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# Tools for the Improvement of the Efficiency and Sustainability of Shore Nourishments – Results of the research project STENCIL

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# Tools for the Improvement of the Efficiency and Sustainability of Shore Nourishments – Results of the research project STENCIL

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## Summary

Shore nourishments have been carried out worldwide for several decades and are nowadays seen as an almost routine coastal protection measure. However, the recent paradigm shift to an Integrated Coastal Zone Management (ICZM) and to an Ecosystem Approach to Management (EAM) requires new concepts, models and tools for the implementation of more sustainable and environmentally friendly shore nourishments. The interdisciplinary research project STENCIL aimed at making a first step towards the long-term goal of establishing an EAM for shore nourishments. Joining the expertise of coastal engineers, geologists and ecologists, the project has provided improved tools and methods for the prediction of coastal hydro- and morphodynamics and related ecological impacts. A combination of field measurements, laboratory experiments and analytical as well as numerical models has resulted in valuable new data sets. This paper summarizes the most important scientific outcomes of the project, like the use of Artificial Neural Networks (ANN) to overcome weaknesses of current hydrodynamic numerical modelling tools; the analysis and interpretation of long-term beach profile data using a newly developed data-driven methodology, allowing for a better assessment of the morphological development of beach and foreshore nourishments; the improvement of a sediment transport model for mixed sand based on a unique large-scale experiment in the Large Wave Flume (GWK) carried out within the project; an impact assessment of dredging activities around Sylt, based on six extensive survey cruises at the sand extraction sites "Westerland II & III"; and the application of the hydrotoxicology method for the same study site to determine the influence of sand extraction on chemical-physical water quality parameters and the ecotoxicological potential. These interdisciplinary project results were combined with a comprehensive state of the art review in a SWOT analysis for shore nourishments, which shall provide a tool for decision-makers and a basis for upcoming research projects addressing the identified knowledge gaps.

## Keywords

coastal protection, shore nourishment, coastal zone management, sustainability, environmental impacts, ecosystem approach

## Zusammenfassung

*Seit mehreren Jahrzehnten werden weltweit Sandaufspülungen durchgeführt, die heute entsprechend zu den gängigen Küstenschutzmaßnahmen zählen. Der aktuelle Paradigmenwechsel, hin zu einem Integrierten Küstenzonenmanagement (IKZM) und einem „Ecosystem Approach to Management" (EAM), erfordert jedoch neue Konzepte, Modelle und Werkzeuge, um Sandaufspülungen nachhaltiger und umweltfreundlicher umzusetzen als bisher. Mit dem interdisziplinären Forschungsprojekt STENCIL sollte ein erster Schritt in Richtung des langfristigen Ziels der Etablierung eines EAM für Sandaufspülungen geleistet werden. Durch die gemeinsame Expertise von Küsteningenieuren, Geologen und Ökologen lieferte das Projekt Werkzeuge und Methoden zur verbesserten Vorhersage der Hydro- und Morphodynamik an der Küste sowie zur Abschätzung der ökologischen Auswirkungen. Weiterhin entstanden wertvolle Datensätze aus Feldmessungen und Laborexperimenten sowie aus der Anwendung analytischer und numerischer Modelle. Dieser Artikel fasst die wesentlichen wissenschaftlichen Ergebnisse des Projekts zusammen: die Anwendung Künstlicher Neuronaler Netzwerke (KNN), um Schwächen aktueller hydrodynamisch-numerischer Modelle auszugleichen; die Analyse und Interpretation langfristiger Strandprofildaten mit einer neuen datengetriebenen Methode, die eine bessere Bewertung der morphologischen Entwicklung von Strand- und Vorstrandaufspülungen erlaubt; die Verbesserung eines Sedimenttransportmodells für gemischte Sande, basierend auf einem einzigartigen großmaßstäblichen Modellversuch, der im Rahmen des Projekts im Großen Wellenkanal (GWK) durchgeführt wurde; eine Abschätzung der Auswirkungen von Sandentnahmen vor Sylt, auf Grundlage von sechs umfangreichen Messfahrten zu den Sandentnahmegebieten „Westerland II & III"; sowie die Anwendung hydrotoxikologischer Methoden zur Bestimmung des Einflusses von Sandentnahmen auf die chemisch-physikalischen Wasserqualitätsparameter und auf das ökotoxikologische Potential. Diese interdisziplinären Projektergebnisse wurden mit einer umfangreichen Studie zum aktuellen Stand von Wissenschaft und Technik kombiniert und in einer SWOT-Analyse für Sandaufspülungen zusammengefasst, welche eine Entscheidungsunterstützung für die Praxis und eine Grundlage für zukünftige Forschungsprojekte darstellt.*

