Nephtys cirrosa
Nephtys hombergii
Nephtys juveniles
Nereis longissima
Oligochaeta species
Owenia fusiformis
Paraonis fulgens
Poecilochaetus serpens
Pontocrates arenarius
Pygospio elegans
Sagitta species
Scoloplos armiger
Scolelepis squamata
Sigalion mathildae
Spiophanes bombyx
Spio species
Urothoe poseidonis

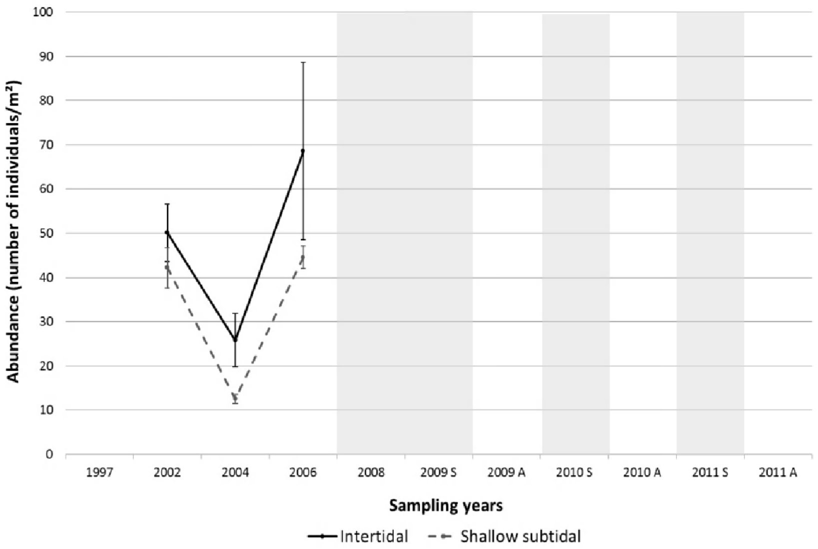
Shallow subtidal

Abra alba
Actinaria species
Angulus fabula
Arenicola marina
Atylus falcatus
Atylus swammerdami
Autolytus species
Bathyporeia elegans
Bathyporeia guilliamsoniana
Bathyporeia pelagica
Bathyporeia sarsi
Capitella capitata
Cirratulidae species
Cumopsis goodsir
Diastylis bradyi
Diastylis rathkei
Diogenes pugilator
Donax vittatus
Donax vittatus juveniles
Ensis directus
Eteone longa
Eurydice pulchra
Glycera species
Haustorius arenarius
Iphinoe trispinosa
Lanice conchilega
Macoma balthica
Magelona species
Microprotopus maculatus
Mysella bidentata
Nemertea species
Nephtys caeca
Nephtys cirrosa
Nephtys hombergii
Nephtys juveniles
Nephtys species
Nereis longissima
Pariambus typicus
Phyllodoce species
Pontocrates altamarinus
Pontocrates arenarius
Pseudocuma longicorne
Scoloplos armiger
Sigalion mathildae
Spiophanes bombyx
Spio species
Spisula subtruncata
Talitrus saltator
Urothoe poseidonis

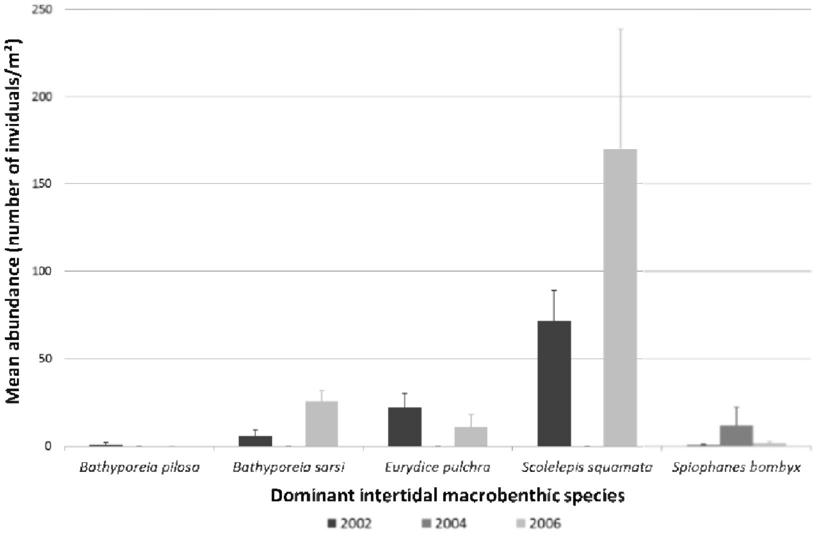
f


Median grain size (µm)
300
280
260
240
220
200
180
160
140
120
100
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal

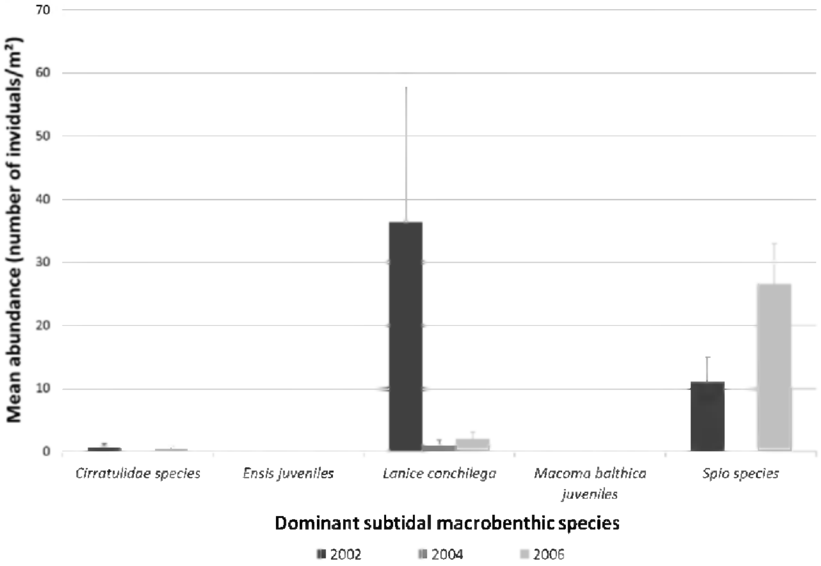
g


Abundance (number of individuals/m²)
100
90
80
70
60
50
40
30
20
10
0
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal

h


Mean abundance (number of invividuals/m²)
250
200
150
100
50
0
Bathyporeia pilosa
Bathyporeia sarsi
Eurydice pulchra
Scolelepis squamata
Spiophanes bombyx
Dominant intertidal macrobenthic species
2002
2004
2006

i

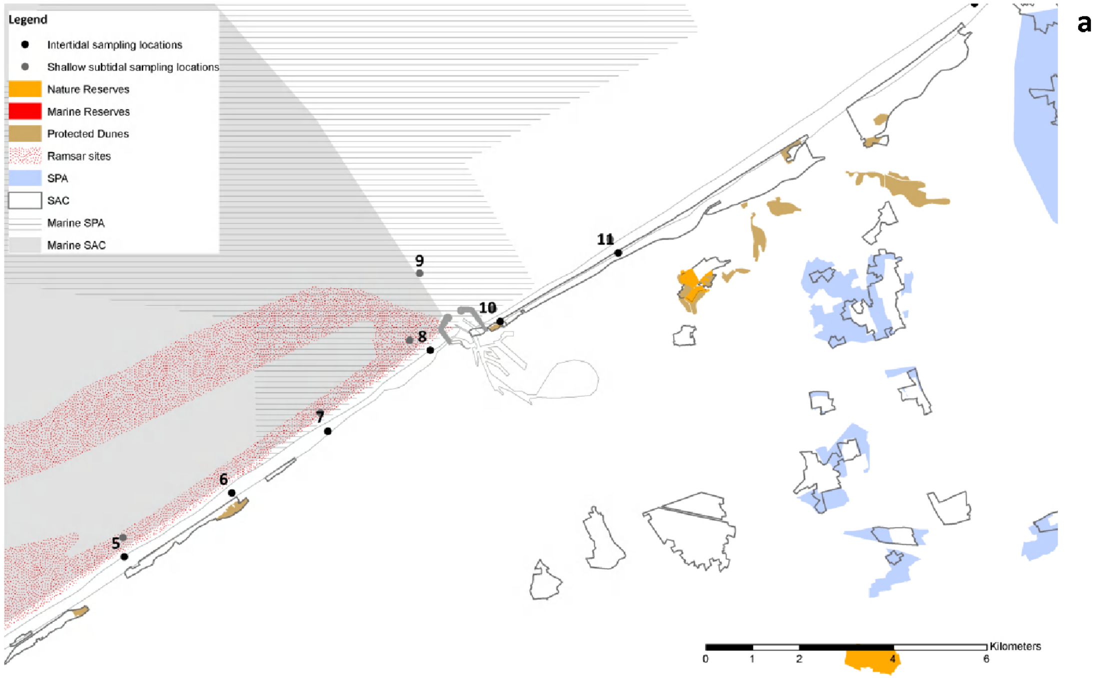

Mean abundance (number of invividuals/m²)
70
60
50
40
30
20
10
0
Cirratulidae species
Ensis juveniles
Lanice conchilega
Macoma balthica juveniles
Spio species
Dominant subtidal macrobenthic species
2002
2004
2006

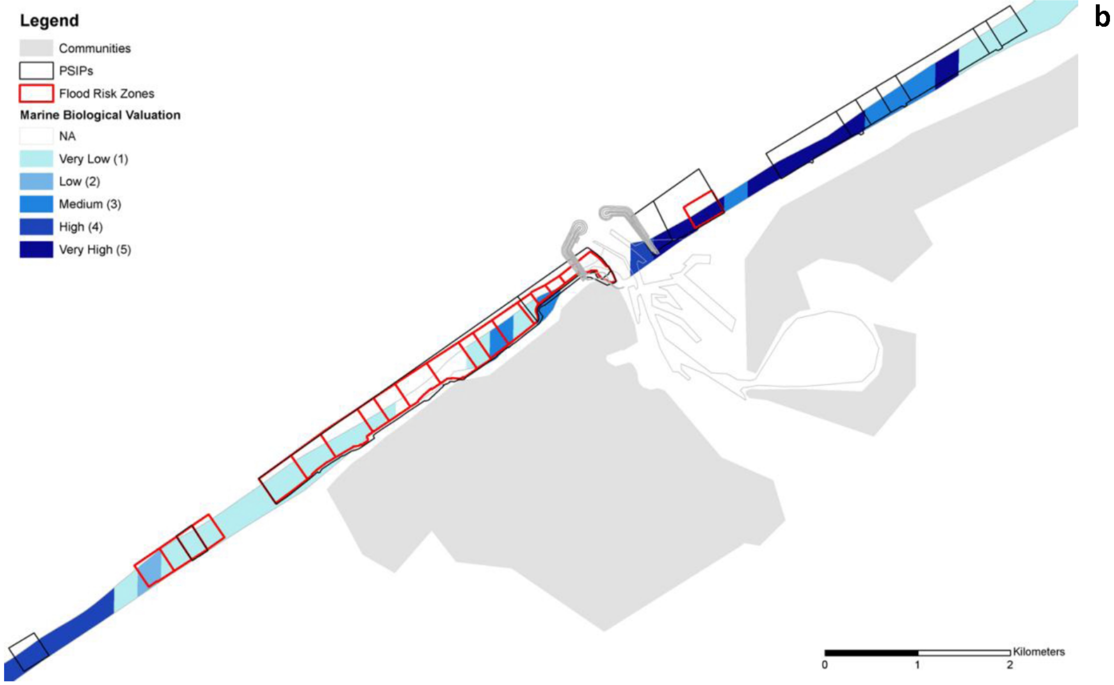
Raversijde

The beach of Raversijde is situated in between the beaches of Middelkerke and Mariakerke (location 6 in all figures). It does not hold any conservation status but it is located in front of protected dunes and a Special Area of Conservation (SAC) (a). The intertidal zone of this beach is not covered by any Provincial Spatial Implementation Plan (PSIP) (b).

According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Raversijde has a very low to low biological value (b). Due to the coastal flood risk at the beach of Raversijde and in the PSIP of Oostende, the entire area will receive coastal defence measures. Between Raversijde and the fairway of Oostende, a huge beach nourishment is planned (1 500 000 m³ - location F in b) after September 2013. A storm wall will be built where the tram rails lie on the dyke.

All the following findings (c, e and f) are based on 20 intertidal samples taken in autumn 1997, at the beach of Raversijde (Speybroeck et al. 2005). In total, 16 species were identified (d). All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches (c). The median grain size decreased from MHW (225.10 µm) to MLW (195 µm) (e) and *Bathyporeia sarsi* was clearly the most dominant species in the intertidal zone of Raversijde.





c

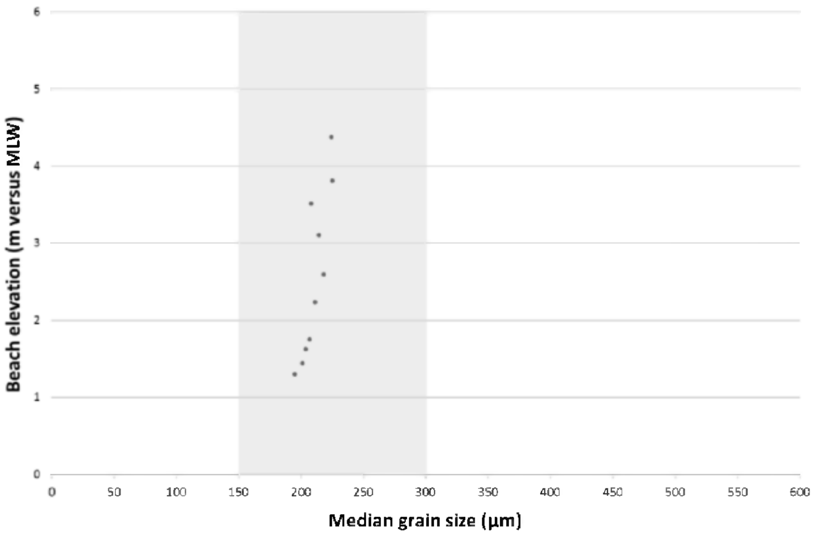
Raversijde	INTERTIDAL			SHALLOW SUBTIDAL		
	1997	minimum	maximum	1997	minimum	maximum
Beach elevation (m versus MLW)	2.57 ± 0.24	0.00	5.41	-	-10.00	2.67
Median grain size (µm)	210.89 ± 2.15	175.00	464.00	-	0.00	319.73
Silt fraction (%)	0.00 ± 0.00	0.00	2.25	-	0.00	89.30
Total organic matter (%)	0.57 ± 0.03	0.00	1.84	-	0.00	11.75
Total organic carbon (%)	0.00 ± 0.00	0.00	19.00	-	0.00	4.85
Carbonate content (%)	0.42 ± 0.07	0.00	36.68	-	0.00	36.10
Species richness (number of species)	7.20 ± 0.35	0.00	19.00	-	0.00	28.00
Abundance (number of individuals.m^{-2})	196.10 ± 43.07	0.00	3988.75	-	0.00	1949.32
Biomass (g AFDW.m^{-2})	0.00 ± 0.00	0.00	6.95	-	0.00	246.14

d

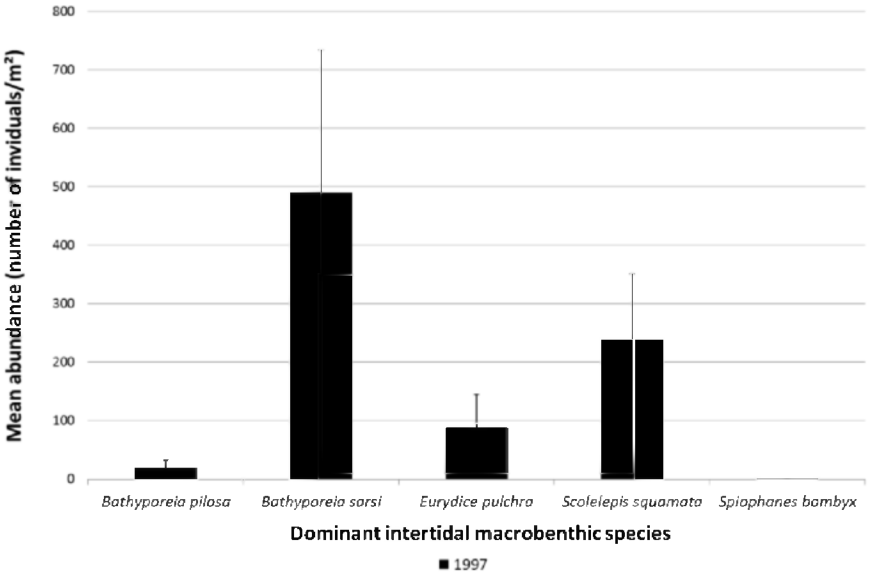
Species list Raversijde

Arenicola marina
Bathyporeia pilosa
Bathyporeia sarsi
Capitella capitata
Cumopsis goodsir
Eteone longa
Eurydice affinis
Eurydice pulchra
Magelona species
Nephtys cirrosa
Nephtys hombergii
Psammodrilus balanoglossoides
Pygospio elegans
Scolelepis squamata
Spiophanes bombyx
Spio species

e



f

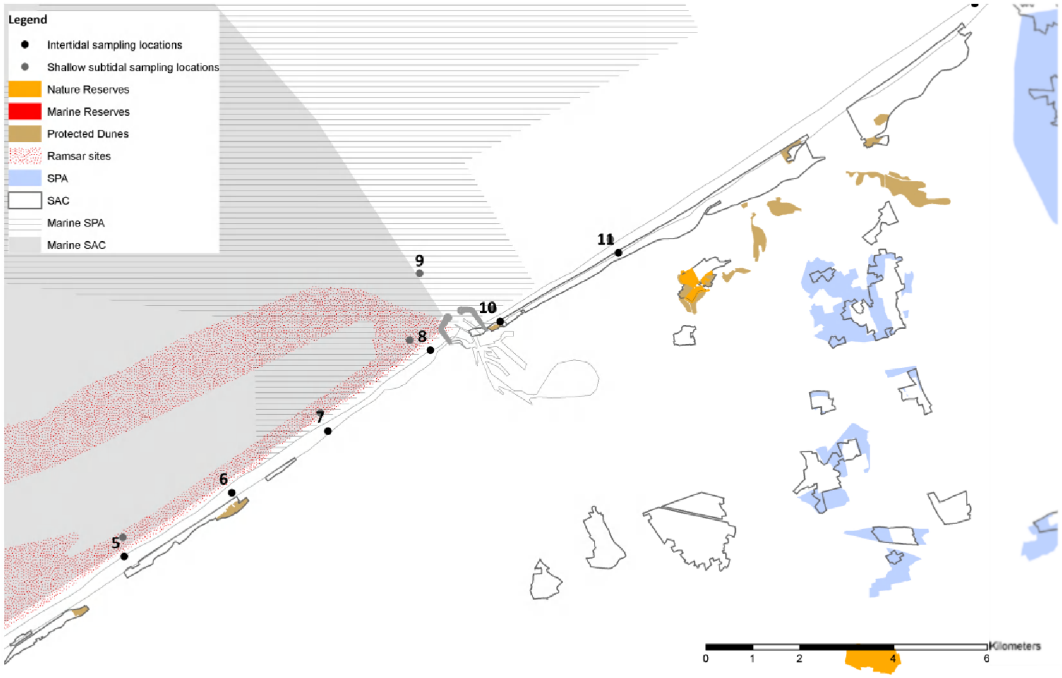


Mariakerke

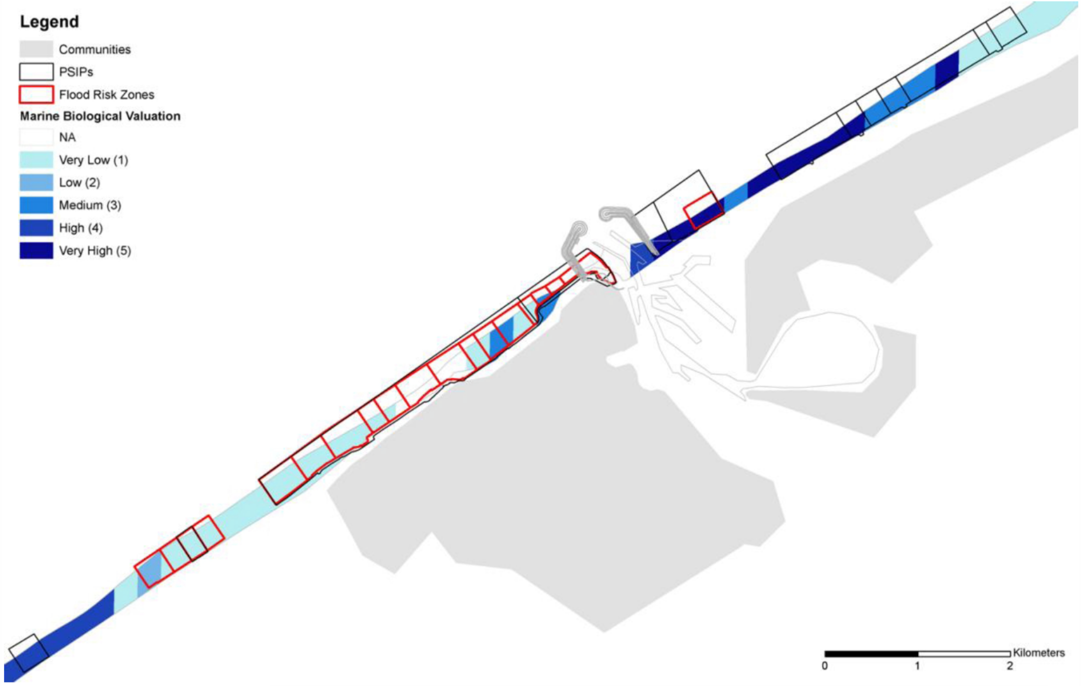
The beach of Mariakerke is situated in front of the western part of the municipality of Oostende, named Mariakerke (location 7 in all figures). It does not hold any conservation status and is regarded as a touristic beach area (a). The Provincial Spatial Implementation Plan (PSIP) of Oostende covers the intertidal zone of this beach (b).

According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Mariakerke has a very low biological value (b). Due to the coastal flood risk in the PSIP of Oostende, the entire area will receive coastal defence measures. Between Raversijde and the fairway of Oostende, a huge beach nourishment is planned (1 500 000 m³ - location F in b) after September 2013. A storm wall will be built where the tram rails lie on the dyke and the slope of the sea dyke will be altered in the touristic beach area of Mariakerke. In front of Mariakerke, a foreshore nourishment is planned in the shallow subtidal zone, after October 2013.