## Schlagwörter

*Küstenschutz, Sandaufspülung, Küstenzonenmanagement, Nachhaltigkeit, Umweltverträglichkeit, Ökosystemansatz*

## 1 Introduction

Shore nourishments, i.e. refilling of the natural beach profile with imported or recycled sediment, are an important part of coastal protection and management strategies worldwide. Over the past few centuries, sound practical experience has been gathered by many countries, based on monitoring of morphological shoreline changes and scientific studies on beach erosion.

Although coastal engineers consider nourishments to be "soft" or "low-regret" coastal protection measures – in contrast to "hard", fixed measures like breakwaters or seawalls – the sediment extraction and nourishment activities may have significant large-scale and long-term impacts on the coastal environment: Sediment extraction and dumping activities disturb or destroy marine habitats (e.g. Greene 2002, De Jong et al. 2015, Rosov et al. 2016). Regular activities within the same coastal areas inhibit the regeneration of benthic communities, which can take up to several years depending on the species and local conditions (Speybroeck 2007, Leewis et al. 2012, Rosov et al. 2016, Wooldridge et al. 2016). In case the sediment characteristics are significantly altered, a shift in benthic species occurs (Speybroeck et al. 2006, Vanden Eede et al. 2014, Wooldridge et al. 2016). Large management schemes with regular re-nourishments change the sediment composition and morphology along whole coastal stretches (Armstrong and Lazarus 2019). Despite several known deteriorating impacts on the marine environment, nourishments are often preferred over concrete structures, as they are perceived to be more "natural". However, many ecological impacts, especially those that only surface in the longer term, are not fully understood; their consequences could e.g. significantly affect the food web and subsequently local and regional fisheries (Essink et al. 1997, Vanden Eede et al. 2014).

In order to reach the UN's Sustainable Development Goals (e.g. Goal 14: Oceans) or the goals of the EU's Marine Strategy Framework Directive (good environmental status), coastal protection strategies must adhere to the ecosystem approach, i.e. offer efficient and sustainable protection of people and infrastructure, while at the same time maintaining or improving the environmental status of the coastline. To successfully implement sustainable coastal management strategies, it is thus crucial 1) to improve the understanding of the morphological and ecological impacts, and 2) to optimize the extraction and nourishment activities in order to maximize their efficiency while minimizing their ecological impact. To approach this demand, the research project STENCIL (Strategies and Tools for Environment-friendly Shore Nourishments as Climate Change Impact Low-Regret Measures), which was funded by the German Ministry for Education and Research from 2016 to 2019 (contract no. 03F0761 A-D), aimed to develop new tools for the design of sustainable nourishments and to provide the next step towards an ecosystem approach for shore nourishments.

This paper describes the most important outcomes of the project, which covered six work areas, combining the expertise of coastal engineers, geologists and ecologists. Five work areas (chapters 2–6) focused on the development of new tools and methods (e.g. numerical models, laboratory methods or protocols) that support the planning of physically and ecologically sustainable sediment extraction and nourishment activities. In addition, the research results are used to improve the understanding of hydrodynamic, morphodynamic and biological processes in the coastal zone. While chapter 2 investigates hydrodynamic processes in the coastal zone, chapters 3 and 4 address coastal morphodynamics on meso- and microscales, respectively (with time scales ranging from seconds to years). Chapter 5 in turn investigates regional refill processes at sediment extraction sites. Chapter 6 combines knowledge about sediment dynamics with ecotoxicology to investigate potential impacts of fine sediment on marine organisms. In the overarching sixth work area (chapter 7), all project partners collaborated in the development of a comprehensive, interdisciplinary tool to help decision-makers and coastal managers to choose a suitable nourishment type (i.e. location, repetition rate etc.) for their application.

## 2 Assessment and modeling of storm-tides and nearshore waves/currents

The first part of STENCIL aims to identify methods for the modelling of processes which are relevant for the development of physically sustainable sand nourishments for coastal protection. Detailed knowledge about coastal hydrodynamics is essential for assessment and classification of coastal morphologic processes. Especially the processes driving sediment transport – such as sea state, tides, winds and currents – should be understood to apply well-directed coastal protection measures. A main question regarding the design of sand nourishment concerns the ideal placement within the coastal profile and the effect of the nourishment body on the coastal hydrodynamics. As part of STENCIL, different sand nourishment strategies were investigated regarding their impacts on hydrodynamic processes in coastal areas, storm surge-induced current properties, and the potential lifetime of sand nourishments.

The different nourishment measures are classified – regarding the placement location – as supralittoral nourishments, beach nourishments and shoreface nourishments. The deduction of an appropriate modelling system for analyzing the processes mentioned above is an integral part of the project's objective. The main intention is to identify an optimal sand nourishment strategy, which extends the nourishment lifetime and protects the shore from storm surge-induced erosion.

### 2.1 Methodology: Pilot Study Data Acquisition, Analysis and Modelling

In the framework of STENCIL, a comprehensive database of hydrologic data sets (water levels, sea states, currents, meteorological data, and bathymetry) within the pilot study domain west of Sylt was assembled. Data from the following measurement stations was used for the analysis: wave buoy 'Sylt' (~38 km west of Westerland, operated by HZG), wave buoy 'Westerland' (~5 km west of Westerland, operated by LKN.SH), measuring pile 'Westerland' (<2 km west of Westerland, operated by LKN.SH), and the wave buoy 'Bunkerhill' (in the southern area of Sylt, <1 km away from the coast, operated by HZG). The data was provided by the German institutions LKN.SH (Landesbetrieb für Küstenschutz, Nationalpark und Meeresschutz Schleswig-Holstein), HZG (Helmholtz-Zentrum Geesthacht) and WSV (Wasserstraßen- und Schifffahrtsverwaltung).

Extreme events were analyzed statistically to determine magnitudes which are relevant for the design of sand nourishments. The long-term data of the buoy 'Westerland' is used as input data and for boundary conditions in the simulation models. *Storm surge conditions* and *mean conditions* are differentiated.

In a first step, numerical model results were validated and compared with experimental results (chapter 2.2). Subsequently, we set up a modelling system which combines process-based hydrodynamic modelling and data-based modelling (chapter 2.3). Additionally, physical experiments with a fixed bed were conducted in order to measure resulting currents within the surf zone – induced by different sand nourishment strategies (chapter 2.4).

## 2.2 Numerical Model: Assessment and Validation

The performance of two common open source numerical models *XBeach* (https://oss.deltares.nl/web/xbeach/) and *Delft3D* (https://oss.deltares.nl/web/delft3d) was evaluated focusing on the representation of 3D surf zone processes. The model results were validated and compared with available data sets of large-scale flume experiments by van der A et al. (2017) in order to determine the models' advantages and applicability. The hydrodynamic parameters and the bathymetry of the experiments are similar to those of the pilot study of STENCIL. The research focused on the detailed investigation of hydrodynamics – vertical flow profile, including the ground-level boundary layer and turbulence – in the case of plunging breakers (fixed bed, regular waves). The water level, mean flow velocity and turbulent kinetic energy were measured. Details about the experimental setup can be found in van der A et al. 2017.

Figure 1 shows the graphs of measured and numerically calculated (best adjustments) wave heights $\overline{\mathrm{H}}$. Offshore, where the water depth is constant, both *XBeach* and *Delft3D* calculate the given wave heights. However, discrepancies between measurement and calculations were observed. Both models give an inadequate result for the energy dissipation due to wave breaking. While *XBeach* (non-hydrostatic mode) is able to reproduce the wave shape, it does not manage to model the wave breaking process in detail at the same time. The lower energy dissipation is affecting the other hydrodynamic parameters, such as the undertow. The 'roller model' in *Delft3D* achieves a substantially better magnitude of energy dissipation but cannot reproduce the wave shape properly. These results underline the models' limitations in the coastal area. The shortcomings of existing models have led to the development of a combined data- and process-based model for reproducing the coastal processes in the pilot study domain.