All the following findings (c, e, f and g) are based on 15 intertidal and 15 shallow subtidal samples taken in autumn 2011 (2011 A), at the beach of Mariakerke (Vanden Eede et al. 2013). In total, 10 intertidal and 32 shallow subtidal species were identified (d). All mean



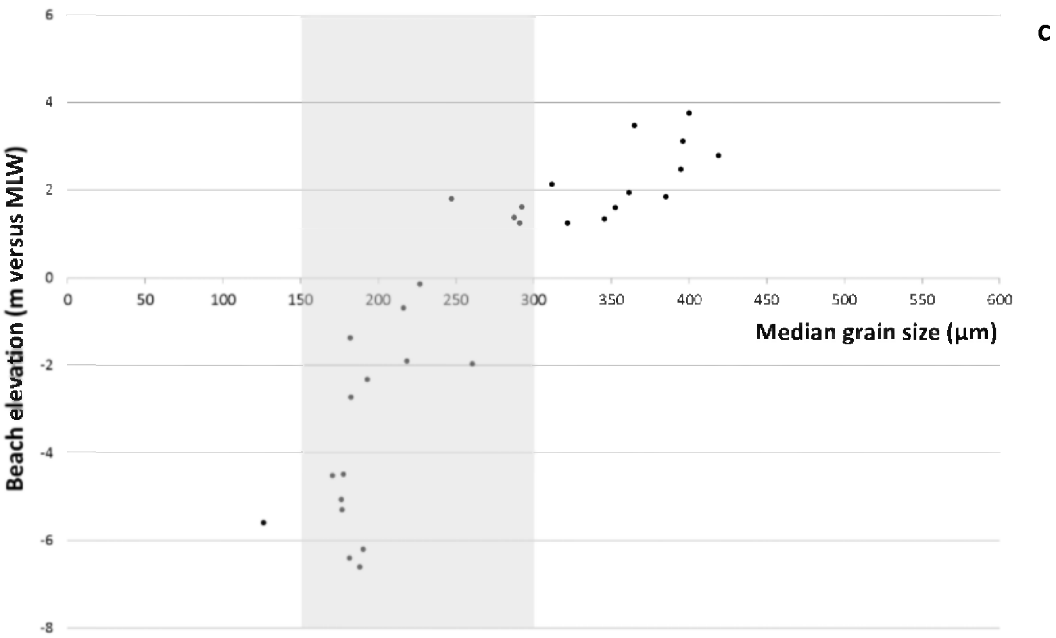
a



b

measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches ed). The median grain size decreased from MHW (419.03 µm) to MLW (247.14 µm) while the shallow subtidal values fluctuated between 150 and 250 µm (c). Eleven of the 15 intertidal samples had a median grain size higher than 300 µm (the highest 10 samples and the lowest sample in the intertidal zone). One very low value (< 150 µm) was detected in the shallow subtidal zone. The intertidal median grain size (344.64 ± 13.12 µm) and abundance (204.08 ± 130.40 individuals.m^{-2}) were higher than the shallow subtidal values (191.22 ± 7.86 µm; 166.05 ± 38.61 individuals.m^{-2}).

In the intertidal zone, *Scolelepis squamata* was clearly the most dominant species. *Bathyporeia pilosa*, *Eurydice pulchra* and *Spiophanes bombyx* were not present in the intertidal zone of Mariakerke. In the shallow subtidal zone, *Spio* species was clearly the most dominant species. *Ensis* juveniles and *Macoma balthica* juveniles were not present in the shallow subtidal samples of Mariakerke.



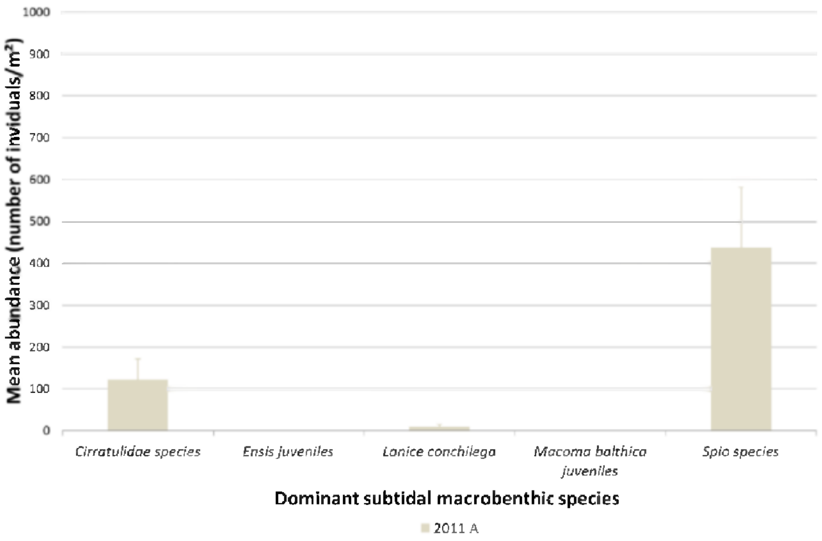
d

Species list Mariakerke			
Intertidal	Shallow subtidal		
Bathyporeia pelagica	*Abra alba*	*Ensis species*	*Nereis longissima*
Bathyporeia sarsi	*Atylus falcatus*	*Eteone longa*	*Owenia fusiformis*
Cumopsis goodsir	*Atylus swammerdami*	*Glycera species*	*Pholoe minuta*
Eteone flava	*Autolytus species*	*Heteromastus filiformis*	*Phyllodoce species*
Eteone longa	*Bathyporeia pelagica*	*Jassa herdmani*	*Pontocrates altamarinus*
Haustorius arenarius	*Capitella capitata*	*Lanice conchilega*	*Scoloplos armiger*
Nemertea species	*Cirratulidae species*	*Macoma balthica*	*Scolelepis squamata*
Nephtys cirrosa	*Corophium species*	*Magelona species*	*Spiophanes bombyx*
Oligochaeta species	*Cumopsis goodsir*	*Nemertea species*	*Spio species*
Scolelepis squamata	*Diogenes pugilator*	*Nephtys cirrosa*	*Urothoe poseidonis*
	Donax vittatus juveniles	*Nephtys hombergii*	

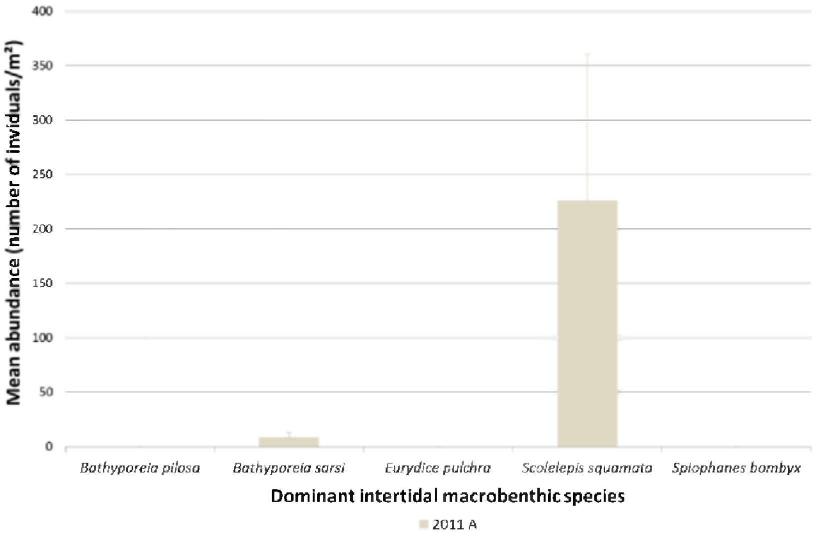
e

Mariakerke	INTERTIDAL			SHALLOW SUBTIDAL		
	2011	minimum	maximum	2011	minimum	maximum
Beach elevation (m versus MLW)	2.12 ± 0.21	0.00	5.41	-3.69 ± 0.57	-10.00	2.67
Median grain size (µm)	344.64 ± 13.12	175.00	464.00	191.22 ± 7.86	0.00	319.73
Silt fraction (%)	0.00 ± 0.00	0.00	2.25	3.96 ± 1.89	0.00	89.30
Total organic matter (%)	0.62 ± 0.05	0.00	1.84	1.40 ± 0.29	0.00	11.75
Total organic carbon (%)	0.08 ± 0.01	0.00	19.00	0.18 ± 0.04	0.00	4.85
Carbonate content (%)	13.97 ± 1.47	0.00	36.68	9.42 ± 0.63	0.00	36.10
Species richness (number of species)	2.53 ± 0.34	0.00	19.00	10.13 ± 1.08	0.00	28.00
Abundance (number of individuals.m^{-2})	204.08 ± 130.40	0.00	3988.75	166.05 ± 38.61	0.00	1949.32
Biomass (g AFDW.m^{-2})	0.62 ± 0.34	0.00	6.95	45.20 ± 19.89	0.00	246.14

f



g



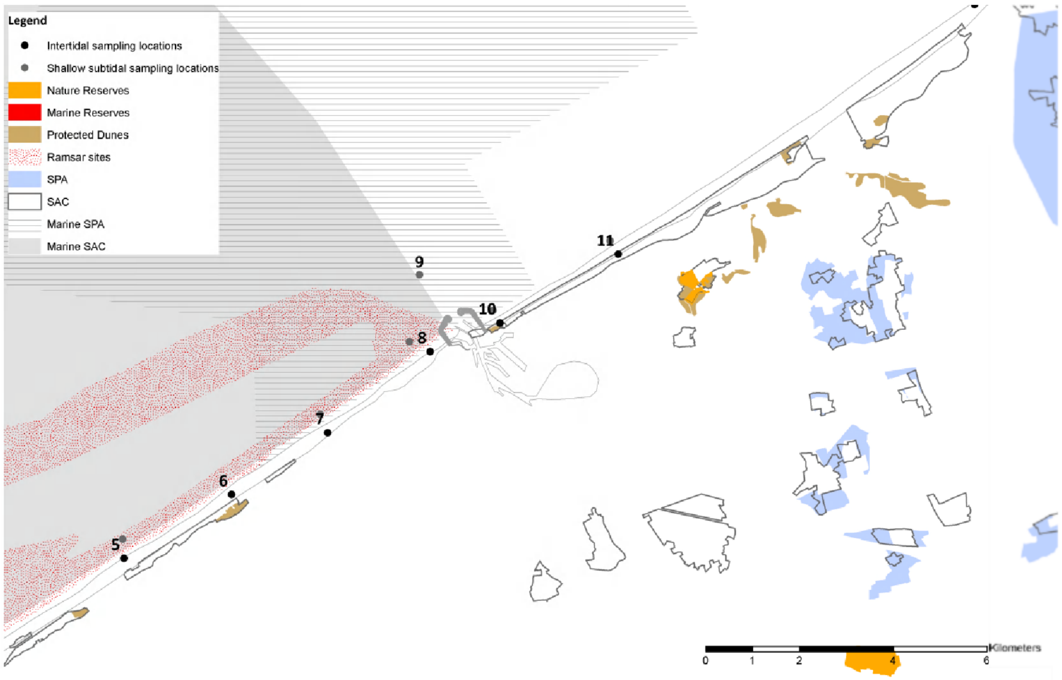
Oostende-Center

The beach of Oostende-Center is situated in front of the municipality of Oostende, at the western side of the fairway to Oostende (location 8 in all figures). It does not hold any conservation status and is regarded as a touristic beach area (a). The Provincial Spatial Implementation Plan (PSIP) of Oostende covers the intertidal zone of this beach (b).

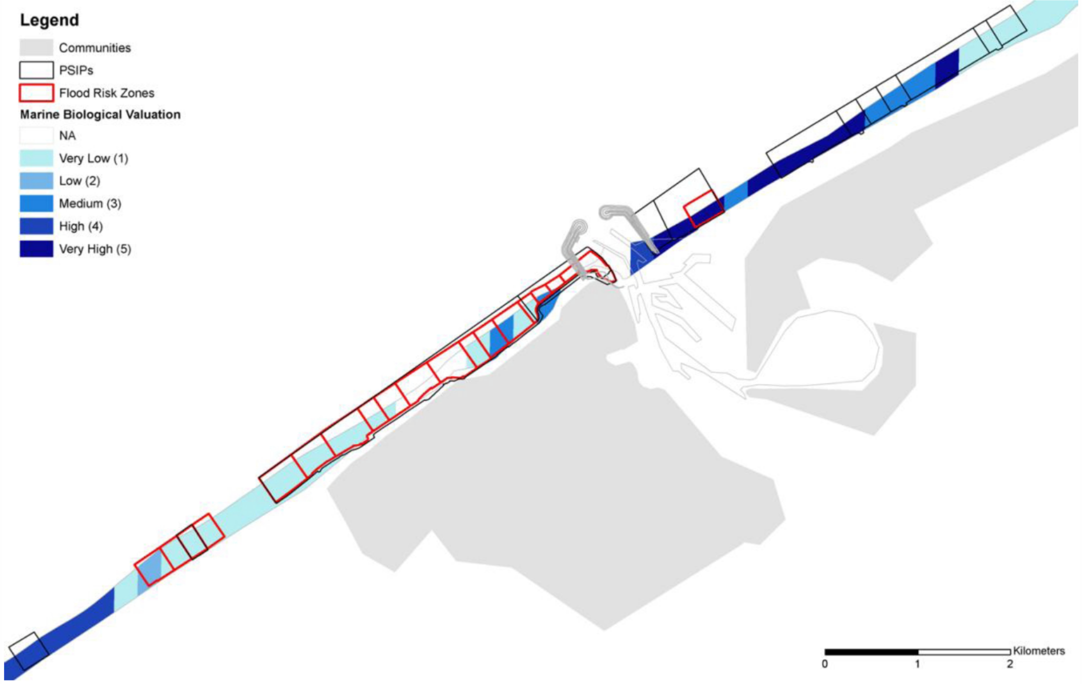
According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Oostende-Center has a very low to medium biological value (b). Due to the high coastal flood risk at the beach of Oostende-Center, it was nourished with coarse offshore sediment in June 2004 (600 000 m³), spring 2005 (100 000 m³), spring and autumn 2007 (each 75 000 m³), autumn 2008 (75 000 m³), spring 2010 (75 000 m³) and autumn 2011 (75 000 m³). Between Raversijde and the fairway of Oostende, a huge beach nourishment is planned (1 500 000 m³ - location F in b) after September 2013.

All the following findings (c, d, f, g, h and i) are based on 103 intertidal and 112 shallow subtidal samples taken in 2002, 2004 and 2006 (Speybroeck et al. 2003; Welvaert 2005; Van Ginderdeuren et al. 2007):

Oostende-Center	**2002**	**2004**	**2006**
Intertidal zone	33	45	25
Shallow subtidal zone	42	45	25



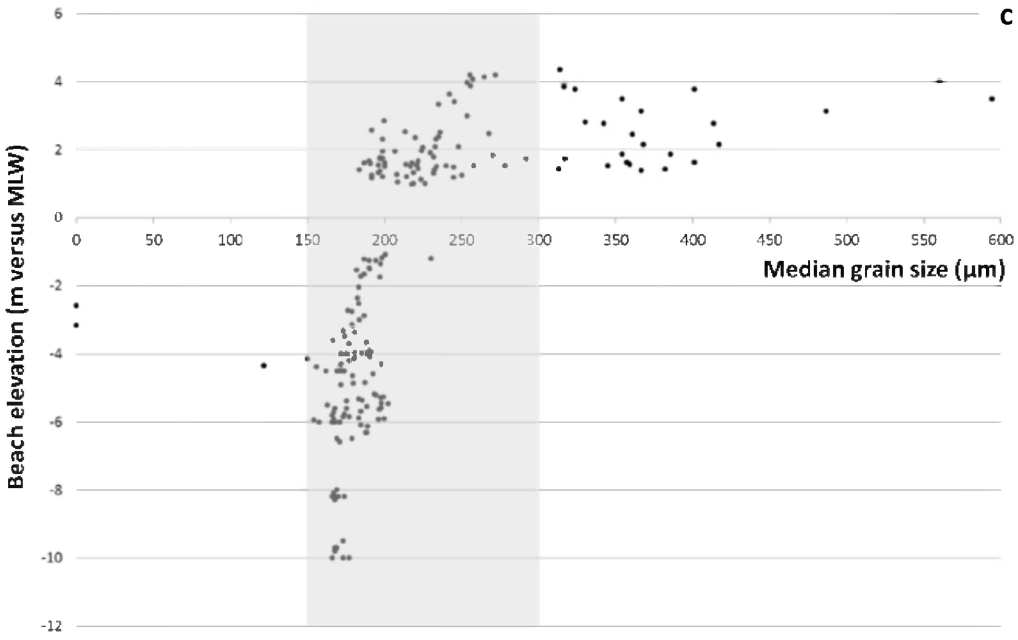
a



b

In total, 29 intertidal and 59 shallow subtidal species were identified (e). All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches (d). The median grain size decreased from MHW (594.24 µm) to MLW (184 µm) while the shallow subtidal values fluctuated between 150 and 200 µm (c). 25 intertidal samples had a median grain size higher than 300 µm (4 in 2002: 2 in 2004; 19 in 2006). One very low value (< 150 µm) and two null values were detected in the shallow subtidal zone. The intertidal median grain size was always higher than the shallow subtidal values, with a peak value in 2006 (397.44 ± 14.85 µm). In 2002 and 2006, the intertidal abundance value was respectively four and three times higher than the shallow subtidal value. In 2004 however, the shallow subtidal value was three times higher than the intertidal value (f and g).

In the intertidal zone, *Scolelepis squamata* was clearly the most dominant species in all years. Only in 2002, *Bathyporeia pilosa* was found in very low abundance in the intertidal zone of Oostende-Center. In the shallow subtidal zone, *Lanice conchilega* was the most dominant species in 2002 and 2004 while *Cirratulidae* species dominated in 2006. *Ensis* juveniles and *Macoma balthica* juveniles were not present in the shallow subtidal samples of Oostende-Center.



d

Oostende-Center	**INTERTIDAL**				
	2002	2004	2006	minimum	maximum
Beach elevation (m versus MLW)	2.51 ± 0.21	1.67 ± 0.05	2.54 ± 0.18	0.00	5.41
Median grain size (µm)	245.38 ± 6.57	223.16 ± 5.44	397.44 ± 14.85	175.00	464.00
Silt fraction (%)	0.009 ± 0.009	0.003 ± 0.002	0.003 ± 0.003	0.00	2.25
Total organic matter (%)	0.51 ± 0.05	0.09 ± 0.003	0.00 ± 0.00	0.00	1.84
Total organic carbon (%)	0.00 ± 0.00	0.25 ± 0.03	2.08 ± 0.30	0.00	19.00
Carbonate content (%)	0.00 ± 0.00	15.55 ± 0.69	16.86 ± 1.57	0.00	36.68
Species richness (number of species)	3.19 ± 0.27	3.60 ± 0.24	1.79 ± 0.21	0.00	19.00
Abundance (number of individuals.m^{-2})	118.08 ± 36.12	46.87 ± 8.42	139.96 ± 53.13	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.00 ± 0.00	0.34 ± 0.06	0.12 ± 0.03	0.00	6.95
	SHALLOW SUBTIDAL				
	2002	2004	2006	minimum	maximum
Beach elevation (m versus MLW)	-3.72 ± 0.27	-6.33 ± 0.32	-4.55 ± 0.25	0.00	5.41
Median grain size (µm)	174.47 ± 6.46	172.42 ± 1.12	185.85 ± 1.37	175.00	464.00
Silt fraction (%)	7.26 ± 1.47	0.96 ± 0.18	4.48 ± 1.03	0.00	2.25
Total organic matter (%)	0.77 ± 0.07	0.32 ± 0.04	0.00 ± 0.00	0.00	1.84
Total organic carbon (%)	0.00 ± 0.00	0.35 ± 0.04	1.30 ± 0.06	0.00	19.00
Carbonate content (%)	0.00 ± 0.01	12.57 ± 0.47	11.99 ± 0.24	0.00	36.68
Species richness (number of species)	6.36 ± 0.36	8.19 ± 0.52	8.12 ± 0.62	0.00	19.00
Abundance (number of individuals.m^{-2})	31.15 ± 2.14	141.35 ± 29.44	48.04 ± 3.54	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.00 ± 0.00	1.32 ± 0.24	6.80 ± 0.57	0.00	6.95

e

Species list Oostende-Center

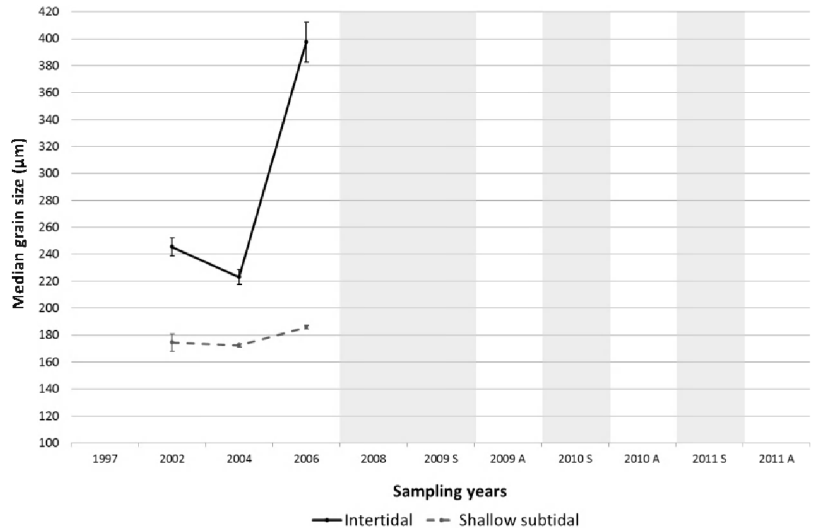
Intertidal

Actinaria species
Arenicola marina
Bathyporeia elegans
Bathyporeia guilliamsoniana
Bathyporeia pelagica
Bathyporeia pilosa
Bathyporeia sarsi
Capitella capitata
Corophium species
Cumopsis goodsir
Ensis directus
Eteone flava
Eteone longa
Eurydice pulchra
Haustorius arenarius
Iphinoe trispinosa
Lanice conchilega
Magelona species
Malmgreniella juveniles
Nemertea species
Nephtys cirrosa
Nephtys hombergii
Nephtys juveniles
Polydora species
Pontocrates arenarius
Pygospio elegans
Scolelepis squamata
Spiophanes bombyx
Urothoe poseidonis