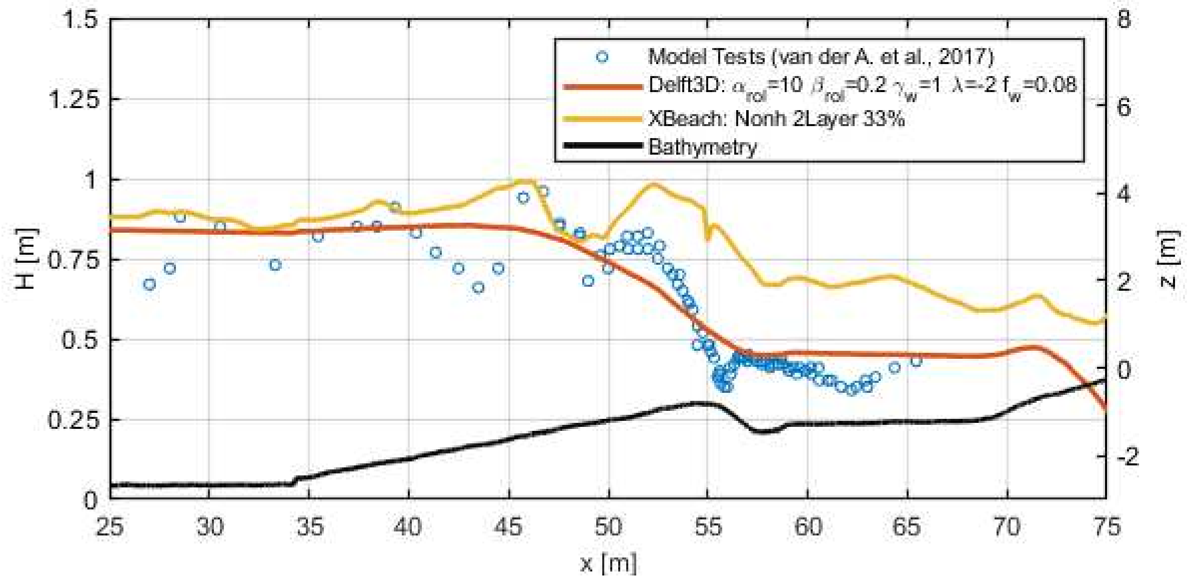


Figure 1: Curves of measured (blue, data source: van der A et al. 2017) and calculated wave heights (orange: Delft3D, yellow: XBeach) alongside wave flume axis.

## 2.3 Combined Data- and Process-based Model for the Pilot Study

The measured sea state parameters of the buoy 'Bunkerhill' differ from those of buoys 'Sylt' and 'Westerland' due to the bathymetry's impact. An artificial neural network (ANN) was configured to predict the sea state parameters at the location of buoy 'Bunkerhill'. The sea state parameters of buoy 'Sylt' ($H_{m0}$ and $\theta$) and measuring pile 'List' ($u_{10}$ and $\theta_{wind}$) are used as input data. Thus, the ANN's capability of transforming sea state parameters into nearshore conditions can be evaluated. The ANN does not model the bathymetry's impact explicitly, but implicitly by considering the weighted functional connectivity of input and output data (Browne et al. 2007). Since the calculation is based on time series, a nonlinear autoregressive model with exogenous input (NARX) is used (Beale et al. 2018). Recurrent networks with backpropagation are appropriate, especially for this kind of prediction (Mandal and Prabaharan 2006). The NARX model includes not only exogenous input values (buoy 'Sylt' and measuring pile 'List') but also past values of the intended output (buoy 'Bunkerhill'), and thus considers the sea state's temporal evolution.