Shallow subtidal

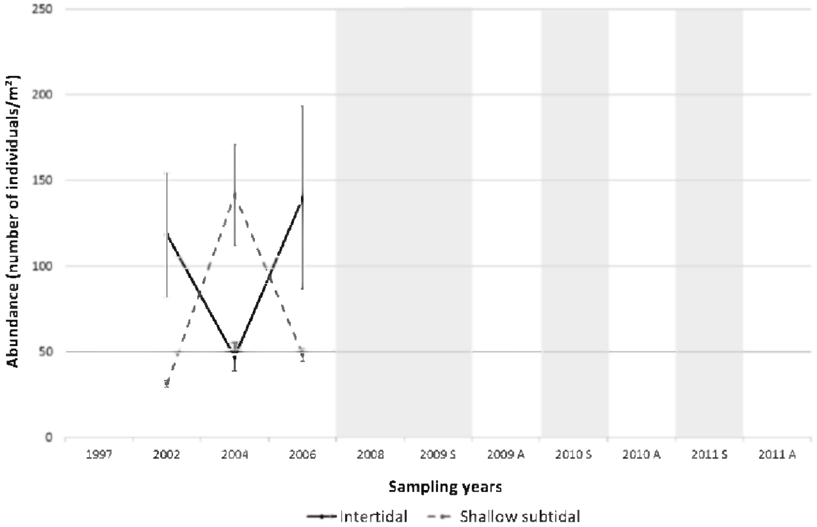
Abra alba
Actinaria species
Angulus fabula
Angulus tenuis
Aphelochaeta marioni
Arenicola marina
Atylus falcatus
Atylus swammerdami
Autolytus species
Bathyporeia guilliamsoniana
Bathyporeia pelagica
Bathyporeia sarsi
Capitella capitata
Chaetozone setosa
Cirratulidae species
Corophium species
Cumopsis goodsir
Diastylis bradyi
Diastylis rathkei
Diogenes pugilator
Donax vittatus
Donax vittatus juveniles
Ensis directus
Eteone longa
Eumida sanguinea
Eumida juveniles
Eurydice pulchra
Gammarus species
Glycera species
Iphinoe trispinosa
Lagis koreni
Lanice conchilega
Macoma balthica
Magelona species
Malmgreniella juveniles
Microprotopus maculatus
Mysella bidentata
Mytilus edulis
Nemertea species
Nephtys assimilis
Nephtys cirrosa
Nephtys hombergii
Nephtys juveniles
Nephtys species
Nereis longissima
Notomastus latericeus
Oligochaeta species
Phyllodoce species
Polydora species
Pontocrates altamarinus
Pseudocuma longicorne
Pygospio elegans
Scoloplos armiger
Scolelepis squamata
Spiophanes bombyx
Spionidae species
Spio species
Spisula subtruncata
Urothoe poseidonis

f



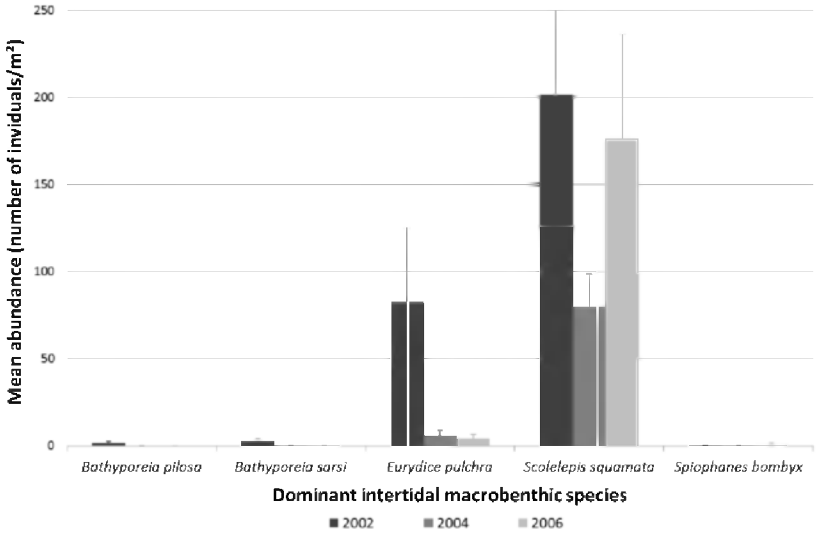
Median grain size (µm)
420
400
380
360
340
320
300
280
260
240
220
200
180
160
140
120
100
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal

g



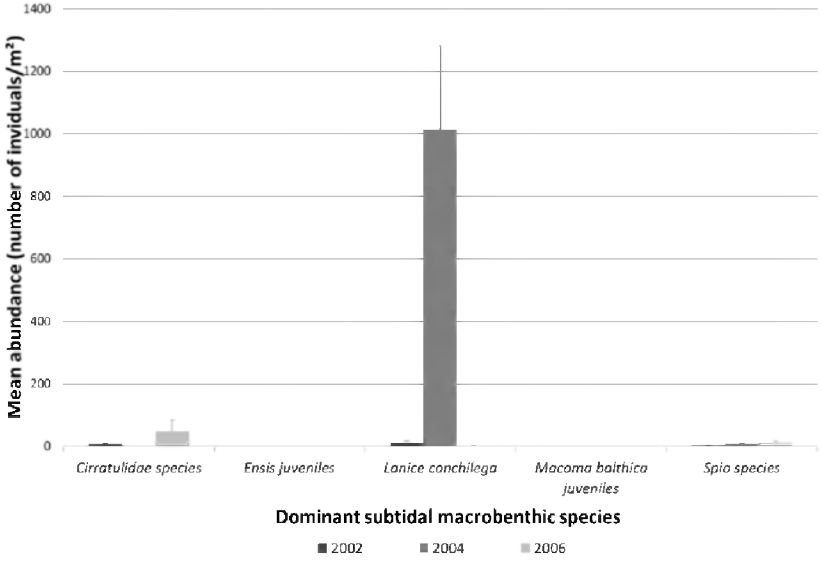
Abundance (number of individuals/m²)
250
200
150
100
50
0
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal

h



Mean abundance (number of inviduals/m²)
250
200
150
100
50
0
Bathyporeia pilosa
Bathyporeia sarsi
Eurydice pulchra
Scolelepis squamata
Spiophanes bombyx
Dominant intertidal macrobenthic species
2002
2004
2006

i



Mean abundance (number of inviduals/m²)
1400
1200
1000
800
600
400
200
0
Cirratulidae species
Ensis juveniles
Lanice conchilega
Macoma balthica juveniles
Spio species
Dominant subtidal macrobenthic species
2002
2004
2006

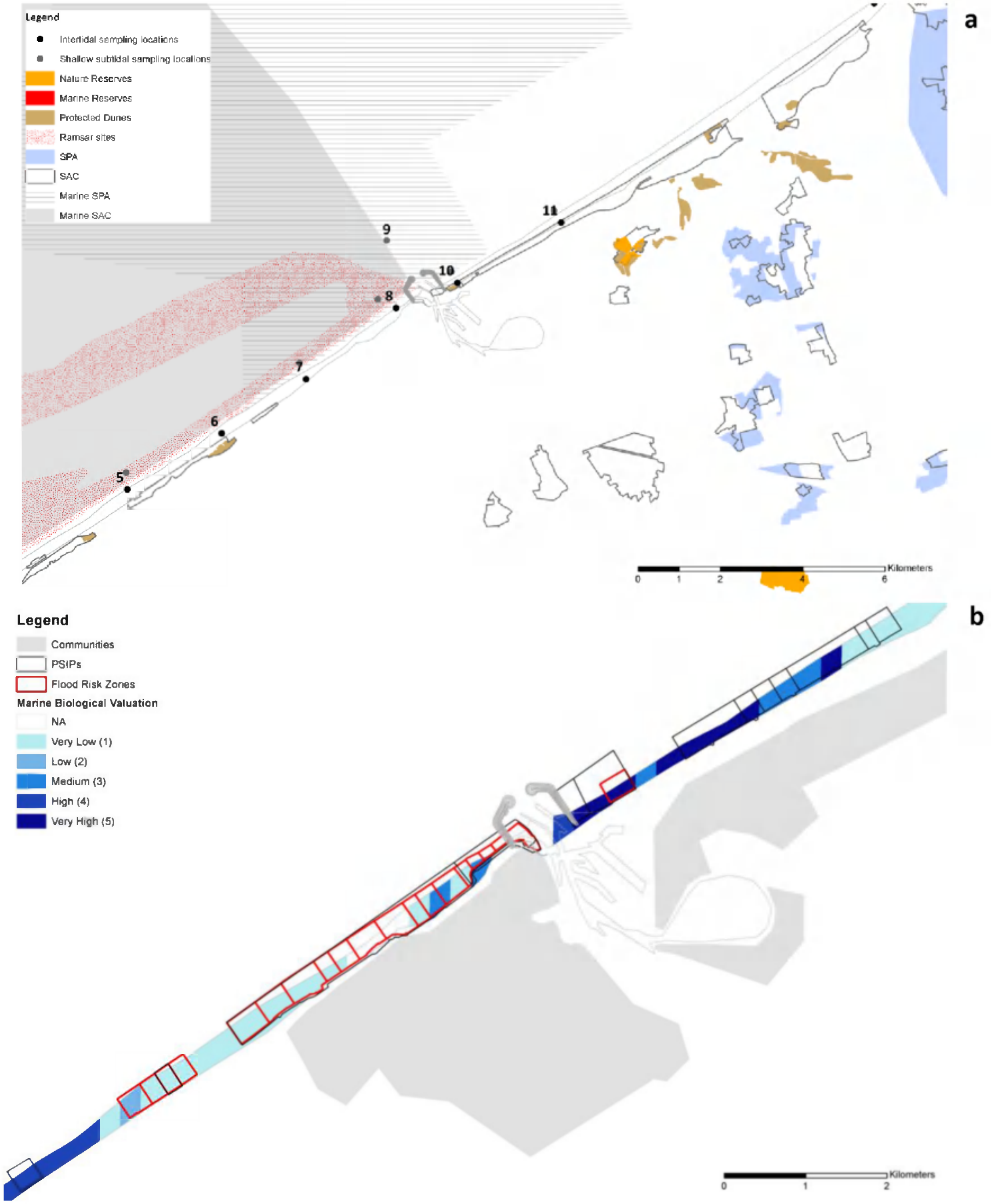
Oostende-Fairway

Oostende-Fairway is situated in the fairway of Oostende (location 9 in all figures). It is the only sampling location that is not situated on a beach. It falls within the marine Special Area of Conservation (SAC) and the marine Special Protection Area 1 (SPA1). No Provincial Spatial Implementation Plan (PSIP) covers this sampling location (b) and it does not fall within the area of the marine biological valuation map of the shallow Belgian coastal zone.

Ostend has an active harbour which needs regular maintenance. A large amount of structural harbour works and dredging activities have been and are being executed to create a deeper and wider shipping lane, e.g. elongation of the harbour jetties. Due to the coastal flood risk in the PSIP of Oostende, the entire area will receive coastal defence measures in the following years, even the harbour area.

All the following findings (c, e and f) are based on 27 shallow subtidal samples taken in 2004 (15 samples) and 2006 (12 samples) (Welvaert 2005; Van Ginderdeuren et al. 2007).

In total, 30 shallow subtidal species were identified (e). All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches



(d). The shallow subtidal median grain size fluctuated between 150 and 250 µm (e). Four very low values (< 150 µm) were detected in the shallow subtidal zone in 2006.
The shallow subtidal median grain size and abundance were higher in 2004 (198.87 ± 2.99 µm; 38.91 ± 5.30 individuals.m^{-2}) than in 2006 (159.26 ± 20.69 µm; 25.56 ± 3.04 individuals.m^{-2}).

In the shallow subtidal zone, *Spio* species was the most dominant species in 2004 while *Cirratulidae* species dominated in 2006. *Ensis* juveniles and *Macoma balthica* juveniles were not present in the shallow subtidal samples of Oostende-Fairway (f).

c

Oostende-Fairway	SHALLOW SUBTIDAL			
	2004	2006	minimum	maximum
Beach elevation (m versus MLW)	-6.25 ± 0.29	-6.73 ± 0.30	0.00	5.41
Median grain size (µm)	198.87 ± 2.99	159.26 ± 20.69	175.00	464.00
Silt fraction (%)	0.81 ± 0.16	23.99 ± 6.73	0.00	2.25
Total organic matter (%)	0.34 ± 0.04	0.00 ± 0.00	0.00	1.84
Total organic carbon (%)	0.30 ± 0.05	2.02 ± 0.36	0.00	19.00
Carbonate content (%)	9.27 ± 0.40	13.52 ± 1.16	0.00	36.68
Species richness (number of species)	6.80 ± 0.63	4.25 ± 0.52	0.00	19.00
Abundance (number of individuals.m^{-2})	38.91 ± 5.30	25.56 ± 3.04	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.59 ± 0.17	0.68 ± 0.16	0.00	6.95

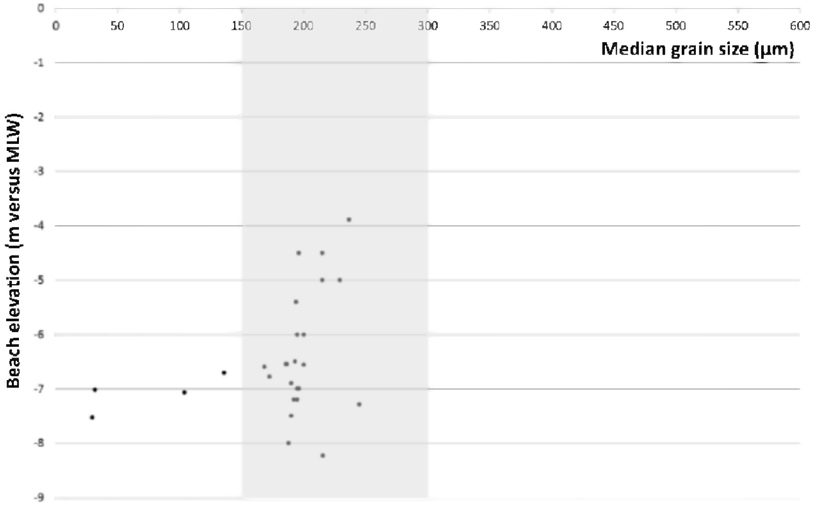
d

Species list Oostende-Fairway

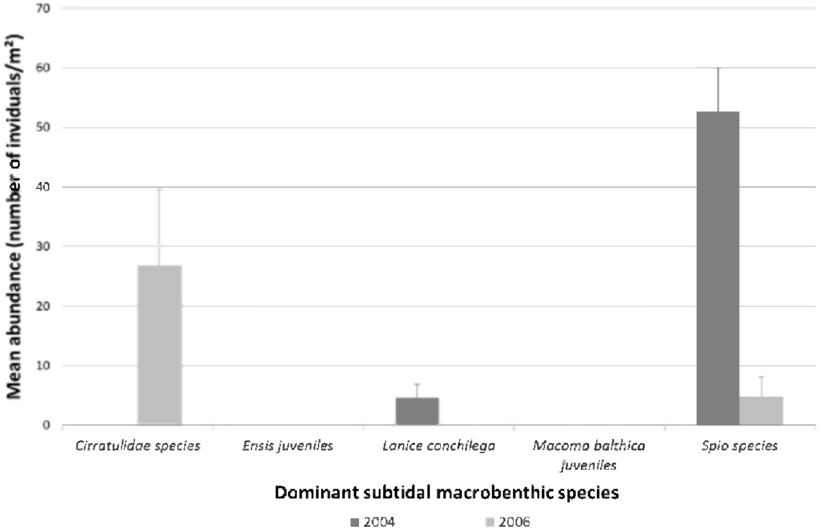
Shallow subtidal

Abra alba
Angulus tenuis
Aphelochaeta marioni
Atylus falcatus
Bathyporeia pelagica
Capitella capitata
Cirratulidae species
Diastylis bradyi
Donax vittatus
Ensis directus
Glycera species
Lagis koreni
Lanice conchilega
Macoma balthica
Magelona species
Nemertea species
Nephtys assimilis
Nephtys cirrosa
Nephtys hombergii
Nephtys juveniles
Nephtys species
Oligochaeta species
Owenia fusiformis
Poecilochaetus serpens
Pontocrates altamarinus
Sagitta species
Scoloplos armiger
Spiophanes bombyx
Spio species
Urothoe poseidonis

e



f



Oostende-East

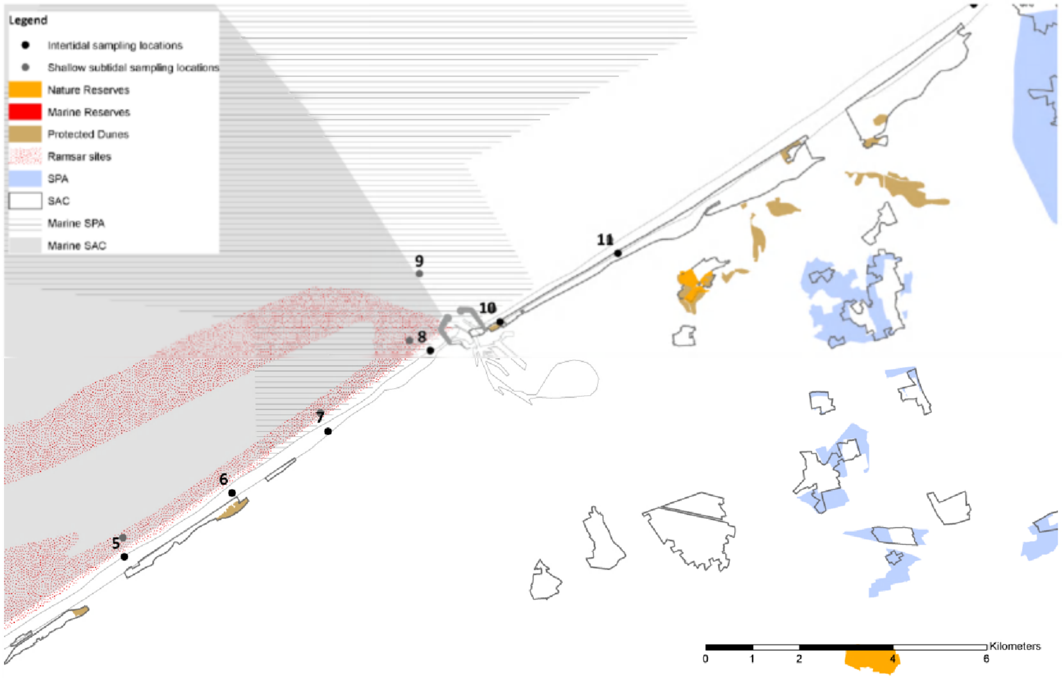
The beach of Oostende-East is situated at the eastern side of the fairway to Oostende (location 10 in all figures). It does not hold any conservation status but it is located in front of protected dunes and a Special Area of Conservation (SAC) (a). The Provincial Spatial Implementation Plan (PSIP) of Oostende covers the intertidal zone of this beach (b).

According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Oostende-East has a high to very high biological value (b). Due to the coastal flood risk in the PSIP of Oostende, the entire area will receive coastal defence measures. The beach of Oostende-East will receive a beach nourishment (500 000 m³) in the near future (location G in b).

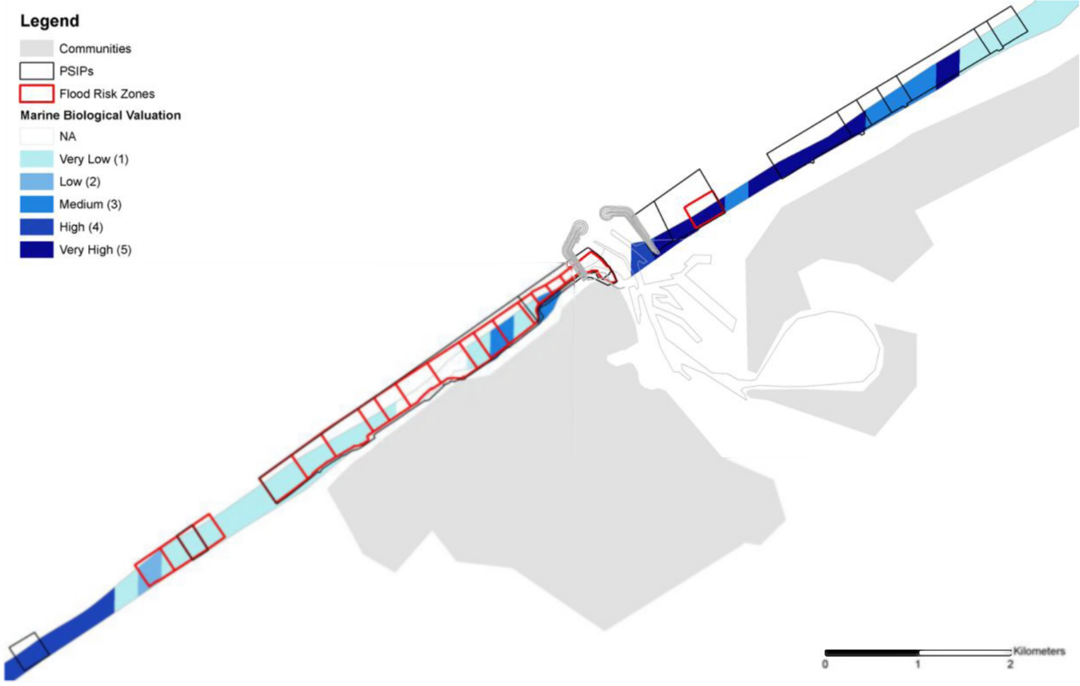
All the following findings (c, d, f, g, h and i) are based on 110 intertidal and 110 shallow subtidal samples taken in 2004, 2006 and in spring (S) and autumn (A) of 2010 and 2011 (Welvaert 2005; Van Ginderdeuren et al. 2007; Vanden Eede et al. 2011 and 2013):

Oostende-East	2004	2006	2010		2011	
		A	S	A	S	A
Intertidal zone	25	25	15	15	15	15
Shallow subtidal zone	25	25	15	15	15	15

In total, 49 intertidal and 67 shallow subtidal species were identified (e). All mean measured abiotic and biotic



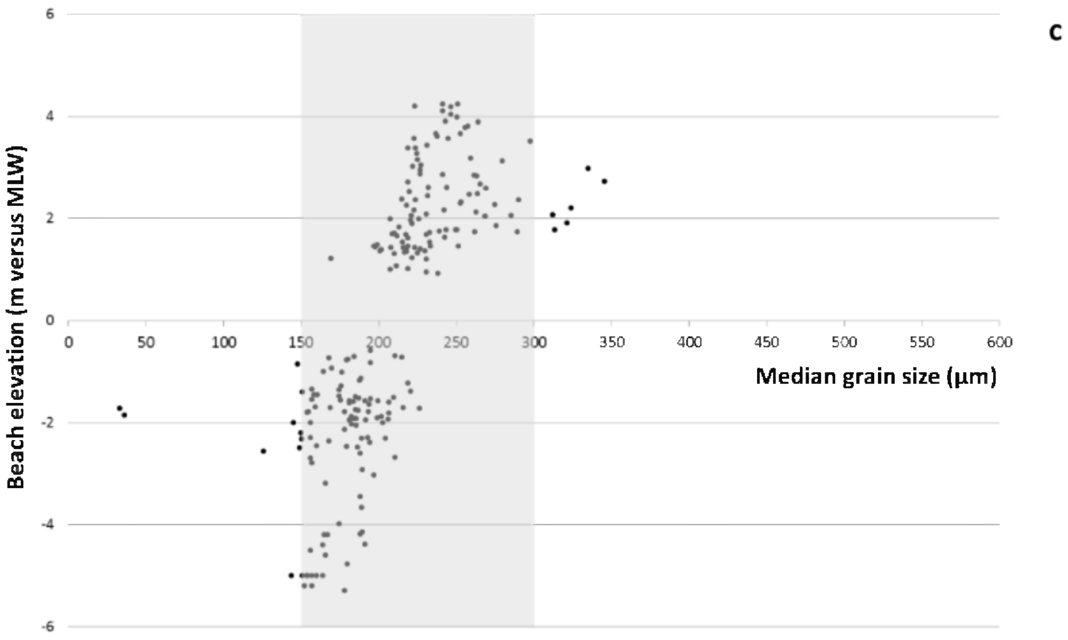
a



b

variables fell between the maximum and minimum values of the Belgian beaches (d). The median grain size decreased from MHW (345.65 µm) to MLW (169.48 µm) while the shallow subtidal values fluctuated between 150 and 250 µm (c). Six intertidal samples, all sampled in autumn 2010, had a median grain size higher than 300 µm. Seven very low values (< 150 µm) were detected in the shallow subtidal zone (3 in 2004; 1 in 2006; 1 in spring 2011; 2 in autumn 2011). The intertidal median grain size was always higher than the shallow subtidal values. In 2004 and 2006, the intertidal abundance was higher than the shallow subtidal abundance. However, from spring 2010 onwards, the shallow subtidal abundance in autumn was higher than the intertidal abundance while in spring the values were lower (2010) or quite similar (2011) (f and g).

In the intertidal zone, *Bathyporeia pilosa* and *Scolelepis squmata* were clearly the most dominant species in all years (h), except in 2004 when *Spiophanes bombyx* dominated. In the shallow subtidal zone, *Ensis* juveniles (1416.50 ± 615.06 individuals.m^{-2}) and *Macoma balthica* juveniles (1044.19 ± 257.78 individuals.m^{-2}) showed extreme peak values in autumn 2010. These values were not shown in figure i to allow for a better interpretation of the remaining abundance values. In autumn 2010, *Cirratulidae* species were also very abundant. *Spio* species was the only dominant species found in all years.



d

Oostende-East	INTERTIDAL							
	2004	2006	2010		2011		minimum	maximum
Beach elevation (m versus MLW)	1.86 ± 0.14	2.96 ± 0.20	2.25 ± 0.22	2.17 ± 0.10	2.08 ± 0.29	2.42 ± 0.16	0.00	5.41
Median grain size (µm)	215.68 ± 2.00	231.76 ± 2.90	246.27 ± 4.95	299.38 ± 7.10	245.69 ± 6.02	233.00 ± 3.38	175.00	464.00
Silt fraction (%)	0.04 ± 0.01	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00	2.25
Total organic matter (%)	0.10 ± 0.005	0.00 ± 0.01	0.60 ± 0.03	0.61 ± 0.03	0.47 ± 0.03	0.77 ± 0.05	0.00	1.84
Total organic carbon (%)	0.13 ± 0.01	2.04 ± 0.71	0.21 ± 0.02	0.08 ± 0.03	0.17 ± 0.02	0.10 ± 0.01	0.00	19.00
Carbonate content (%)	9.22 ± 0.37	12.50 ± 0.40	11.96 ± 0.97	10.55 ± 0.74	15.89 ± 2.05	13.69 ± 1.76	0.00	36.68
Species richness (number of species)	8.40 ± 0.89	3.56 ± 0.33	3.29 ± 0.40	6.15 ± 0.54	3.80 ± 0.57	3.14 ± 0.61	0.00	19.00
Abundance (number of individuals.m^{-2})	84.18 ± 20.02	80.35 ± 23.80	85.86 ± 39.99	107.00 ± 28.22	46.35 ± 12.49	49.19 ± 14.55	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.27 ± 0.06	0.37 ± 0.11	0.29 ± 0.07	0.72 ± 0.20	0.14 ± 0.03	0.30 ± 0.10	0.00	6.95
	SHALLOW SUBTIDAL							
	2004	2006	2010		2011		minimum	maximum
Beach elevation (m versus MLW)	-3.90 ± 0.26	-2.35 ± 0.26	-1.41 ± 0.23	-1.28 ± 0.30	-1.64 ± 0.15	-2.01 ± 0.12	0.00	5.41
Median grain size (µm)	157.29 ± 1.57	180.88 ± 2.40	195.14 ± 3.63	201.28 ± 5.52	165.33 ± 10.49	168.74 ± 11.33	175.00	464.00
Silt fraction (%)	1.22 ± 0.18	5.02 ± 1.21	1.06 ± 0.66	1.95 ± 0.99	9.58 ± 4.34	7.72 ± 4.69	0.00	2.25
Total organic matter (%)	0.22 ± 0.03	0.00 ± 0.00	2.39 ± 0.30	1.15 ± 0.09	2.97 ± 0.47	1.96 ± 0.31	0.00	1.84
Total organic carbon (%)	0.21 ± 0.03	1.84 ± 0.07	0.34 ± 0.15	0.30 ± 0.11	0.43 ± 0.11	0.27 ± 0.07	0.00	19.00
Carbonate content (%)	13.41 ± 0.44	13.78 ± 0.25	10.36 ± 0.52	11.47 ± 0.58	14.58 ± 1.23	13.09 ± 1.04	0.00	36.68
Species richness (number of species)	9.33 ± 0.98	7.00 ± 0.59	4.79 ± 0.50	11.20 ± 1.40	5.80 ± 0.70	9.67 ± 0.91	0.00	19.00
Abundance (number of individuals.m^{-2})	34.65 ± 5.92	32.40 ± 2.52	30.22 ± 7.51	373.29 ± 98.08	46.65 ± 7.10	64.76 ± 10.25	0.00	3988.75
Biomass (g AFDW.m^{-2})	4.13 ± 3.69	2.81 ± 0.49	2.64 ± 1.28	23.42 ± 9.59	2.00 ± 0.74	1.20 ± 0.36	0.00	6.95

e

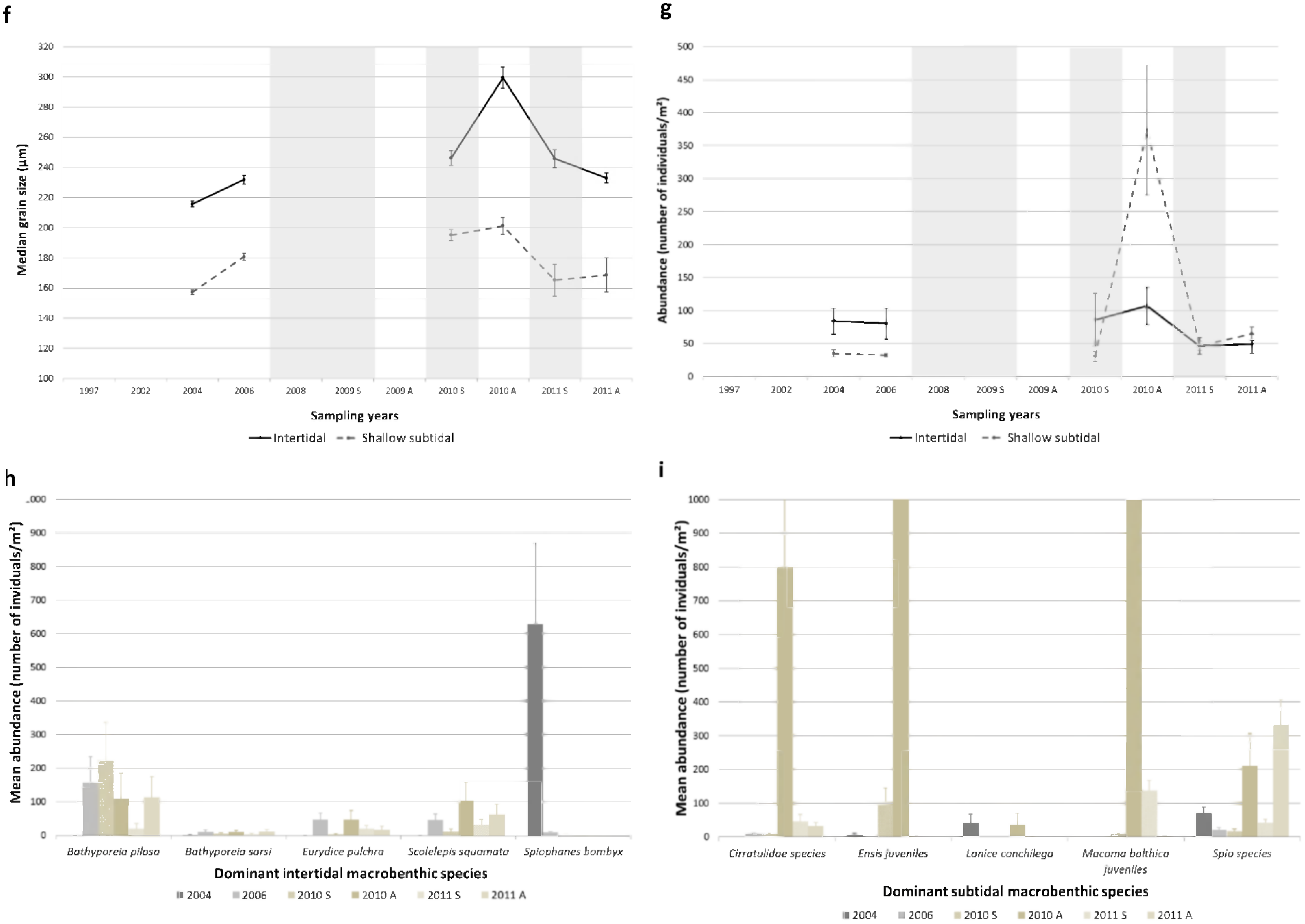
Species list Oostende-East

Intertidal

Abra alba
Actinaria species
Angulus fabula
Atylus swammerdami
Bathyporeia elegans
Bathyporeia guilliamsoniana
Bathyporeia pelagica
Bathyporeia pilosa
Bathyporeia sarsi
Capitella capitata
Chaetozone setosa
Diastylis bradyi
Diastylis rathkei
Diogenes pugilator
Donax vittatus juveniles
Ensis directus
Ensis juveniles
Eteone longa
Eumida sanguinea
Eurydice affinis
Eurydice pulchra
Haustorius arenarius
Lagis koreni
Lanice conchilega
Macoma balthica
Macoma balthica juveniles
Magelona species
Microprotopus maculatus
Mysella bidentata
Mytilus edulis
Nemertea species
Nephtys caeca
Nephtys cirrosa
Nephtys hombergii
Nephtys juveniles
Oligochaeta species
Owenia fusiformis
Phyllodoce species
Poecilochaetus serpens
Pontocrates altamarinus
Sagitta species
Scoloplos armiger
Scolelepis squamata
Spiophanes bombyx
Spio species
Spisula subtruncata
Talitrus saltator
Urothoe brevicornis
Urothoe poseidonis

Shallow subtidal

Abra alba
Actinaria species
Ampharetidae species
Angulus fabula
Angulus tenuis
Aphelochaeta marioni
Arenicola marina
Atylus falcatus
Atylus swammerdami
Autolytus species
Bathyporeia guilliamsoniana
Bathyporeia pelagica
Bathyporeia sarsi
Capitella capitata
Chaetozone setosa
Cirratulidae species
Cumopsis goodsir
Diastylis bradyi
Diastylis rathkei
Diogenes pugilator
Donax vittatus
Donax vittatus juveniles
Ensis directus
Ensis juveniles
Ensis species
Eteone longa
Eumida sanguinea
Eumida juveniles
Eurydice pulchra
Gammarus species
Glycera species
Haustorius arenarius
Heteromastus filiformis
Lagis koreni
Lanice conchilega
Macoma balthica
Macoma balthica juveniles
Magelona species
Malmgreniella juveniles
Microprotopus maculatus
Mysella bidentata
Mytilus edulis
Nemertea species
Nephtys assimilis
Nephtys caeca
Nephtys cirrosa
Nephtys hombergii
Nephtys juveniles
Nephtys species
Nereis longissima
Notomastus latericeus
Oligochaeta species
Owenia fusiformis
Pariambus typicus
Pholoe minuta
Phyllodoce species
Poecilochaetus serpens
Polydora species
Pontocrates altamarinus
Pseudocuma longicornis
Pygospio elegans
Sagitta species
Scoloplos armiger
Spiophanes bombyx
Spio species
Urothoe poseidonis
Urothoe pulchella


f
Median grain size (µm)
320
300
280
260
240
220
200
180
160
140
120
100
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal
g
Abundance (number of individuals/m²)
500
450
400
350
300
250
200
150
100
50
0
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal
h
Mean abundance (number of inviduals/m²)
1000
900
800
700
600
500
400
300
200
100
0
Bathyporeia pilosa
Bathyporeia sarsi
Eurydice pulchra
Scolelepis squamata
Spiophanes bombyx
Dominant intertidal macrobenthic species
2004
2006
2010 S
2010 A
2011 S
2011 A
i
Mean abundance (number of inviduals/m²)
1000
900
800
700
600
500
400
300
200
100
0
Cirratulidae species
Ensis juveniles
Lanice conchilega
Macoma balthica juveniles
Spio species
Dominant subtidal macrobenthic species
2004
2006
2010 S
2010 A
2011 S
2011 A

Bredene

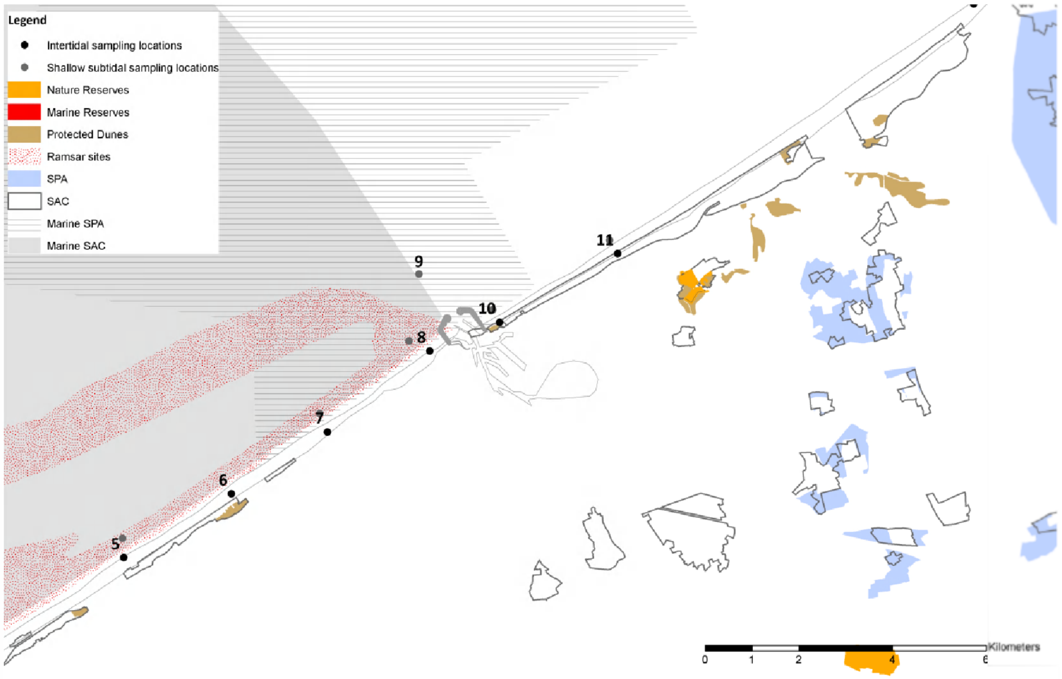
The beach of Bredene is situated 3 km west of Oostende-East (location 11 in all figures). It does not hold any conservation status but it is located in front of a Special Area of Conservation (SAC) (a). The Provincial Spatial Implementation Plan (PSIP) of Bredene covers the intertidal zone of this beach (b).

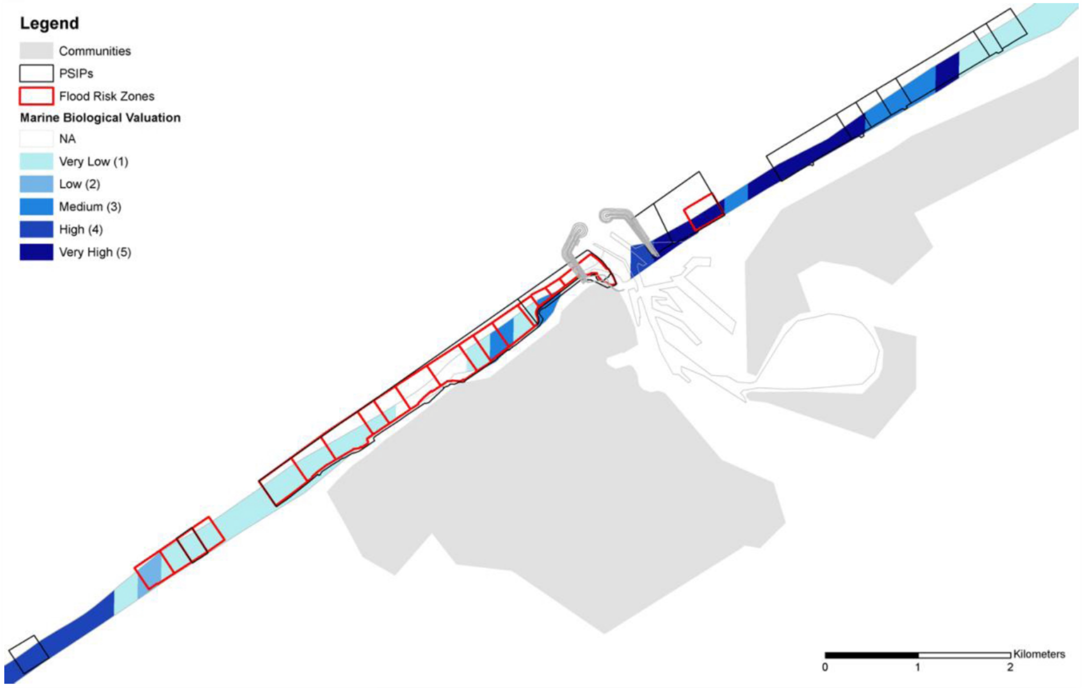
According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Bredene has a medium biological value (b). There is no coastal flood risk at the beach of Bredene. The dune area at the backshore provides the hinterland with sufficient coastal protection against flooding.

All the following findings (c, d, f, g, h and i) are based on 60 intertidal and 60 shallow subtidal samples taken in spring (S) and autumn (A) of 2010 and 2011 (Vanden Eede et al. 2011 and 2013):

Bredene	2010		2011	
	S	A	S	A
Intertidal zone	15	15	15	15
Shallow subtidal zone	15	15	15	15

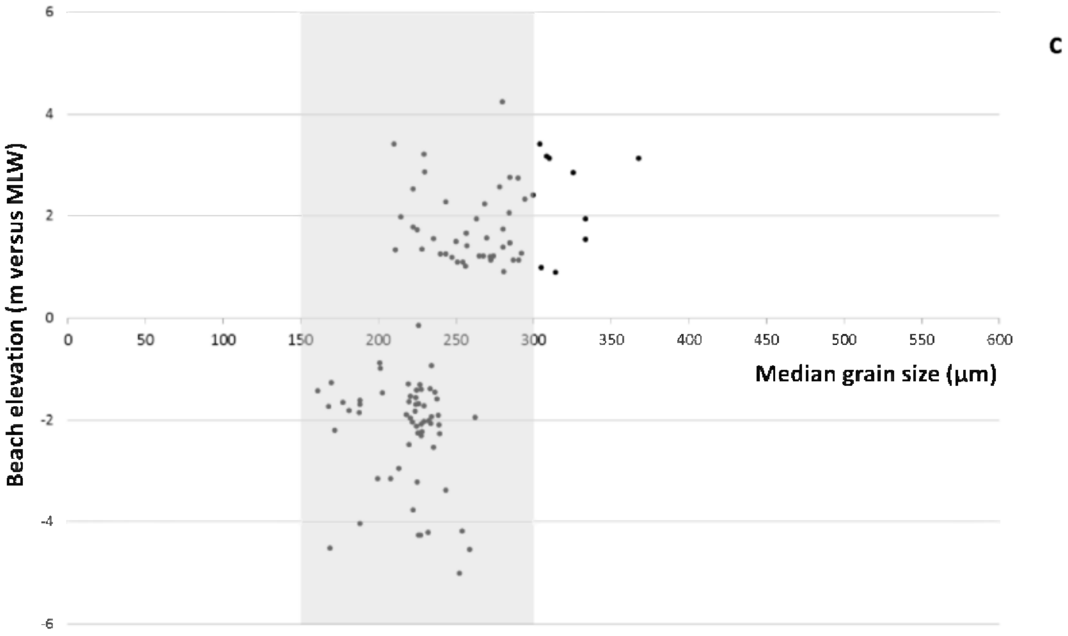
In total, 18 intertidal and 40 shallow subtidal species were identified (e). All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches (d). The median grain size


a


b

decreased from MHW (367.79 µm) to MLW (210.04 µm) while the shallow subtidal values fluctuated between 150 and 250 µm (c). Nine intertidal samples had a median grain size higher than 300 µm (1 in spring 2010; 4 in spring 2011; 4 in autumn 2011). The intertidal median grain size was always higher than the shallow subtidal values while the opposite trend could be observed for the abundance values. Normally, spring abundance values were lower than autumn values but the shallow subtidal value of autumn 2011 is even lower than the value found in spring (f and g).

In the intertidal zone, *Bathyporeia sarsi* was the most dominant species in spring and autumn 2010 and *Scolelepis squmata* in spring and autumn 2011. Only in spring 2011, *Spiophanes bombyx* was found in very low abundance in the intertidal zone of Bredene. In the shallow subtidal zone, *Ensis* juveniles were the most dominant species in spring 2010. In autumn 2011, *Cirratulidae* species and *Ensis* juveniles were found in equally large abundances while in spring 2011, only *Cirratulidae* species was still abundantly present. *Spio* species and *Cirratulidae* species dominated the shallow subtidal zone of Bredene in autumn 2011 (i).