For model evaluation, both the hydrodynamic model and the neural network calculated the resulting sea state conditions at buoy 'Bunkerhill' for selected individual storm surge events. Both models received data of buoy 'Sylt' as input (wave height $H_{m0}$ and wave angle $\theta$) and additional information from the *CoastDat* database about wave period $T_p$ and wave angle variance. The selection criteria for the storm surge events was a significant wave height that satisfies an exceeding probability of P = 1 % at least.

A regression analysis showed the large statistical variation of estimated magnitudes vs. physically measured values. Both the ANN and the *Delft3D* calculation overestimate lower wave heights ($H_{m0}$ < 2 m), while both models generally underestimate higher waves. In general, the ANN achieves a better approximation.

The discrepancies of the *Delft3D* model are induced i.a. by the input data: the model-based *CoastDat* database contains good sea state estimations, but calculated water levels during storm surges are not sufficiently accurate. As soon as the *EasyGSH* database is entirely available, this is expected to improve the boundary condition control. In near-shore areas, a better sea state prediction is obtained by the data-based model. The buoy 'Bunkerhill' is already located in the transition zone, so that nonlinear processes are relevant for the sea state development here.

## 2.4 Physical Experiments

Systematic studies about the vertical current profile within the surf zone, which can be typically found along Sylt's west coast, are either unavailable or not detailed enough for investigating the effectiveness of different sand nourishment strategies. For this reason, scaled experiments were conducted in the wave flume of LWI (Leichtweiß-Institute for Hydraulic Engineering and Water Resources) in Braunschweig. The experiments were designed according to the pilot study. The objective was to investigate the effectiveness of various nourishment strategies. The experiments' scaling is based on the Froude similarity (scaling factor $^1/_{20}$). The following beach profiles and nourishment strategies were investigated (cf. Figure 2):

1. The **Baseline Profile (Setup 0, S0)** is derived from long-term measurements during winter (October to March) at the cross-profile km 0+600 around Westerland.
2. The **Outer Shoreface Nourishment (Setup 1, S1)** is based on actual nourishments from 2017 at the location mentioned above.
3. The **Inner Shoreface Nourishment (Setup 2, S2)** enlarges the reef towards the coast.
4. The **Beach Nourishment (Setup 3, S3)** increases the sand volume within the zone of fluctuating water levels.

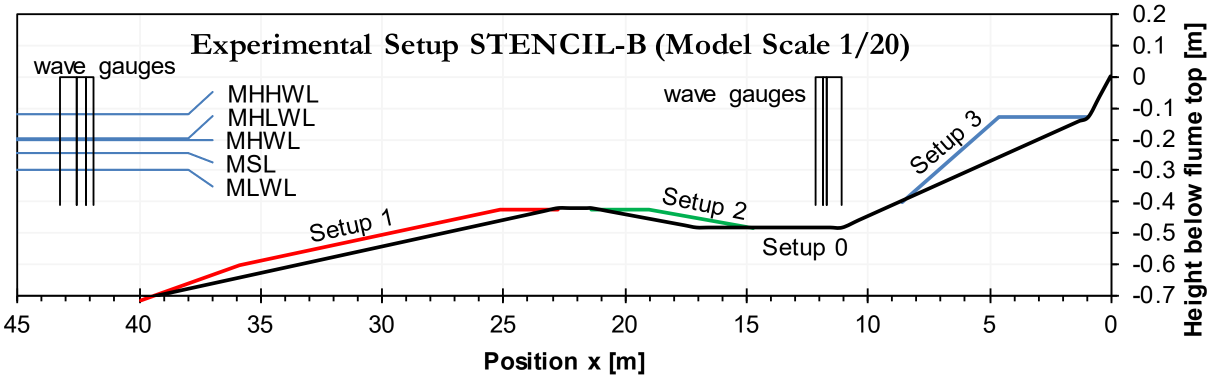


Figure 2: Model Setup: Cross Profiles of tested Nourishment Strategies and Wave Gauges.

The test parameters are derived from the model results for buoy and gauge 'Westerland'. Since nourishments in the pilot study domain are repeated once a year, the water levels and wave parameters are based on annual indexes (see Table 1). Storm surge events induce higher seaward currents; subsequently, the tests focused on extreme values with an exceedance of 1 % or an annuality of 1 a. In each test run, a JONSWAP spectrum with a minimum of 1000 waves was generated.