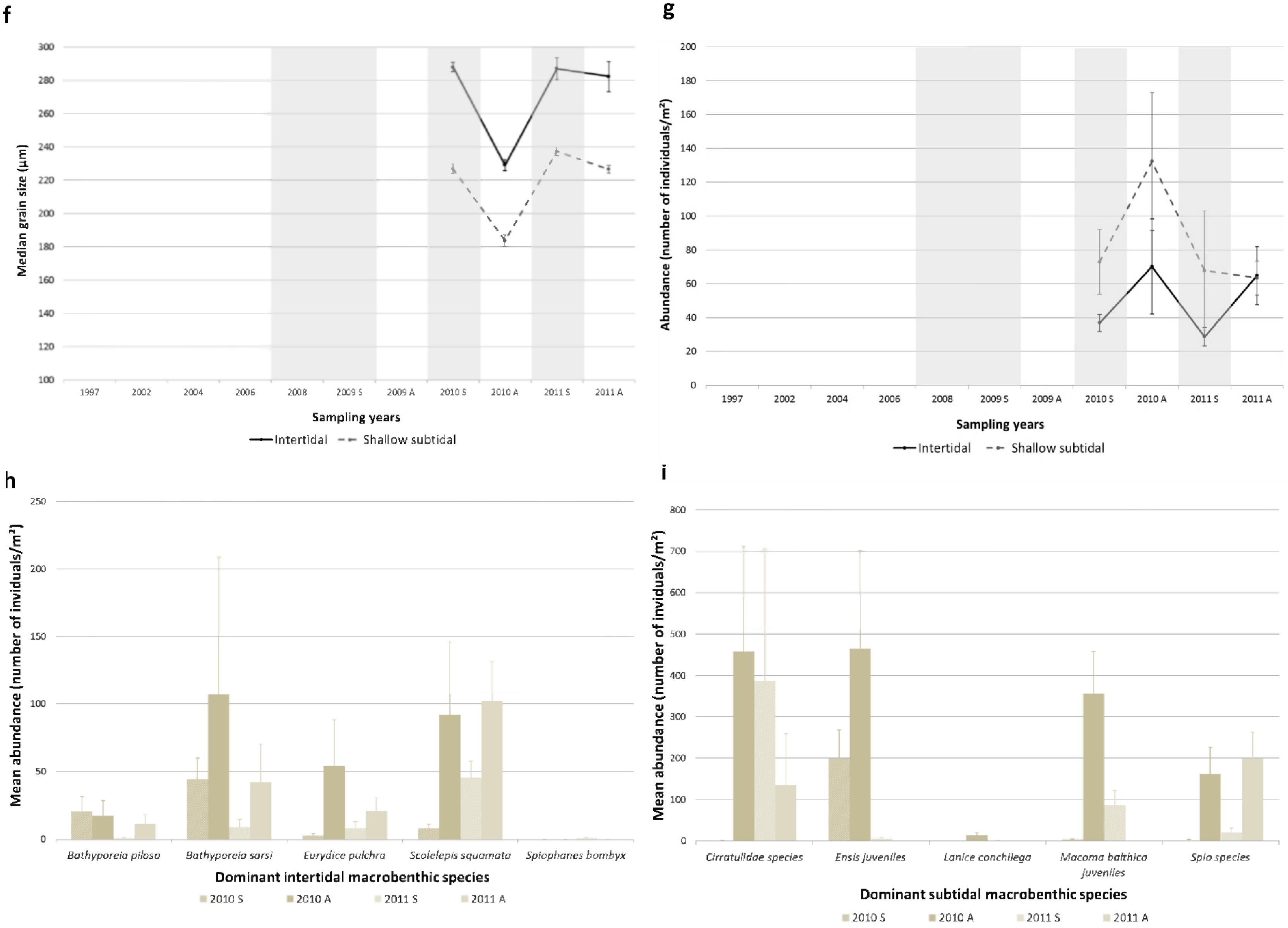


d

Bredene	**INTERTIDAL**					
	2010		2011		minimum	maximum
Beach elevation (m versus MLW)	1.76 ± 0.18	2.00 ± 0.19	1.85 ± 0.28	1.89 ± 0.20	0.00	5.41
Median grain size (µm)	287.92 ± 2.80	229.02 ± 3.22	287.00 ± 6.52	282.42 ± 8.98	175.00	464.00
Silt fraction (%)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00	2.25
Total organic matter (%)	0.50 ± 0.02	0.49 ± 0.02	0.39 ± 0.03	0.57 ± 0.04	0.00	1.84
Total organic carbon (%)	0.14 ± 0.02	0.06 ± 0.03	0.11 ± 0.01	0.08 ± 0.01	0.00	19.00
Carbonate content (%)	9.31 ± 0.59	9.04 ± 0.83	10.26 ± 1.30	8.67 ± 0.75	0.00	36.68
Species richness (number of species)	3.09 ± 0.36	4.57 ± 0.51	3.50 ± 0.59	4.20 ± 0.60	0.00	19.00
Abundance (number of individuals.m^{-2})	36.85 ± 5.10	70.18 ± 28.06	28.75 ± 5.45	64.82 ± 17.17	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.28 ± 0.09	0.46 ± 0.14	0.20 ± 0.04	0.50 ± 0.11	0.00	6.95
	SHALLOW SUBTIDAL					
	2010		2011		minimum	maximum
Beach elevation (m versus MLW)	-2.02 ± 0.30	-2.02 ± 0.27	-2.41 ± 0.26	-2.50 ± 0.25	0.00	5.41
Median grain size (µm)	226.84 ± 2.86	183.66 ± 3.56	237.30 ± 2.60	226.51 ± 2.35	175.00	464.00
Silt fraction (%)	0.58 ± 0.58	0.54 ± 0.53	0.09 ± 0.09	0.00 ± 0.00	0.00	2.25
Total organic matter (%)	1.16 ± 0.07	0.88 ± 0.10	1.33 ± 0.10	0.80 ± 0.05	0.00	1.84
Total organic carbon (%)	0.12 ± 0.02	0.07 ± 0.03	0.18 ± 0.02	0.11 ± 0.01	0.00	19.00
Carbonate content (%)	7.11 ± 0.67	8.19 ± 0.62	7.10 ± 0.64	6.32 ± 0.22	0.00	36.68
Species richness (number of species)	4.87 ± 0.34	10.80 ± 0.81	6.87 ± 0.72	10.20 ± 0.79	0.00	19.00
Abundance (number of individuals.m^{-2})	72.88 ± 19.06	132.22 ± 40.77	67.79 ± 35.18	63.45 ± 10.03	0.00	3988.75
Biomass (g AFDW.m^{-2})	5.65 ± 1.75	23.37 ± 7.75	2.99 ± 0.64	9.65 ± 3.61	0.00	6.95

e

Species list Bredene		
Intertidal	Shallow subtidal	
Bathyporeia pelagica	*Abra alba*	*Haustorius arenarius*
Bathyporeia pilosa	*Ampharetidae species*	*Heteromastus filiformis*
Bathyporeia sarsi	Atylus falcatus	*Jassa species*
Cumopsis goodsir	*Bathyporeia pelagica*	*Lanice conchilega*
Ensis juveniles	*Bathyporeia sarsi*	*Macoma balthica*
Eteone longa	*Capitella capitata*	*Macoma balthica juveniles*
Eurydice affinis	*Cirratulidae species*	*Magelona species*
Eurydice pulchra	*Cumopsis goodsir*	*Nephtys cirrosa*
Haustorius arenarius	*Diastylis bradyi*	*Nephtys hombergii*
Macoma balthica	*Diastylis rathkei*	*Oligochaeta species*
Macoma balthica juveniles	*Diogenes pugilator*	*Owenia fusiformis*
Nemertea species	*Donax vittatus*	*Pholoe minuta*
Nephtys caeca	*Donax vittatus juveniles*	*Phyllodoce species*
Nephtys cirrosa	*Ensis juveniles*	*Pontocrates altamarinus*
Owenia fusiformis	*Ensis species*	*Pygospio elegans*
Scolelepis squamata	*Eteone flava*	*Scoloplos armiger*
Spiophanes bombyx	*Eteone longa*	*Scolelepis squamata*
Spio species	*Eumida sanguinea*	*Spiophanes bombyx*
	Eurydice pulchra	*Spio species*
	Glycera species	*Urothoe poseidonis*


f
Median grain size (µm)
300
280
260
240
220
200
180
160
140
120
100
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal
g
Abundance (number of individuals/m²)
200
180
160
140
120
100
80
60
40
20
0
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal
h
Mean abundance (number of inviduals/m²)
250
200
150
100
50
0
Bathyporeia pilosa
Bathyporeia sarsi
Eurydice pulchra
Scolelepis squamata
Spiophanes bombyx
Dominant intertidal macrobenthic species
2010 S
2010 A
2011 S
2011 A
i
Mean abundance (number of inviduals/m²)
800
700
600
500
400
300
200
100
0
Cirratulidae species
Ensis juveniles
Lanice conchilega
Macoma balthica juveniles
Spio species
Dominant subtidal macrobenthic species
2010 S
2010 A
2011 S
2011 A

Wenduine

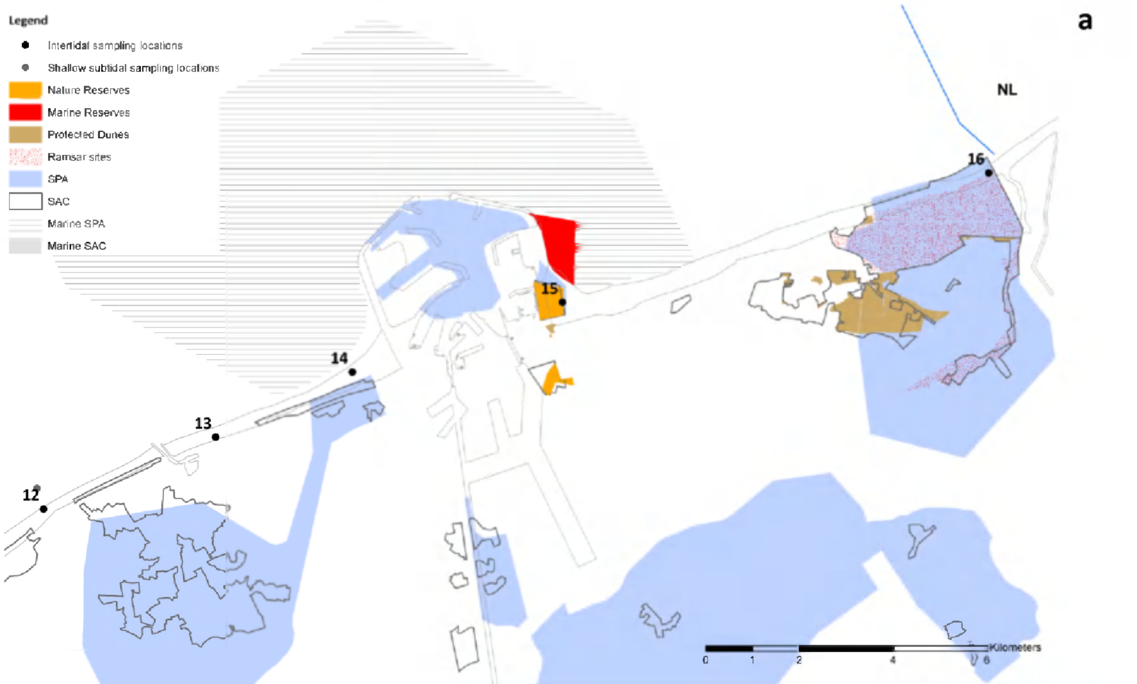
The beach of Wenduine is situated in front of the municipality of Wenduine (location 12 in all figures). It does not hold any conservation status (a). The Provincial Spatial Implementation Plan (PSIP) of De Haan covers the intertidal zone of this beach (b).

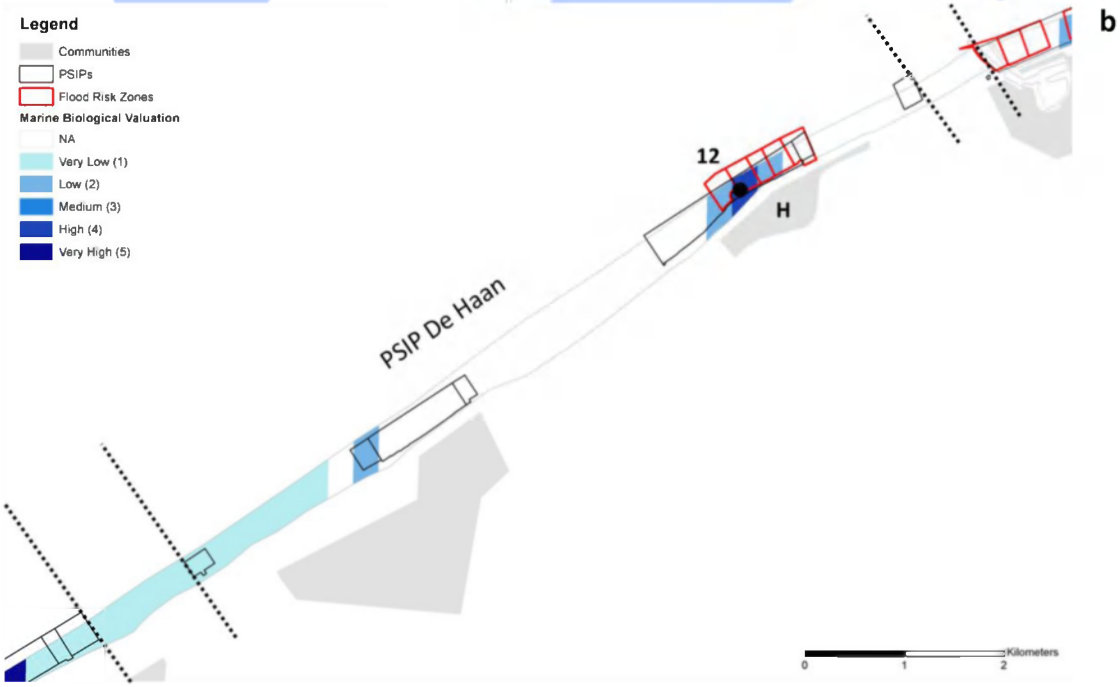
According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Wenduine has a low to medium biological value (b). The coastal flood risk in the PSIP of De Haan is exceptionally high at the beach of Wenduine as it is regarded as one of the weakest links in our coastal defence system. The beach of Wenduine will receive a beach nourishment (700 000 m³ - location H in b) in the near future. A storm wall will be built on the roundabout while a parapet structure (a curled storm wall to deviate splashing sea water) will be integrated in the sea dyke.

All the following findings (c, d, f, g, h and i) are based on 41 intertidal and 40 shallow subtidal samples taken in 2004 and 2006 (Welvaert 2005; Van Ginderdeuren et al. 2007):

Wenduine	**2004**	**2006**
Intertidal zone	15	26
Shallow subtidal zone	15	25

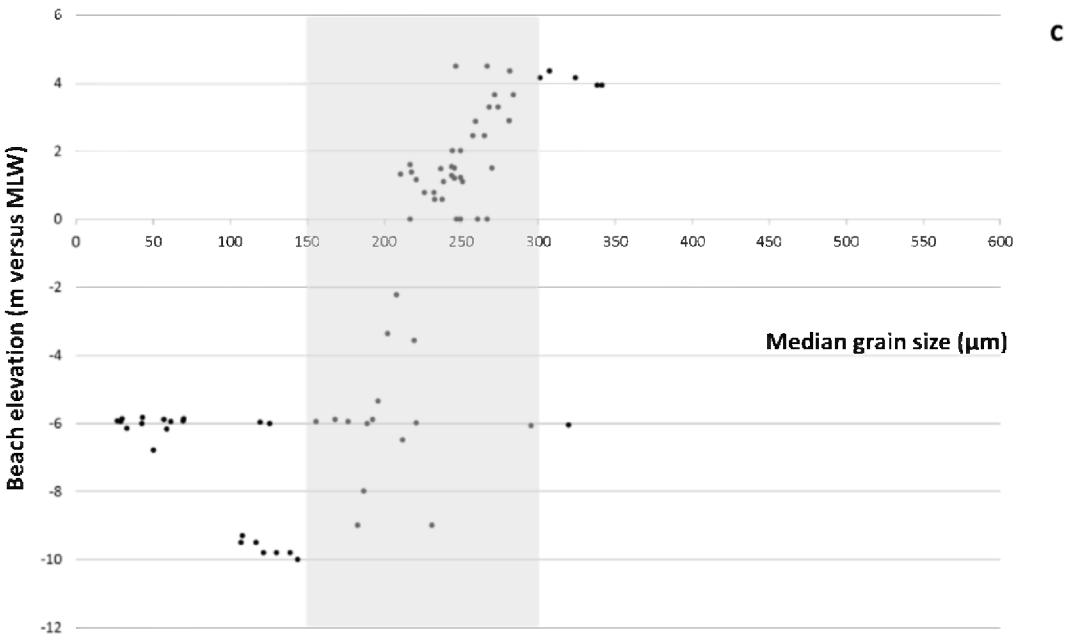
In total, 33 intertidal and 38 shallow subtidal species were identified (e). All mean measured abiotic and biotic





variables fell between the maximum and minimum values of the Belgian beaches (d). The median grain size decreased from MHW (341.53 µm) to MLW (211.00 µm) (c). Six intertidal samples, all taken in 2006, had a median grain size higher than 300 µm. More than half of the shallow subtidal samples (21 samples) had a very low median grain size (< 150 µm) (7 in 2004 and 14 in 2006). The intertidal median grain size was always higher than the shallow subtidal values while the shallow subtidal abundance was always higher than the intertidal values. In 2006, the intertidal median grain size and the abundance value were higher than in 2004 but the shallow subtidal abundance was lower than in 2004 (f and g).

In the intertidal zone, *Scolelepis squamata* was clearly the most dominant species in 2006 while *Spiophanes bombyx* was the only dominant species found in 2004. *Bathyporeia pilosa* was bearly found in the intertidal zone of Wenduine. In the shallow subtidal zone, *Spio* species was the only dominant species appearing in both studied years. It was clearly the most dominant species in 2004. In 2006, *Cirratulidae* species dominated the shallow subtidal zone of Wenduine. *Macoma balthica* juveniles were not present in the shallow subtidal samples of Wenduine.



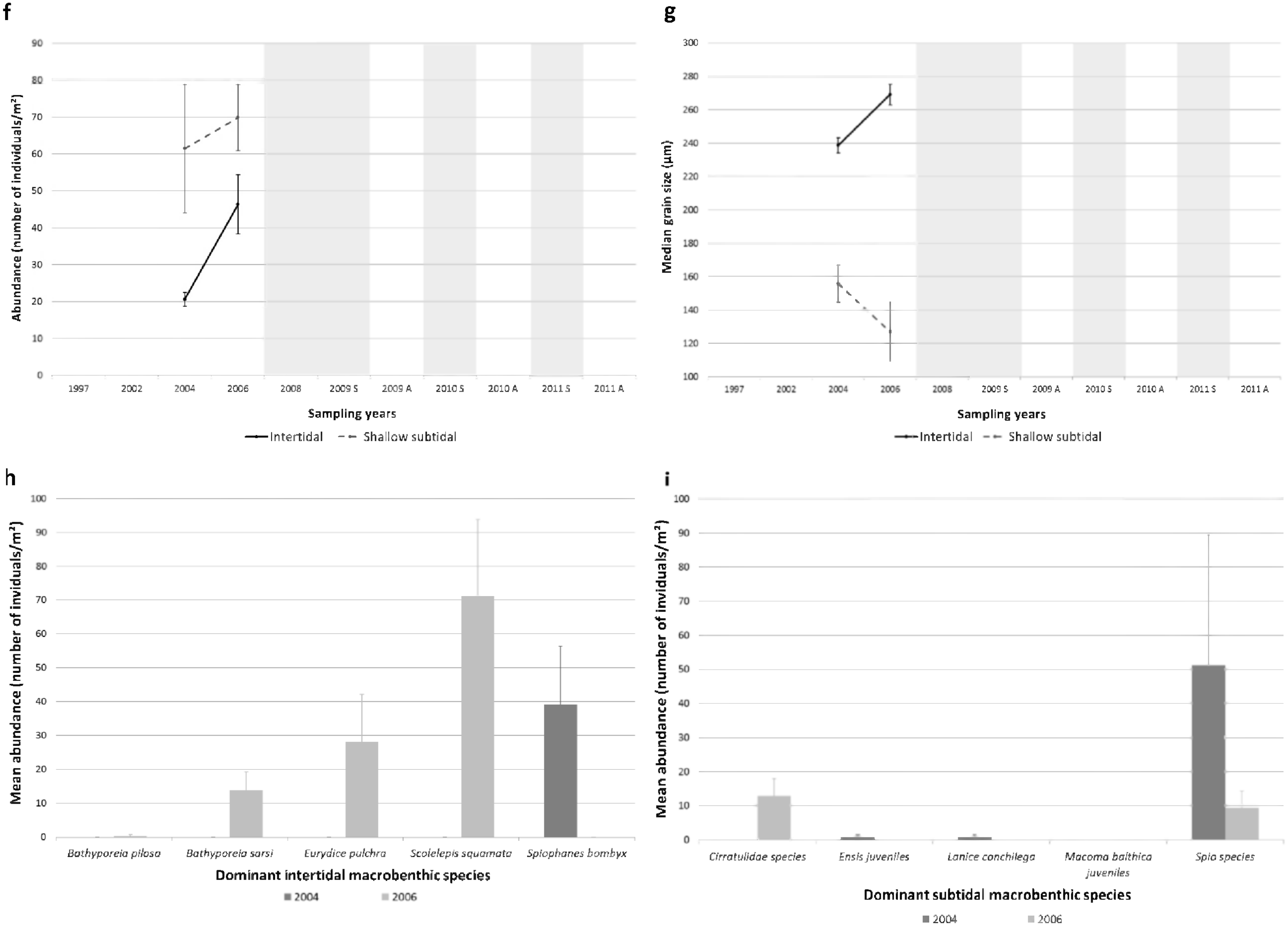
d

Wenduine	INTERTIDAL			
	2004	2006	minimum	maximum
Beach elevation (m versus MLW)	0.90 ± 0.17	2.71 ± 0.27	0.00	5.41
Median grain size (µm)	238.67 ± 4.53	269.24 ± 6.22	175.00	464.00
Silt fraction (%)	0.00 ± 0.00	0.00 ± 0.00	0.00	2.25
Total organic matter (%)	0.12 ± 0.01	0.00 ± 0.00	0.00	1.84
Total organic carbon (%)	0.11 ± 0.01	0.80 ± 0.03	0.00	19.00
Carbonate content (%)	7.19 ± 0.83	9.90 ± 0.12	0.00	36.68
Species richness (number of species)	6.07 ± 0.69	3.31 ± 0.21	0.00	19.00
Abundance (number of individuals.m^{-2})	20.64 ± 1.90	46.33 ± 8.03	0.00	3988.75
Biomass (g AFDW.m^{-2})	1.67 ± 0.61	0.32 ± 0.06	0.00	6.95
	SHALLOW SUBTIDAL			
	2004	2006	minimum	maximum
Beach elevation (m versus MLW)	-8.85 ± 0.34	-5.62 ± 0.20	0.00	5.41
Median grain size (µm)	155.75 ± 11.02	126.97 ± 17.67	175.00	464.00
Silt fraction (%)	16.37 ± 3.45	40.64 ± 5.34	0.00	2.25
Total organic matter (%)	0.98 ± 0.18	0.00 ± 0.00	0.00	1.84
Total organic carbon (%)	0.75 ± 0.14	2.12 ± 0.19	0.00	19.00
Carbonate content (%)	20.78 ± 2.57	13.99 ± 0.47	0.00	36.68
Species richness (number of species)	5.92 ± 1.37	4.16 ± 0.43	0.00	19.00
Abundance (number of individuals.m^{-2})	61.46 ± 17.41	69.83 ± 8.90	0.00	3988.75
Biomass (g AFDW.m^{-2})	1.10 ± 0.28	11.90 ± 2.39	0.00	6.95

e

Species list Wenduine

Intertidal		Shallow subtidal	
Angulus fabula	*Nephtys juveniles*	*Abra alba*	*Macoma balthica*
Bathyporeia elegans	*Oligochaeta species*	*Actinaria species*	*Magelona species*
Bathyporeia guilliamsoniana	*Paraonis fulgens*	*Angulus fabula*	*Mysella bidentata*
Bathyporeia pelagica	*Phyllodoce species*	*Bathyporeia elegans*	*Nemertea species*
Bathyporeia pilosa	*Poecilochaetus serpens*	*Bathyporeia guilliamsoniana*	*Nephtys assimilis*
Bathyporeia sarsi	*Pontocrates arenarius*	*Bathyporeia pelagica*	*Nephtys cirrosa*
Diastylis bradyi	*Pygospio elegans*	*Bathyporeia sarsi*	*Nephtys juveniles*
Donax vittatus	*Sagitta species*	*Capitella capitata*	*Nephtys hombergii*
Eteone longa	*Scoloplos armiger*	*Chaetozone setosa*	*Notomastus latericeus*
Eurydice affinis	*Scolelepis squamata*	*Cirratulidae species*	*Oligochaeta species*
Eurydice pulchra	*Spiophanes bombyx*	*Diastylis bradyi*	*Phyllodoce species*
Haustorius arenarius	*Spio species*	*Donax vittatus*	*Pontocrates altamarinus*
Lanice conchilega	*Urothoe poseidonis*	*Ensis juveniles*	*Scoloplos armiger*
Macoma balthica		*Eteone longa*	*Scolelepis squamata*
Magelona species		*Eurydice affinis*	*Spiophanes bombyx*
Mysella bidentata		*Eurydice pulchra*	*Spio species*
Nemertea species		*Glycera species*	*Spisula subtruncata*
Nephtys assimilis		*Heteromastus filiformis*	*Urothoe poseidonis*
Nephtys caeca		*Lagis koreni*	
Nephtys cirrosa		*Lanice conchilega*	

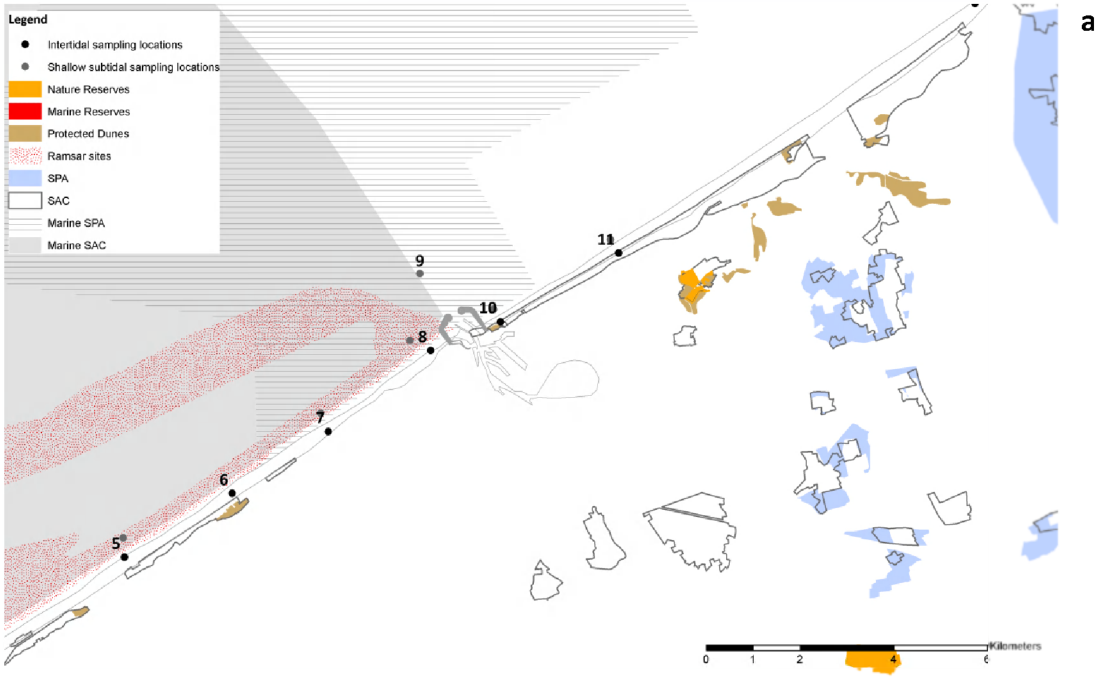

f
90
80
70
60
50
40
30
20
10
0
Abundance (number of individuals/m²)
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal
g
300
280
260
240
220
200
180
160
140
120
100
Median grain size (µm)
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
Sampling years
Intertidal
Shallow subtidal
h
100
90
80
70
60
50
40
30
20
10
0
Mean abundance (number of inviduals/m²)
Bathyporeia pilosa
Bathyporeia sarsi
Eurydice pulchra
Scolelepis squamata
Spiophanes bombyx
Dominant intertidal macrobenthic species
2004
2006
i
100
90
80
70
60
50
40
30
20
10
0
Mean abundance (number of inviduals/m²)
Cirratulidae species
Ensis juveniles
Lanice conchilega
Macoma balthica juveniles
Spio species
Dominant subtidal macrobenthic species
2004
2006

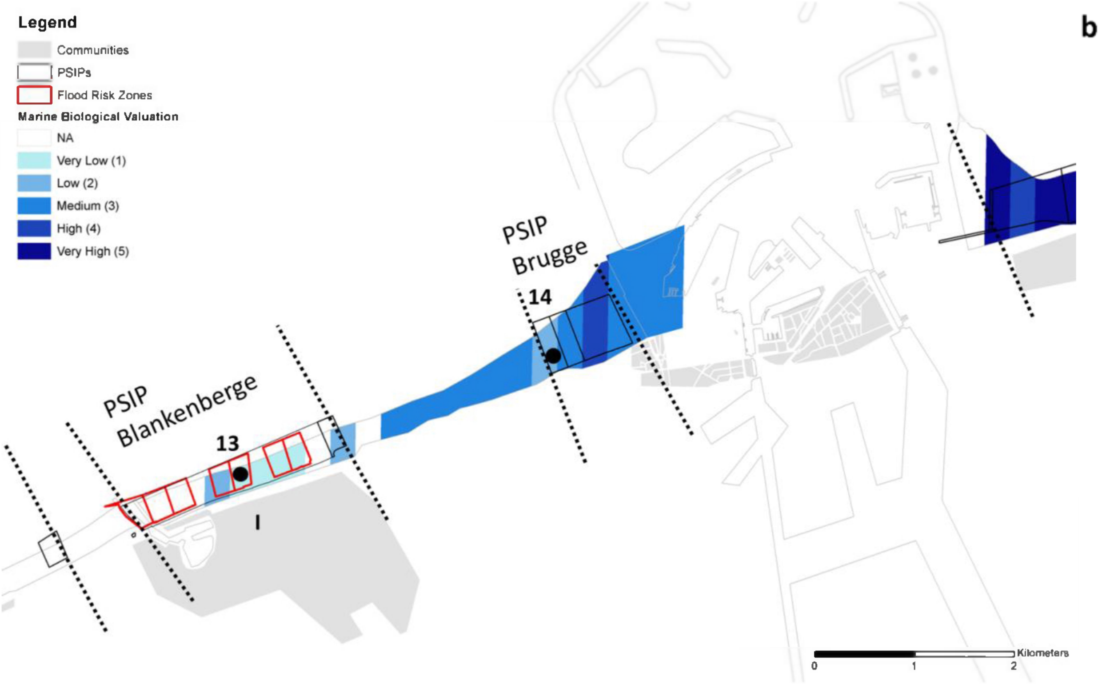
Blankenberge

The beach of Blankenberge is situated in front of the municipality of Blankenberge (location 13 in all figures). It does not hold any conservation status and is regarded as a touristic beach area (a). The Provincial Spatial Implementation Plan of Blankenberge covers the intertidal zone of this beach (b).

According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Blankenberge has a very low to low biological value (b). Due to the coastal flood risk in the PSIP of Blankenberge, the entire area will receive a beach nourishment (384 000 m³ - location I in b) in the near future.

All the following findings (c, e and f) are based on 15 intertidal samples taken in autumn 2011 (2011 A), at the beach of Blankenberge (Vanden Eede et al. 2013). In total, 14 species were identified (d). All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches (c). The median grain size decreased from MHW (268.48 µm) to MLW (221.83 µm) (e) and *Bathyporeia sarsi* was clearly the most dominant species in the intertidal zone of Blankenberge.





c

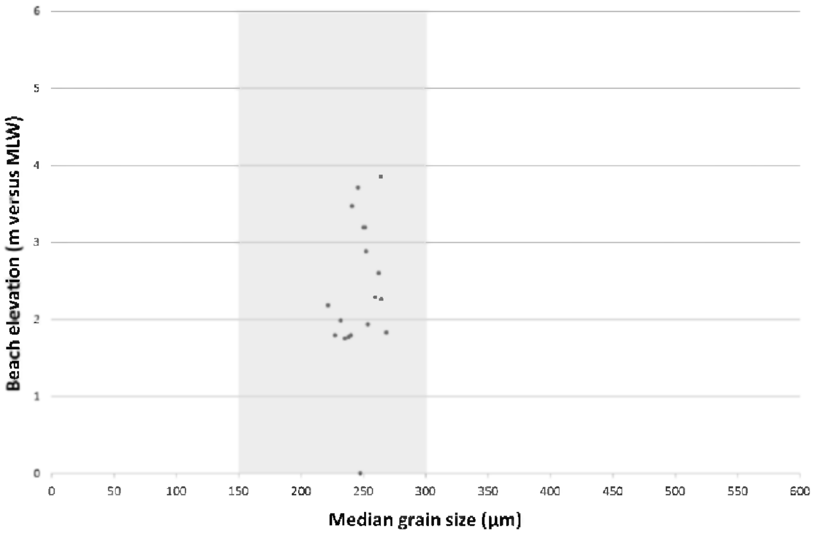
Blankenberge	INTERTIDAL		
	2011	minimum	maximum
Beach elevation (m versus MLW)	2.36 ± 0.24	0.00	5.41
Median grain size (µm)	247.63 ± 3.49	175.00	464.00
Silt fraction (%)	0.00 ± 0.00	0.00	2.25
Total organic matter (%)	0.56 ± 0.03	0.00	1.84
Total organic carbon (%)	0.06 ± 0.003	0.00	19.00
Carbonate content (%)	5.60 ± 0.37	0.00	36.68
Species richness (number of species)	3.28 ± 0.62	0.00	19.00
Abundance (number of individuals.m^{-2})	30.93 ± 6.35	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.14 ± 0.07	0.00	6.95

d

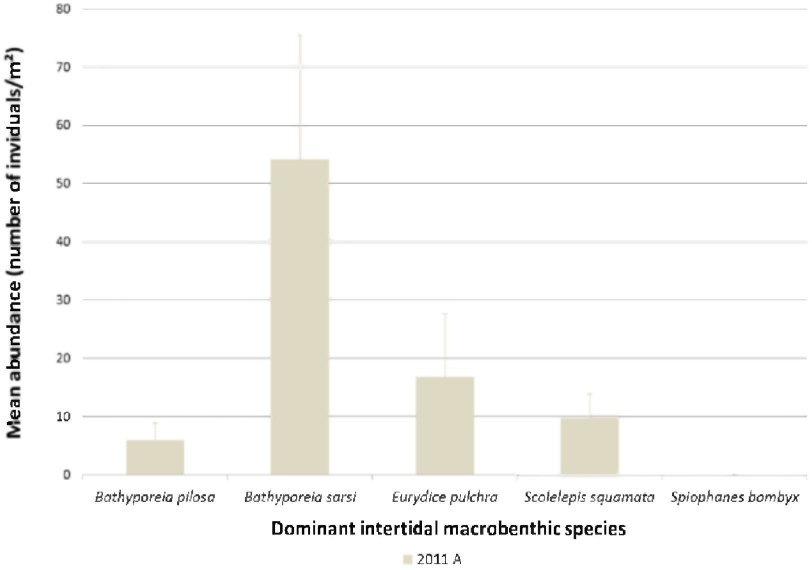
Species list Blankenberge

Bathyporeia pilosa
Bathyporeia sarsi
Cirratulidae species
Eteone longa
Eurydice affinis
Eurydice pulchra
Haustorius arenarius
Jassa herdmani
Macoma balthica
Mytilus edulis
Nemertea species
Nephtys cirrosa
Oligochaeta species
Scolelepis squamata

e



f

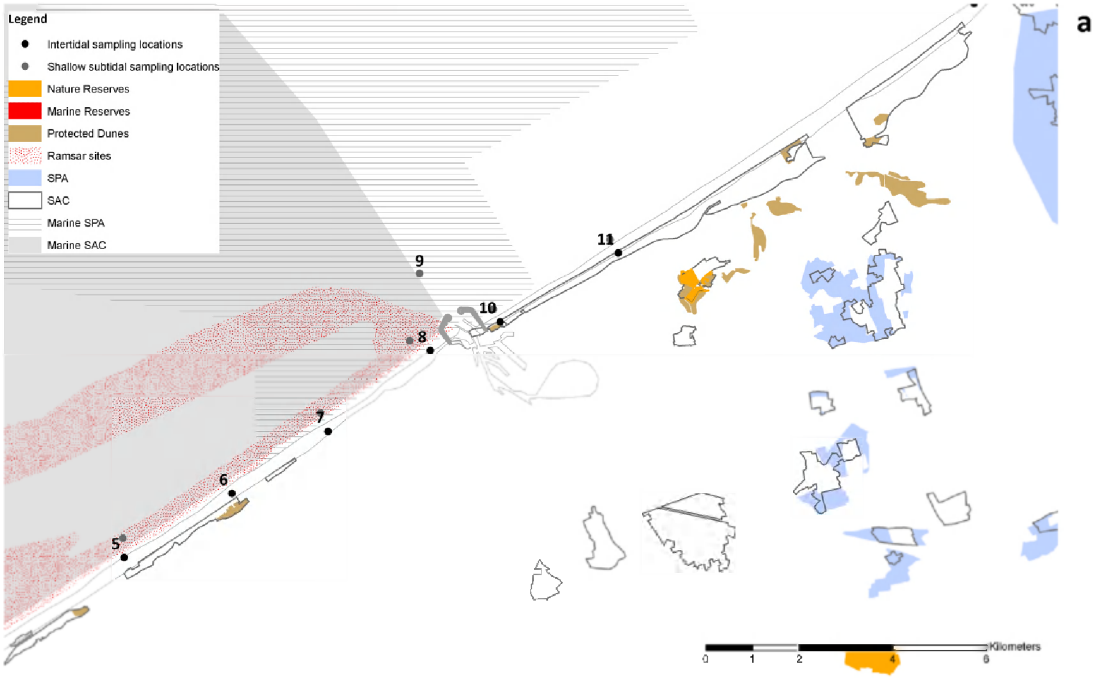


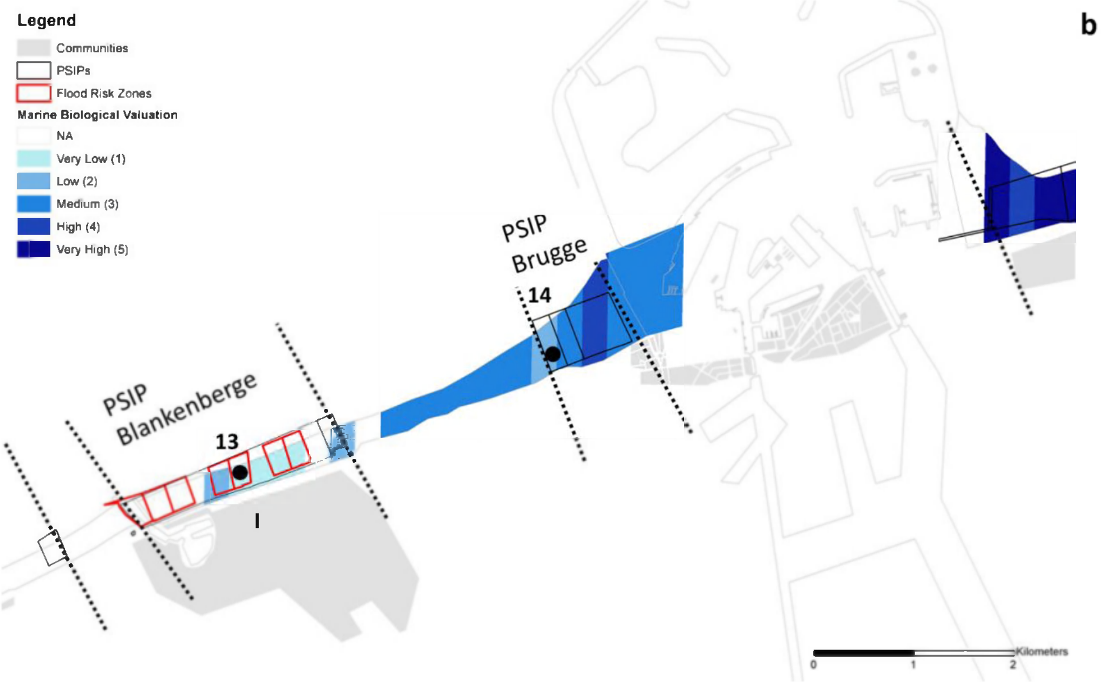
Fonteintjes

The beach of Fonteintjes is situated at the western side of the fairway to Zeebrugge (location 14 in all figures). It does not hold any conservation status but it is located in front of a Special Area of Conservation (SAC) and a Special Special Protection Area (SPA) (a). The Provincial Spatial Implementation Plan (PSIP) of Brugge covers the intertidal zone of this beach (b).

According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Fonteintjes has a low to high biological value (b). There is no coastal flood risk in the PSIP of Brugge so no coastal defence activities are planned.

All the following findings (c, e and f) are based on 22 intertidal samples taken in autumn 1997, at the beach of Fonteintjes (Speybroeck et al. 2005). In total, 14 species were identified (d). All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches (c). The median grain size decreased from MHW (265 µm) to MLW (225.30 µm) (e) and *Scolelepis squamata*, closely followed by *Bathyporeia sarsi,* was clearly the most dominant species in the intertidal zone of Fonteintjes.





c

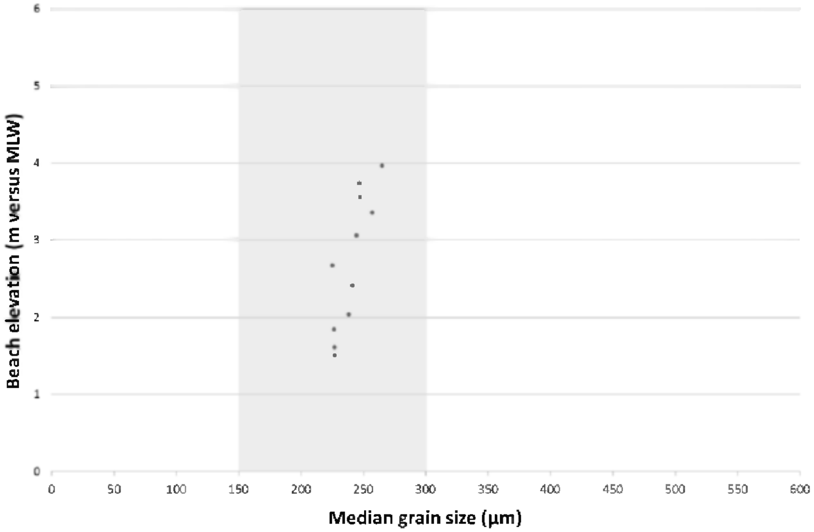
Fonteintjes	INTERTIDAL		
	1997	minimum	maximum
Beach elevation (m versus MLW)	2.70 ± 0.18	0.00	5.41
Median grain size (µm)	240.65 ± 2.78	175.00	464.00
Silt fraction (%)	0.00 ± 0.00	0.00	2.25
Total organic matter (%)	0.53 ± 0.02	0.00	1.84
Total organic carbon (%)	0.00 ± 0.00	0.00	19.00
Carbonate content (%)	0.20 ± 0.02	0.00	36.68
Species richness (number of species)	4.95 ± 0.26	0.00	19.00
Abundance (number of individuals.m^{-2})	79.21 ± 13.16	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.00 ± 0.00	0.00	6.95

d

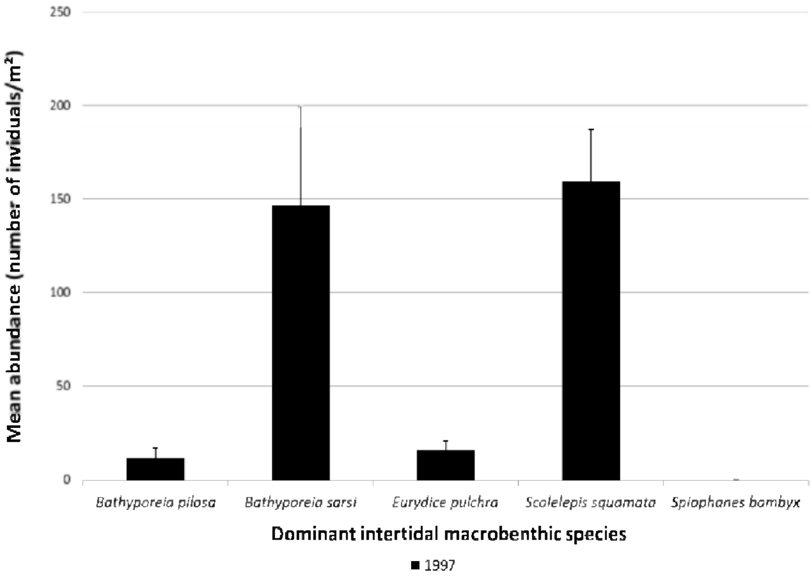
Species list Fonteintjes

Bathyporeia pelagica
Bathyporeia pilosa
Bathyporeia sarsi
Cumopsis goodsir
Eteone longa
Eurydice affinis
Eurydice pulchra
Haustorius arenarius
Macoma balthica
Nephtys cirrosa
Paraonis fulgens
Pontocrates altamarinus
Psammodrilus balanoglossoides
Pygospio elegans
Scolelepis squamata

e



f

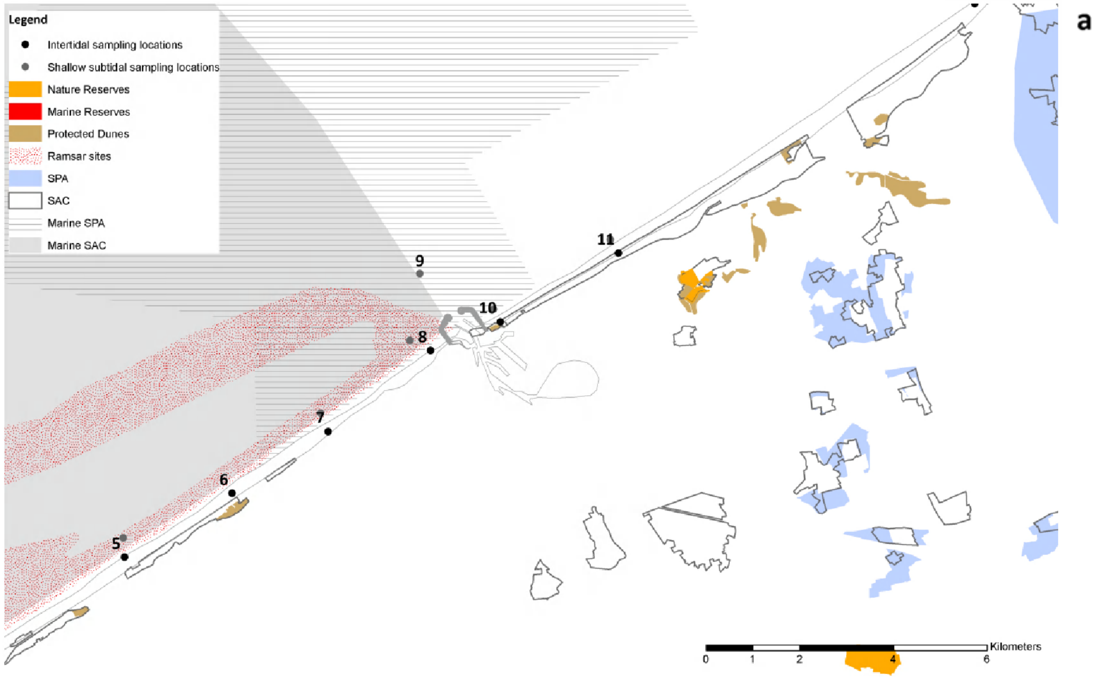


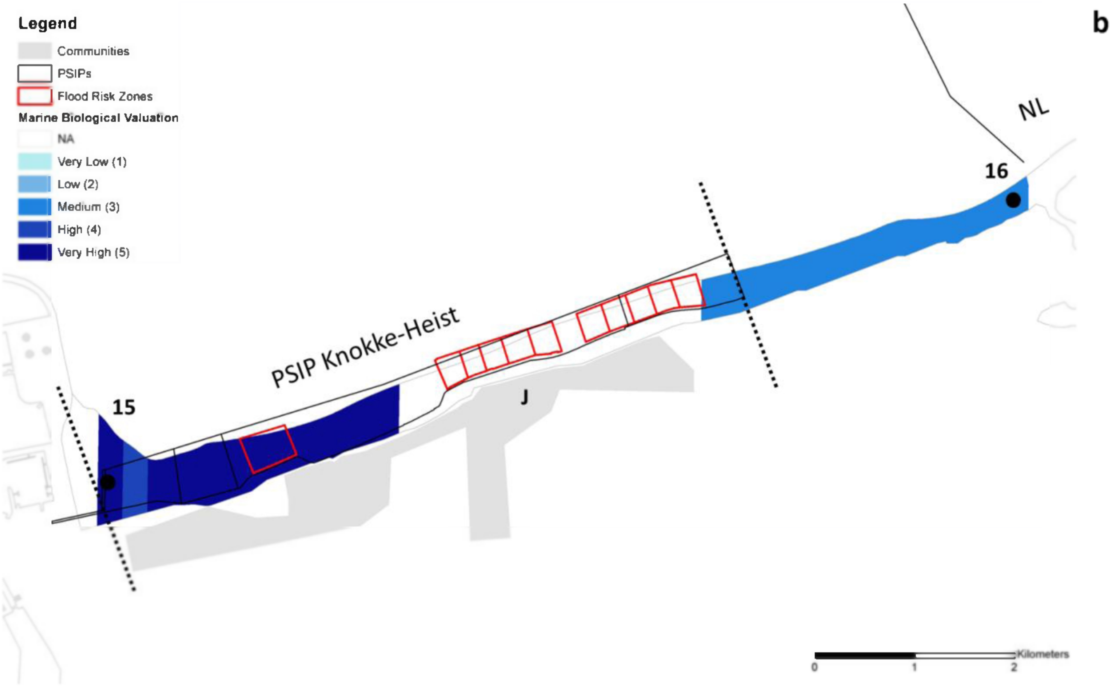
Heist

The beach of Heist is situated at the eastern side of the fairway to Zeebrugge (location 15 in all figures). It is part of the nature reserve ‘Baai van Heist’ and it is located in between protected dunes and Belgium’s only marine reserve (a). The Provincial Spatial Implementation Plan (PSIP) of Knokke-Heist covers the intertidal zone of this beach (b).