Table 1: Physical Experiments of STENCIL-B: Test Series (prototype scale).

| No. | water level scenario* | water level (m NHN) | scenario | wave height $H_{m0}$ (m) | peak period $T_p$ (s) |
|---|---|---|---|---|---|
| **1.1** | MHWL | 0.84 | mean conditions | 0.87 | 6.1 |
| **1.2** | MSL | 0.00 | mean conditions | 0.87 | 6.1 |
| **1.3** | MLWL | −0.97 | mean conditions | 0.87 | 6.1 |
| **2.1** | MHHWL | 2.58 | P = 1 % | 3.50 | 11.0 |
| **2.2** | MHLWL | 0.97 | P = 1 % | 3.50 | 11.0 |
| **3.1** | MHHWL | 2.58 | 1 a max. | 4.71 | 13.0 |

* MHWL = mean high water level, MSL = mean sea level, MLWL = mean low water level, MHHWL = mean highest high water level, MHLWL = mean highest low water level

At three locations, velocity sensors measured the horizontal flow velocity alongside the flume axis: on the **reef top**, in the **reef trough** (x ~ 13 m) and on the **reef slope** in between (x ~ 18 m). Five sensors were placed at each location, allocated equidistantly in the vertical to record the vertical current profile. All measuring devices sampled at a rate of 60 Hz. During post-processing of velocity data, a low pass filter was used to remove frequencies above 50 Hz, thus reducing noise and improving data reliability. Furthermore, data with correlation < 70 % (Sulaiman et al. 2013) and signal-to-noise ratio SNR < 10 (Strom and Papanicolaou 2007) were eliminated. The phase-space method by Goring and Nikora (2002) was applied to find and eliminate spikes.

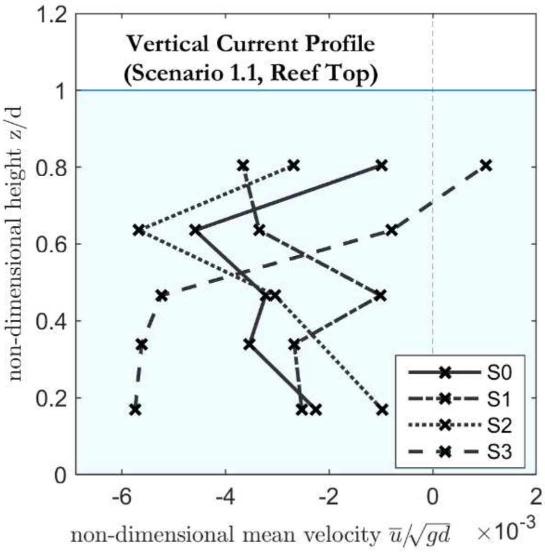


Figure 3: Normalised velocity profiles on the reef top of all tested nourishment strategies and scenario 1.1.

For each sensor in the vertical, the **time-averaged velocity ū** (perpendicular to shoreline) was calculated for delineation of the vertical current profiles. As an example, Figure 3 shows the resulting velocity profiles above the reef for each nourishment strategy under mean conditions (scenario 1.1).

Both, the vertical sensor positions (vertical axis) and the mean velocities (horizontal axis) are referred to the local water depth and thus dimensionless. The measured averaged values of each sensor are connected in the plot for better visualization.

The measurements allow an undertow comparison of all nourishment strategies. In the example shown above – mean high tide (MHWL) and moderate wave conditions – the beach nourishment (S3) causes the highest undertow velocity above the reef. Hence, its transport capacity is higher than the reference (baseline profile, S0) under mean conditions which can cause more disadvantageous conditions regarding transport capacity and morphodynamic evolution. The other strategies' velocities have similar magnitudes at the lowest measuring position, but their curves diverge upwards.