According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Heist has a high to very high biological value (b). There is coastal flood risk in the PSIP of Knokke-Heist so a huge beach nourishment (3 620 000 m³ - location J in b) is planned from March 2013 onwards. The slope of the nourished beach should cause minimal loss in intertidal beach area.

All the following findings (c, e and f) are based on 22 intertidal samples taken in autumn 1997, at the beach of Heist (Speybroeck et al. 2005). In total, 16 species were identified (d). All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches (c). The median grain size decreased from MHW (275.10 µm) to MLW (227.20 µm) (e) and *Scolelepis squamata* and *Bathyporeia sarsi* were clearly the most dominant species in the intertidal zone of Heist.





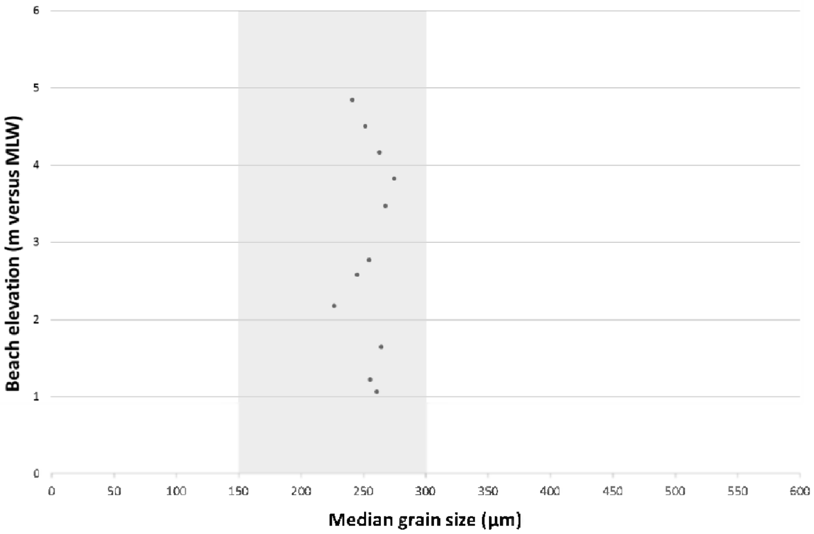
c

Heist	INTERTIDAL		
	1997	minimum	maximum
Beach elevation (m versus MLW)	2.93 ± 0.28	0.00	5.41
Median grain size (µm)	255.42 ± 2.82	175.00	464.00
Silt fraction (%)	0.40 ± 0.17	0.00	2.25
Total organic matter (%)	0.56 ± 0.03	0.00	1.84
Total organic carbon (%)	0.00 ± 0.00	0.00	19.00
Carbonate content (%)	0.33 ± 0.06	0.00	36.68
Species richness (number of species)	6.41 ± 0.48	0.00	19.00
Abundance (number of individuals.m^{-2})	191.33 ± 33.98	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.00 ± 0.00	0.00	6.95

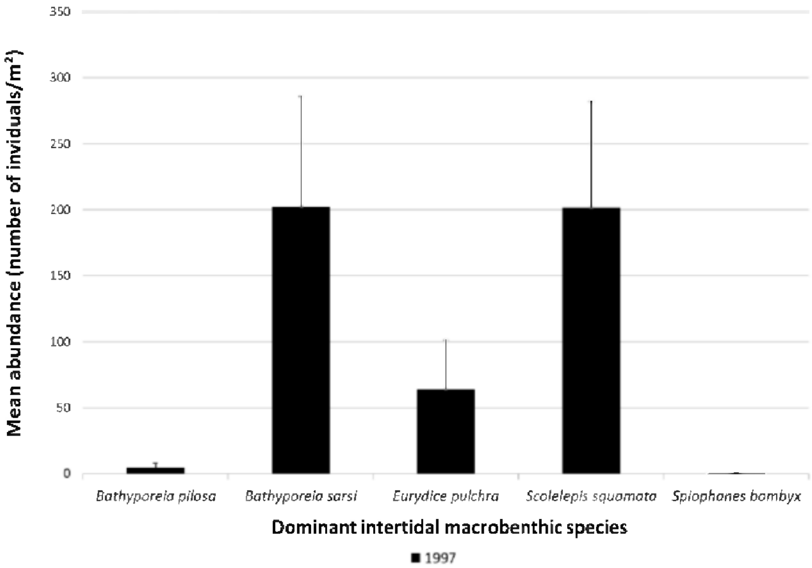
d

Species list Heist	
Bathyporeia pilosa	*Macoma balthica*
Bathyporeia sarsi	*Nephtys cirrosa*
Capitella capitata	*Nephtys hombergii*
Cerastoderma edule	*Psammodrilus balanoglossoides*
Cumopsis goodsir	*Pygospio elegans*
Eteone longa	*Scolelepis squamata*
Eurydice affinis	*Spiophanes bombyx*
Eurydice pulchra	*Spio species*

e



f

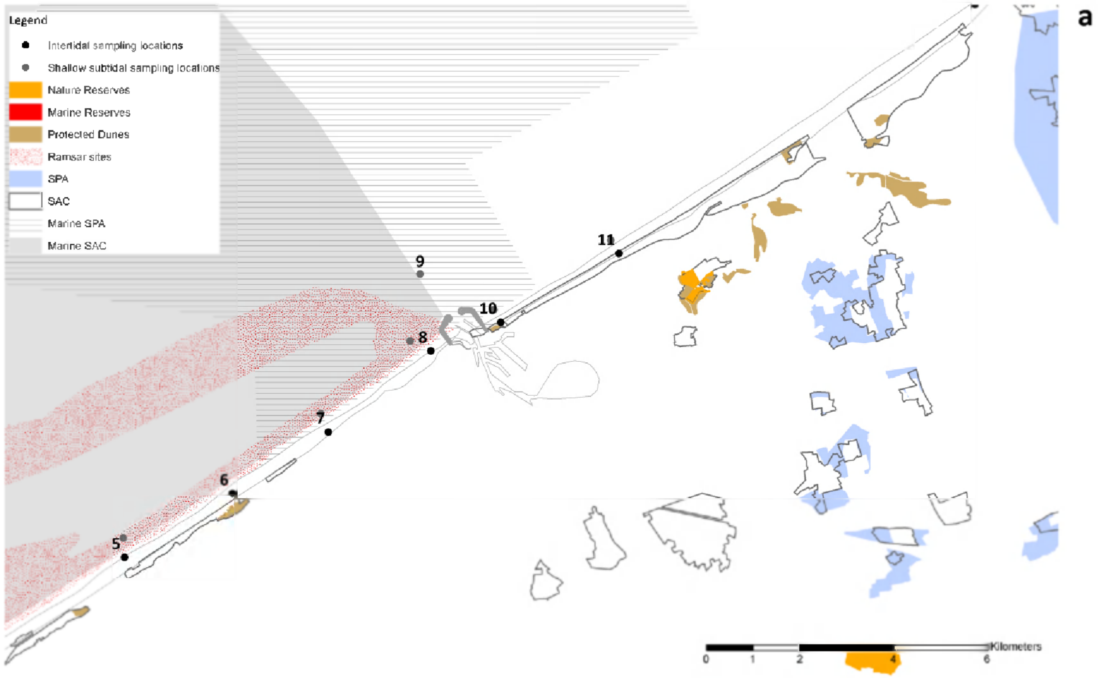


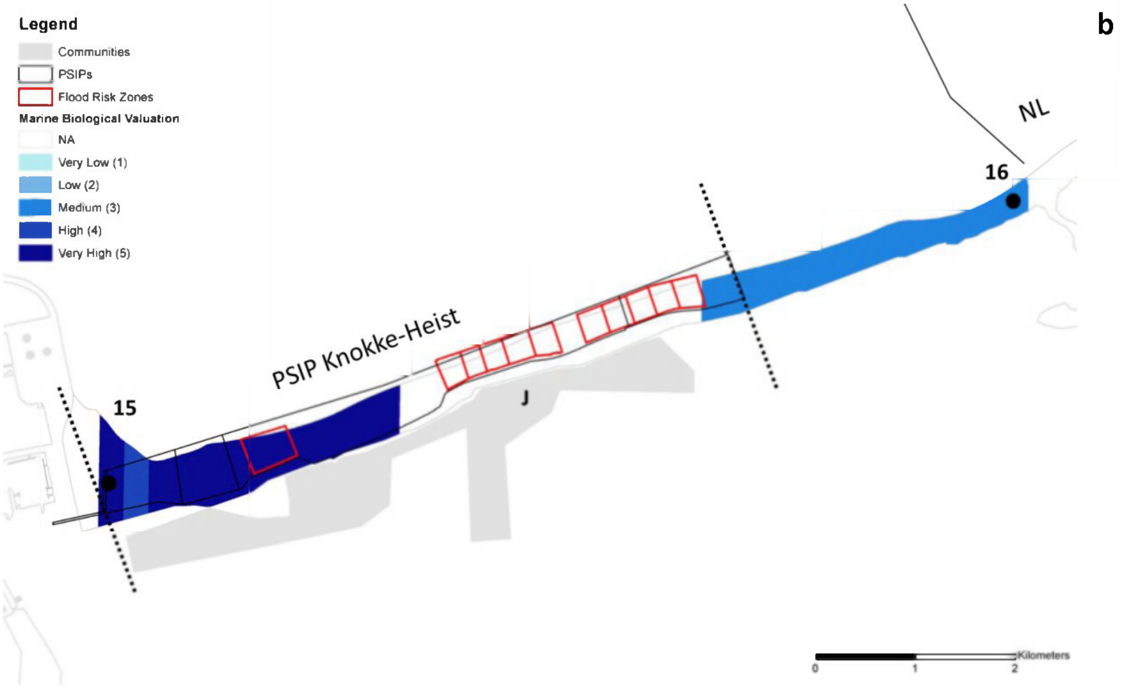
Zwinduinen en Polders

The beach of Zwinduinen en Polders is situated in the protected area ‘Zwin’ (location 16 in all figures). Zwin is a Special Area of Conservation (SAC), a Special Protection Area (SPA) and a Ramsar site (a). The intertidal zone of this beach is not covered by any Provincial Spatial Implementation Plan (PSIP) (b).

According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Zwinduinen en Polders has a medium biological value (b). There is no coastal flood risk in Zwin.

All the following findings (c, e and f) are based on 20 intertidal samples taken in autumn 1997, at the beach of Zwinduinen en Polders (Speybroeck et al. 2005). In total, 15 species were identified (d). All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches (c). The median grain size fluctuated from MHW (276.10 µm) to MLW (249.30 µm). Only two samples had a median grain size higher than 300 µm (408.30 µm) (e). *Scolelepis squamata* was clearly the most dominant species in the intertidal zone of Zwinduinen en Polders.





c

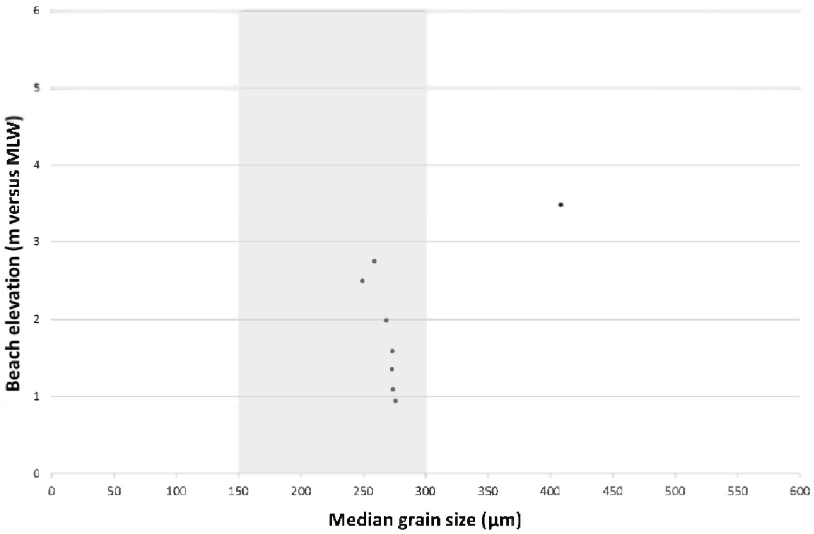
Zwinduinen en Polders	INTERTIDAL		
	1997	minimum	maximum
Beach elevation (m versus MLW)	1.96 ± 0.19	0.00	5.41
Median grain size (µm)	285.15 ± 10.93	175.00	464.00
Silt fraction (%)	0.00 ± 0.00	0.00	2.25
Total organic matter (%)	0.64 ± 0.05	0.00	1.84
Total organic carbon (%)	0.00 ± 0.00	0.00	19.00
Carbonate content (%)	1.29 ± 0.62	0.00	36.68
Species richness (number of species)	5.75 ± 0.41	0.00	19.00
Abundance (number of individuals.m^{-2})	67.21 ± 15.27	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.00 ± 0.00	0.00	6.95

d

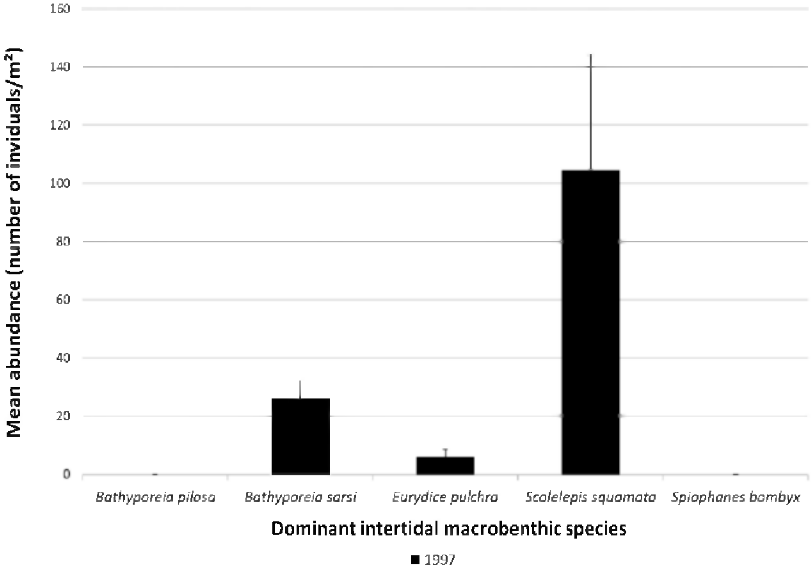
Species list Zwinduinen en Polders

Bathyporeia pelagica
Bathyporeia sarsi
Capitella capitata
Eteone longa
Eurydice pulchra
Haustorius arenarius
Nephtys cirrosa
Nephtys hombergii
Ophelia rathkei
Paraonis fulgens
Pontocrates arenarius
Psammodrilus balanoglossoides
Pygospio elegans
Scolelepis squamata
Spio species

e



f



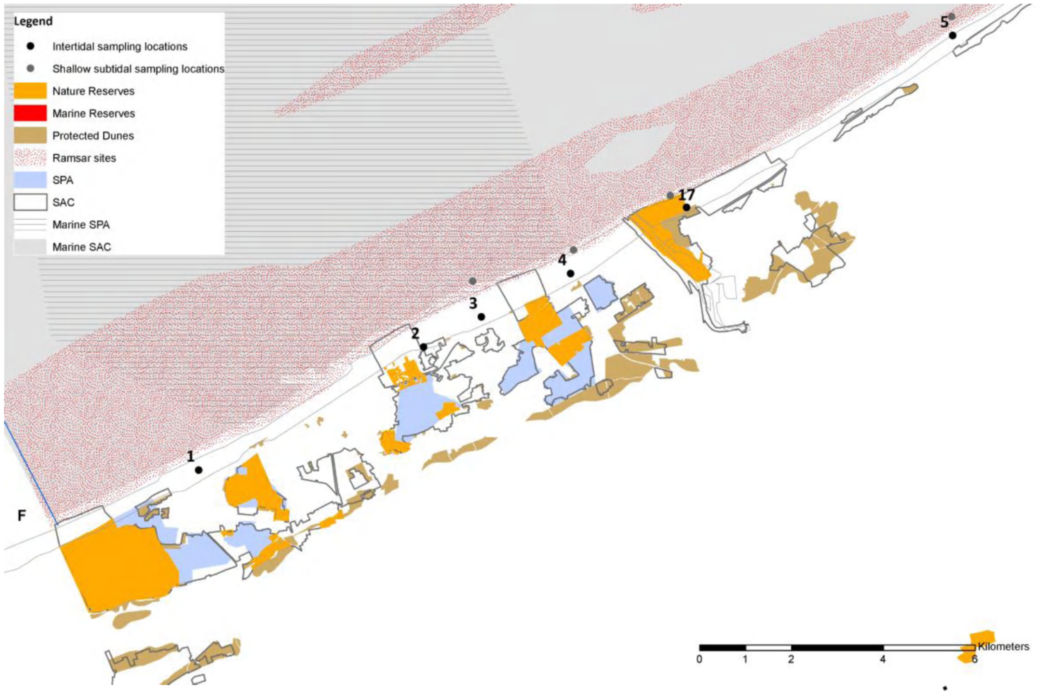
Lombardsijde

The beach of Lombardsijde is situated at the eastern side of the fairway to Nieuwpoort (location 17 in all figures). The whole beach is part of a Special Area of Conservation (SAC). The western part falls within the nature reserve 'IJzermonding' and is located in front of a Special Protection Area (SPA) and protected dunes (a). The eastern part falls within a military domain. The intertidal zone of this beach is not covered by any Provincial Spatial Implementation Plan (PSIP) (b).

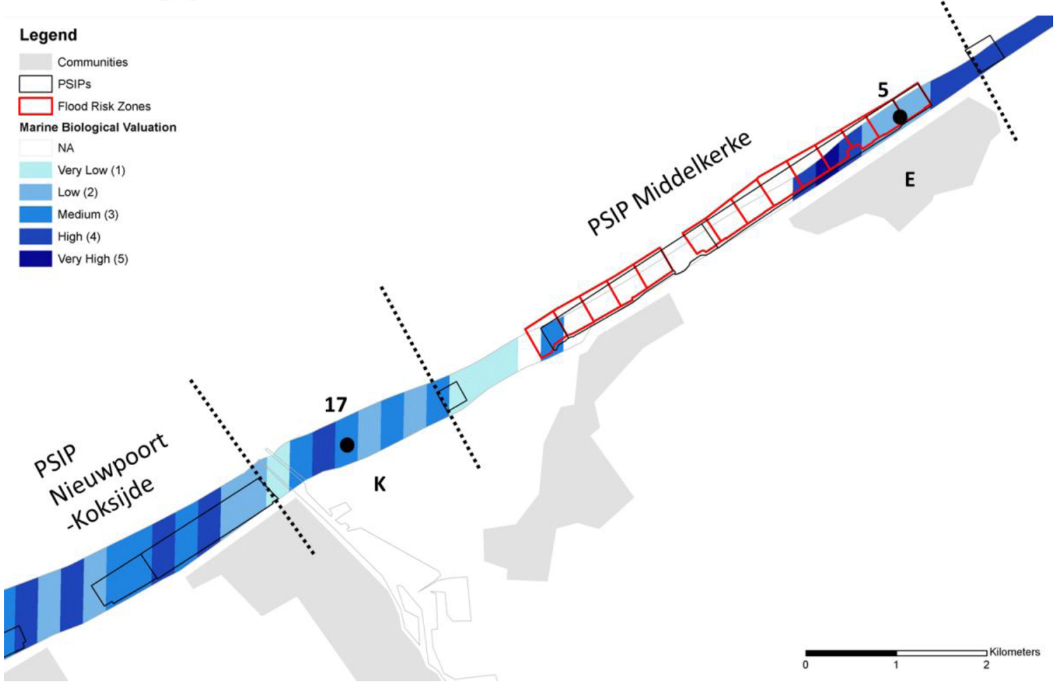
According to the marine biological valuation map of the shallow Belgian coastal zone, the beach of Lombardsijde has a low to high biological value (b). There is no coastal flood risk at the beach of Lombardsijde. However, this beach was nourished from March until September 2009. Approximately 650 000 m³ of sand was deposited on top of the beach over a distance of around 1 200 m. The nourished sand had a grain size between 200 and 250 µm and originated from the new shipping lane to Oostende.

All the following findings (c, d, f, g, h and i) are based on 255 intertidal and 239 shallow subtidal samples taken before (2004, 2006, 2008) (Welvaert 2005; Van Ginderdeuren et al. 2007; Vanden Eede et al. 2008), during (spring (S) and autumn (A) 2009) (Vanden Eede

a



b



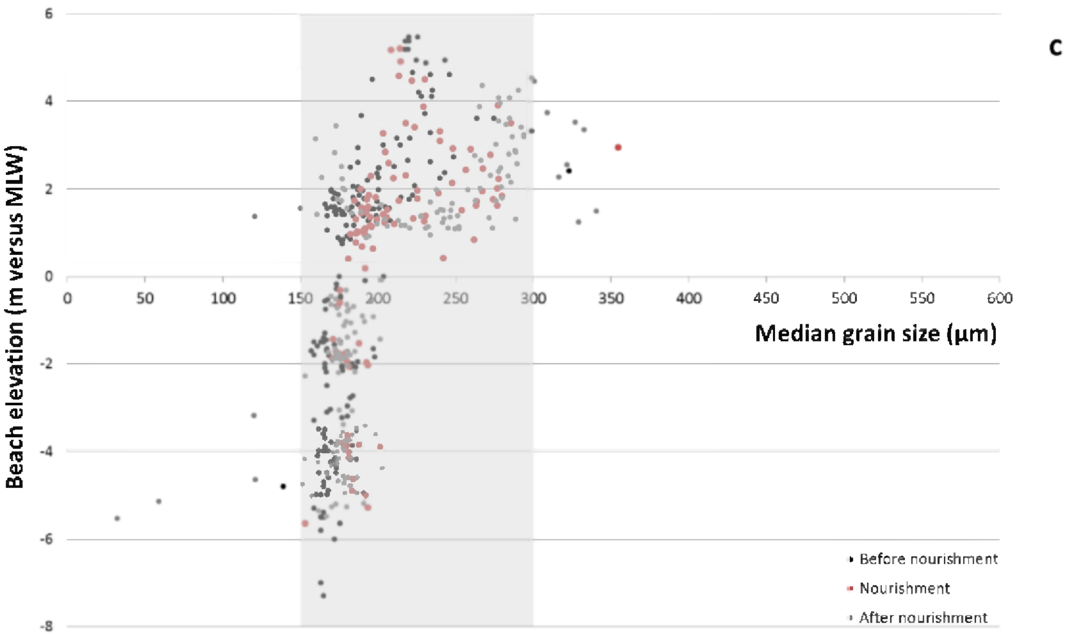
et al. 2010) and after (S and A 2010, 2011, 2012) (Vanden Eede et al. 2011, 2013) the nourishment.

Lombardsijde	2004	2006	2008	2009		2010		2011		2012	
			S	S	A	S	A	S	A	S	A
Intertidal zone	50	25	25	39	26	15	15	15	15	15	15
Shallow subtidal zone	50	25	25	25	24	15	15	15	15	15	15

In total, 52 intertidal and 81 shallow subtidal species were identified. The number of species found differed between before, during and after the nourishment (e):

	Intertidal	Shallow subtidal
Before	39	48
Nourishment	25	69
After	30	60

All mean measured abiotic and biotic variables fell between the maximum and minimum values of the Belgian beaches (d). The median grain size decreased from MHW (354.60 µm) to MLW (120.49 µm) while the shallow subtidal values fluctuated between 150 and 200 µm (c). Ten intertidal samples had a median grain size higher than 300 µm (1 sample before the nourishment (2004), 1 during the nourishment (autumn 2009) and 8 after the nourishment (3 2010 S, 2 2011 A, 1 2012 S and 2 2012 A). Seven very low values (< 150 µm) were detected in the shallow subtidal zone, 1 before (2008), 1 during (2009 A) and 5 after the nourishment (2 2010 S, 1 2010 A and 2 2011 A). The intertidal median grain size was always higher than the shallow subtidal values, except in 2010 A. In autumn, the shallow subtidal abundance was always much higher than the intertidal abundance, while in spring the values were quite similar or higher. However, the intertidal values in 2006 were higher than the shallow subtidal ones and 2011 A values were not higher than 2011 S values (f and g).



In the intertidal zone, *Bathyporeia pilosa* and *Scolelepis squamata* were clearly the most dominant species before the nourishment (h). In 2009 S, 2010 S and A, 2011 S and 2012 S *Bathyporeia pilosa* and *Scolelepis squamata* dominated while *Bathyporeia pilosa* and *B. sarsi* dominated in 2009 A. *Eurydice pulchra* was the most dominant species in 2011 and *Bathyporeia sarsi* and *Scolelepis squamata* dominated in 2012 A. *Spiophanes bombyx* was bearly found in the intertidal zone of Lombardsijde. In the shallow subtidal zone, *Cirratulidae* species (2010 A, 2011 S), *Ensis* juveniles (autumn 2009, 2010 and 2012) and *Macoma balthica* juveniles (autumn 2010 and 2012) showed extreme peak values, e.g. more than 1 000 individuals.m^{-2}. These values are not shown in figure i to allow for a better interpretation of the remaining abundance values.

Before the nourishment and in 2009 S, *Spio* species were the most dominant species. In autumn 2009, 2010 and 2012, *Ensis* juveniles and *Macoma balthica* juveniles dominated. *Cirratulidae* species were dominant in 2010 A and 2011 S while *Spio* species were dominant in 2011 A and 2012 S. *Lanice conchilega* bearly found in the shallow subtidal zone of Lombardsijde.

d - 1

Lombardsijde	INTERTIDAL				
	2004	2006	2008	2009	
Beach elevation (m versus MLW)	1.82 ± 0.09	3.51 ± 0.32	2.49 ± 0.24	2.28 ± 0.22	2.16 ± 0.12
Median grain size (µm)	188.24 ± 4.07	215.92 ± 4.99	215.89 ± 3.63	212.98 ± 3.94	236.84 ± 8.53
Silt fraction (%)	0.03 ± 0.01	0.0007 ± 0.0003	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Total organic matter (%)	0.10 ± 0.01	0.00 ± 0.00	0.24 ± 0.01	0.56 ± 0.02	0.51 ± 0.03
Total organic carbon (%)	0.27 ± 0.03	1.23 ± 0.10	1.50 ± 0.03	1.39 ± 0.06	0.19 ± 0.04
Carbonate content (%)	12.90 ± 0.70	12.26 ± 0.48	12.49± 0.29	11.62 ± 0.53	10.23 ± 1.12
Species richness (number of species)	4.30 ± 0.33	2.52 ± 0.31	3.44 ± 0.30	3.03 ± 0.28	2.96 ± 0.34
Abundance (number of individuals.m^{-2})	50.44 ± 15.71	178.49 ± 52.10	65.89 ± 18.58	46.32 ± 12.57	60.46 ± 19.27
Biomass (g AFDW.m^{-2})	0.55 ± 0.11	6.11 ± 1.62	0.21 ± 0.04	0.33 ± 0.10	0.17 ± 0.06

Lombardsijde	INTERTIDAL							
	2010		2011		2012		minimum	maximum
Beach elevation (m versus MLW)	2.40 ± 0.32	2.08 ± 0.17	2.14 ± 0.28	1.91 ± 0.20	2.58 ± 0.31	1.82 ± 0.06	0.00	5.41
Median grain size (µm)	280.23 ± 8.94	168.12 ± 4.13	262.80 ± 6.84	252.39 ± 9.54	267.89 ± 5.69	247.86 ± 2.87	175.00	464.00
Silt fraction (%)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00	2.25
Total organic matter (%)	0.79 ± 0.04	0.83 ± 0.26	0.56 ± 0.03	0.43 ± 0.04	0.06 ± 0.10	0.88 ± 0.07	0.00	1.84
Total organic carbon (%)	0.12 ± 0.01	0.09 ± 0.05	0.14 ± 0.01	0.09 ± 0.01	0.085 ± 0.005	0.072 ± 0.002	0.00	19.00
Carbonate content (%)	9.87 ± 1.91	5.78 ± 0.28	6.62 ± 0.65	8.52 ± 1.11	6.94 ± 0.54	8.70 ± 0.25	0.00	36.68
Species richness (number of species)	2.47 ± 0.56	5.53 ± 0.50	4.27 ± 0.33	3.27 ± 0.34	3.67 ± 0.51	4.13 ± 0.12	0.00	19.00
Abundance (number of individuals.m^{-2})	36.50 ± 19.56	121.92 ± 33.68	105.59 ± 31.76	46.89 ± 13.54	39.52 ± 11.89	62.26 ± 3.15	0.00	3988.75
Biomass (g AFDW.m^{-2})	0.08 ± 0.02	0.67 ± 0.12	0.46 ± 0.11	0.21 ± 0.07	0.34 ± 0.08	1.10 ± 0.10	0.00	6.95

d - 2

Lombardsijde	SHALLOW SUBTIDAL				
	2004	2006	2008	2009	
Beach elevation (m versus MLW)	-3.84 ± 0.22	-2.07 ± 0.29	-2.78 ± 0.29	-1.59 ± 0.38	-1.48 ± 0.47
Median grain size (µm)	164.64 ± 0.48	171.24 ± 0.75	182.30 ± 2.35	188.42 ± 2.69	195.89 ± 6.62
Silt fraction (%)	0.24 ± 0.05	1.37 ± 0.39	1.01 ± 0.91	0.99 ± 0.49	5.51 ± 4.02
Total organic matter (%)	0.23 ± 0.01	0.00 ± 0.00	0.37 ± 0.05	1.45 ± 0.10	1.44 ± 0.10
Total organic carbon (%)	0.10 ± 0.02	1.56 ± 0.05	1.54 ± 0.03	1.31 ± 0.05	0.21 ± 0.03
Carbonate content (%)	11.91 ± 0.23	11.55 ± 0.17	12.84 ± 0.28	10.88 ± 0.44	11.73 ± 0.51
Species richness (number of species)	4.06 ± 0.30	7.28 ± 0.45	7.56 ± 0.67	5.72 ± 0.44	9.20 ± 1.13
Abundance (number of individuals.m^{-2})	51.20 ± 11.46	41.01 ± 3.25	64.69 ± 10.40	50.71 ± 5.22	275.80 ± 61.06
Biomass (g AFDW.m^{-2})	0.77 ± 0.15	40.79 ± 5.32	2.66 ± 0.47	2.91 ± 0.82	4.40 ± 0.76

Lombardsijde	SHALLOW SUBTIDAL							
	2010		2011		2012		minimum	maximum
Beach elevation (m versus MLW)	-3.05 ± 0.43	-2.78 ± 0.48	-3.14 ± 0.43	-2.57 ± 0.36	-2.98 ± 0.43	-297 ± 0.10	0.00	5.41
Median grain size (µm)	158.35 ± 12.17	190.12 ± 2.20	177.04 ±2.36	168.94 ± 5.46	172.98 ± 1.39	174.73 ± 0.35	175.00	464.00
Silt fraction (%)	10.43 ± 5.27	5.43 ± 1.92	3.65 ± 1.47	5.07 ± 2.82	3.96 ± 1.40	2.26 ± 0.24	0.00	2.25
Total organic matter (%)	2.77 ± 0.57	2.16 ± 0.31	2.22 ± 0.31	1.64 ± 0.29	0.02 ± 0.01	1.93 ± 0.09	0.00	1.84
Total organic carbon (%)	0.51 ± 0.16	0.29 ± 0.08	0.35 ± 0.06	0.23 ± 0.05	0.22 ± 0.03	0.191 ± 0.004	0.00	19.00
Carbonate content (%)	11.62 ± 1.32	12.65 ± 1.05	12.56 ± 0.78	11.07 ± 0.57	11.60 ± 0.57	10.32 ± 0.10	0.00	36.68
Species richness (number of species)	8.53 ± 1.29	13.33 ± 1.26	10.07 ± 0.85	16.60 ± 1.32	12.80 ± 0.91	16.20 ± 0.39	0.00	19.00
Abundance (number of individuals.m^{-2})	54.94 ± 12.34	839.54 ± 201.25	298.85 ± 118.96	218.60 ± 42.55	144.58 ± 24.58	884.70 ± 37.65	0.00	3988.75
Biomass (g AFDW.m^{-2})	4.85 ± 1.15	85.23 ± 16.95	14.88 ± 7.65	4.98 ± 1.64	29.47 ± 9.34	74.23 ± 2.57	0.00	6.95

e - 1

Species list Lombardsijde – intertidal zone					
Before		Nourishment		After	
Abra alba	*Macoma balthica*	*Abra alba*	*Nemertea species*	*Abra alba*	*Haustorius arenarius*
Aphelochaeta marioni	*Magelona species*	*Bathyporeia pelagica*	*Nephtys caeca*	*Bathyporeia pelagica*	*Heteromastus filiformis*
Arenicola marina	*Nemertea species*	*Bathyporeia pilosa*	*Nephtys cirrosa*	*Bathyporeia pilosa*	*Macoma balthica*
Bathyporeia elegans	*Nephtys assimilis*	*Bathyporeia sarsi*	*Nereis longissima*	*Bathyporeia sarsi*	*Macoma balthica juveniles*
Bathyporeia pilosa	*Nephtys caeca*	*Bodotria pulchella*	*Phyllodoce species*	*Cumopsis goodsir*	*Magelona species*
Bathyporeia sarsi	*Nephtys cirrosa*	*Capitella capitata*	*Polydora species*	*Donax vittatus*	*Nemertea species*
Capitella capitata	*Nephtys hombergii*	*Cumopsis goodsir*	*Pseudocuma longicornis*	*Donax vittatus juveniles*	*Nephtys caeca*
Corophium species	*Nephtys species*	*Donax vittatus*	*Scolelepis squamata*	*Ensis juveniles*	*Nephtys cirrosa*
Cumopsis goodsir	*Nereis longissima*	*Ensis juveniles*	*Spiophanes bombyx*	*Eteone flava*	*Owenia fusiformis*
Diastylis rathkei	*Oligochaeta species*	*Eteone longa*	*Spio species*	*Eteone longa*	*Phyllodoce species*
Donax vittatus	*Phyllodoce species*	*Eurydice affinis*	*Spisula subtruncata*	*Eurydice affinis*	*Polydora species*
Eteone longa	*Polydora species*	*Eurydice pulchra*	*Urothoe pulchella*	*Eurydice pulchra*	*Pontocrates altamarinus*
Eumida sanguinea	*Pygospio elegans*	*Haustorius arenarius*		*Harmothoe nodosa*	*Scolelepis squamata*
Eurydice affinis	*Scoloplos armiger*				*Spiophanes bombyx*
Eurydice pulchra	*Scolelepis squamata*				*Spio species*
Gammarus species	*Spiophanes bombyx*				*Urothoe poseidonis*
Glycera species	*Spio species*				*Urothoe pulchella*
Haustorius arenarius	*Talitrus saltator*				
Jassa species	*Urothoe poseidonis*				
Lanice conchilega					

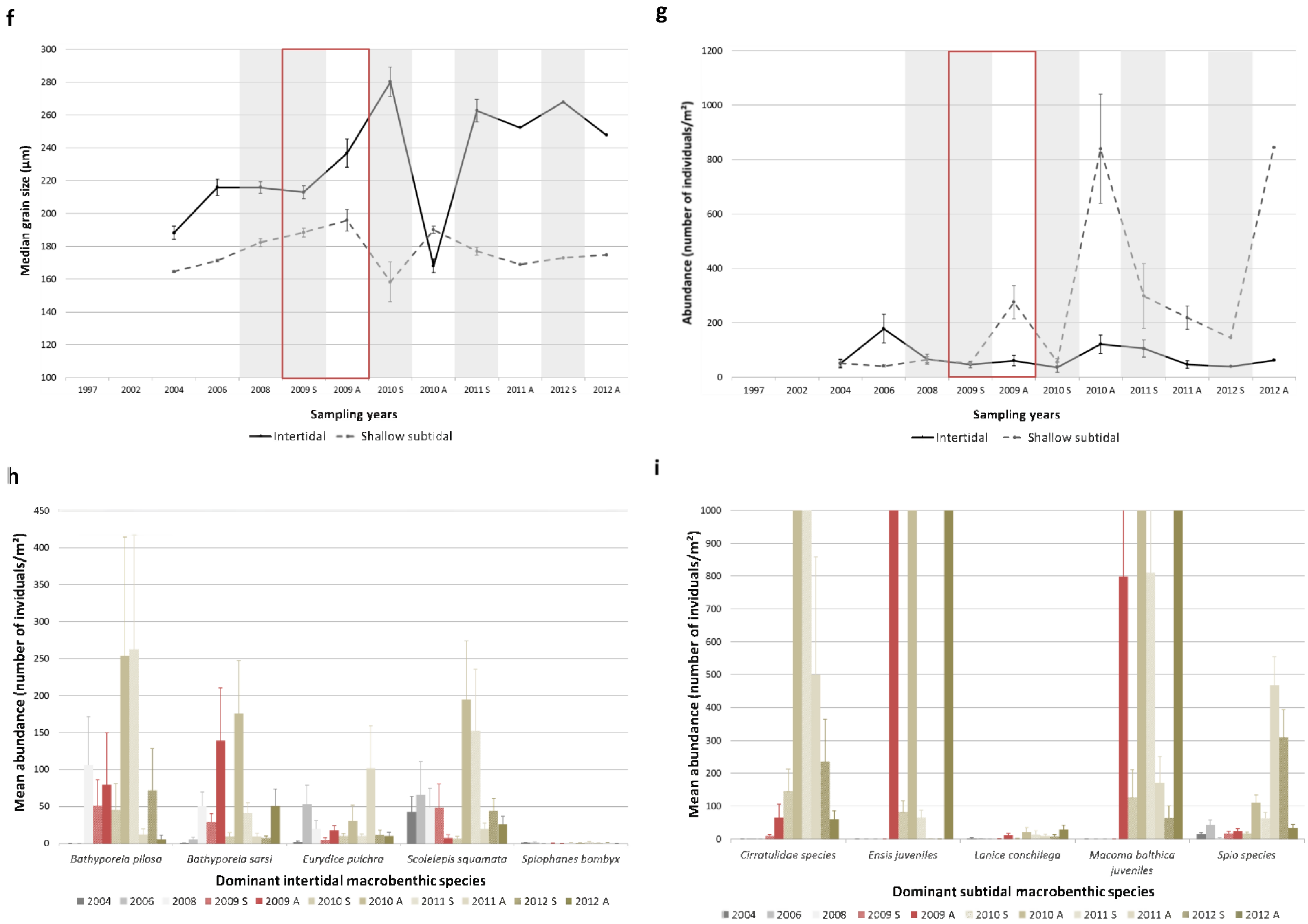
e - 2

Species list Lombardsijde - shallow subtidal zone

Before		Nourishment		After	
Abra alba	*Mysella bidentata*	*Abra alba*	*Magelona species*	*Abra alba*	*Jassa species*
Angulus fabula	*Nemertea species*	*Angulus fabula*	*Nemertea species*	*Ampharetidae species*	*Lanice conchilega*
Atylus falcatus	*Nephtys assimilis*	*Arenicola marina*	*Nephtys assimilis*	*Angulus fabula*	*Macoma balthica*
Autolytus species	*Nephtys caeca*	*Atylus falcatus*	*Nephtys caeca*	*Angulus fabula juveniles*	*Macoma balthica juveniles*
Bathyporeia elegans	*Nephtys cirrosa*	*Bathyporeia pelagica*	*Nephtys cirrosa*	*Angulus pygmaeus*	*Magelona species*
Bathyporeia pelagica	*Nephtys hombergii*	*Bathyporeia sarsi*	*Nephtys hombergii*	*Arenicola marina*	*Mysella bidentata*
Bathyporeia sarsi	*Nephtys species*	*Bodotria pulchella*	*Nereis longissima*	*Atylus falcatus*	*Nemertea species*
Bodotria pulchella	*Nereis longissima*	*Capitella capitata*	*Owenia fusiformis*	*Atylus swammerdami*	*Nephtys assimilis*
Capitella capitata	*Orchomenella nana*	*Cirratulidae species*	*Pariambus typicus*	*Autolytus species*	*Nephtys caeca*
Cirratulidae species	*Owenia fusiformis*	*Corophium species*	*Pectinaria koreni*	*Bathyporeia elegans*	*Nephtys cirrosa*
Cumopsis goodsir	*Pariambus typicus*	*Cumopsis goodsir*	*Pholoe minuta*	*Bathyporeia pelagica*	*Nephtys hombergii*
Diastylis bradyi	*Perioculodes longimanus*	*Diastylis rathkei*	*Phyllodoce species*	*Bathyporeia pilosa*	*Nephtys juveniles*
Donax vittatus	*Phyllodoce species*	*Donax vittatus*	*Polydora species*	*Bathyporeia sarsi*	*Nereis longissima*
Ensis directus	*Polydora species*	*Ensis juveniles*	*Pontocrates arenarius*	*Capitella capitata*	*Oligochaeta species*
Eteone flava	*Pontocrates altamarinus*	*Eurydice affinis*	*Pseudocuma longicornis*	*Cirratulidae species*	*Orchomenella nana*
Eteone longa	*Pontocrates arenarius*	*Eurydice pulchra*	*Scoloplos armiger*	*Cumopsis goodsir*	*Owenia fusiformis*
Eurydice pulchra	*Praunus neglectus*	*Glycera species*	*Scolelepis squamata*	*Diastylis bradyi*	*Pariambus typicus*
Gammarus species	*Pygospio elegans*	*Harmothoe nodosa*	*Sigalion mathildae*	*Diastylis rathkei*	*Pectinaria koreni*
Glycera species	*Scoloplos armiger*	*Haustorius arenarius*	*Spiophanes bombyx*	*Donax vittatus*	*Pholoe minuta*
Harmothoe species	*Scolelepis squamata*	*Heteromastus filiformis*	*Spio species*	*Donax vittatus juveniles*	*Phyllodoce species*

e - 3

Species list Lombardsijde - shallow subtidal zone					
Before		Nourishment		After	
Haustorius arenarius	*Sigalion mathildae*	*Lanice conchilega*	*Spisula subtruncata*	*Ensis juveniles*	*Polydora species*
Lanice conchilega	*Spiophanes bombyx*	*Macoma balthica*	*Urothoe poseidonis*	*Ensis species*	*Pontocrates altamarinus*
Macoma balthica	*Spio species*	*Macoma balthica juveniles*	*Urothoe pulchella*	*Eteone flava*	*Pygospio elegans*
Magelona species	*Urothoe poseidonis*			*Eteone longa*	*Scoloplos armiger*
				Eumida sanguinea	*Sigalion mathildae*
				Eurydice pulchra	*Spiophanes bombyx*
				Gammarus species	*Spio species*
				Glycera species	*Spisula species*
				Harmothoe species	*Urothoe poseidonis*
				Heteromastus filiformis	*Urothoe pulchella*


f
Median grain size (µm)
300
280
260
240
220
200
180
160
140
120
100
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
2012 S
2012 A
Sampling years
Intertidal
Shallow subtidal
g
Abundance (number of individuals/m²)
1200
1000
800
600
400
200
0
1997
2002
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
2012 S
2012 A
Sampling years
Intertidal
Shallow subtidal
h
Mean abundance (number of inviduals/m²)
450
400
350
300
250
200
150
100
50
0
Bathyporeia pilosa
Bathyporeia sarsi
Eurydice pulchra
Scolelepis squamata
Spiophanes bombyx
Dominant intertidal macrobenthic species
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
2012 S
2012 A
i
Mean abundance (number of inviduals/m²)
1000
900
800
700
600
500
400
300
200
100
0
Cirratulidae species
Ensis juveniles
Lanice conchilega
Macoma balthica juveniles
Spio species
Dominant subtidal macrobenthic species
2004
2006
2008
2009 S
2009 A
2010 S
2010 A
2011 S
2011 A
2012 S
2012 A