The post-processed results enable us to assess the nourishment strategies in respect to undertow velocity and transport capacity, referring to the baseline profile (S0) as reference. The mean velocities of the lowest sensor serve as an assessment criterion. If the nourishment undertow is similar to the reference undertow, it is classified as neutral (○); accordingly, the nourishments are classified into undertow velocities that are higher (–) or lower (+, ++) than in the reference case. For the reef top and the reef slope, the classifications of all nourishment measures (S1 – S3) are listed in Table 2. The undertow velocities in the reef trough (not shown in the table) differ only slightly under moderate conditions (scenarios 1.1 – 1.3). In the case of storm surges and high water levels (scenarios 2.1 and 3.1), the beach nourishment (S3) causes higher undertow velocity in the trough.

Table 2: Nourishment assessment based on baseline profile (S0) velocities.

| | | Reef Top | | | Reef Slope | | |
|---|---|---|---|---|---|---|---|
| **conditions** | | **S1** | **S2** | **S3** | **S1** | **S2** | **S3** |
| **mean** | **1.1 MHWL** | ○ | ○ | – | ○ | – | ○ |
| **mean** | **1.2 MSL** | ○ | ○ | ○ | ○ | – | ○ |
| **mean** | **1.3 MLWL** | – | ○ | ○ | ○ | – | ○ |
| **storm surge** | **2.1 MHHWL** | ○ | – | ++ | ++ | ○ | ○ |
| **storm surge** | **3.1 MHHWL** | ○ | – | + | ++ | ++ | ○ |
| **extreme** | **2.2 MHLWL** | + | ○ | + | ++ | ++ | ○ |

deviation less than 7.5 % (○), velocity < 92.5 % of baseline velocity (+), < 80 % (++), > 107.5 % (–)

In general, at mean low tide (MLWL) the outer shoreface nourishment (S1) leads to the highest undertow velocity on the reef top. In the case of mean wave conditions and mean high tide (MHWL), the beach nourishment (S3) results in a significantly increased mean near-bed velocity above the reef referring to the baseline profile (S0). For extreme events and high water levels (scenarios 2.1 and 3.1), the beach nourishment (S3) results in lower mean near-bed velocities than reference, while the inner shoreface nourishment (S2) exceeds the reference velocities above the bar. In general, the outer shoreface nourishment (S1) seems to lead to the largest reduction in undertow velocities and transport capacity, therefore appearing to be the most efficient measure with equal sand volume.

## 2.5 Conclusions

The investigations showed that common 2D-models like *Delft3D* and *XBeach* cannot appropriately model the undertow current. The insufficient modelling of hydrodynamic processes has a direct effect on the storm surge-induced erosion and morphodynamics. Data-based modelling with ANN resulted in significantly better sea state prediction in the pilot study domain than hydrodynamic modelling on its own.

# 3 Drivers and effects of morphological developments of beach and foreshore nourishments on meso-scale domain – data mining and modeling

Although regular beach profiles are collected along many nourished coastal stretches to derive the efficiency of the nourishment activities, the morphological behavior of different nourishment types (i.e. beach vs. shoreface nourishment) is still not fully understood. To improve this understanding and to provide the next step towards the optimization of nourishment design, the morphological behavior and the effects of beach and shoreface nourishments on sandy coastal profiles were studied with a newly developed data-driven methodology, following Zorndt et al. (2010) and Zorndt et al. (2011).

## 3.1 Data mining and setup of a data basis

The elevation (z) of sandy coastal profiles of several islands in the Wadden Sea has been measured for several decades. The work within this part of the project was focused on data sets from two Wadden Sea islands: 1) Ameland in the Netherlands and 2) Sylt in Germany. Since 1965, Rijkswaterstaat (RWS) has collected regular coastal profiles of Ameland by combining aerial photogrammetry (emerged part of the profile) with echosounding (submerged part of the profile). Starting in 1972, the coastal profiles on Sylt have been collected regularly by the Landesbetrieb für Küstenschutz, Nationalpark und Meeresschutz Schleswig-Holstein (LKN.SH), using a combination of RTK-GPS (emerged part of the profile) and echosounding (submerged part of the profile).

To develop a data-driven methodology, important characteristics of the data set were the cross-shore extent of the measurements ($\Delta$x), the spatial interval in cross-shore ($\partial$x) and longshore ($\partial$y) direction, and the measurement frequency ($f_s$). Figure 4 presents these characteristics of the two data sets by means of two example transects.

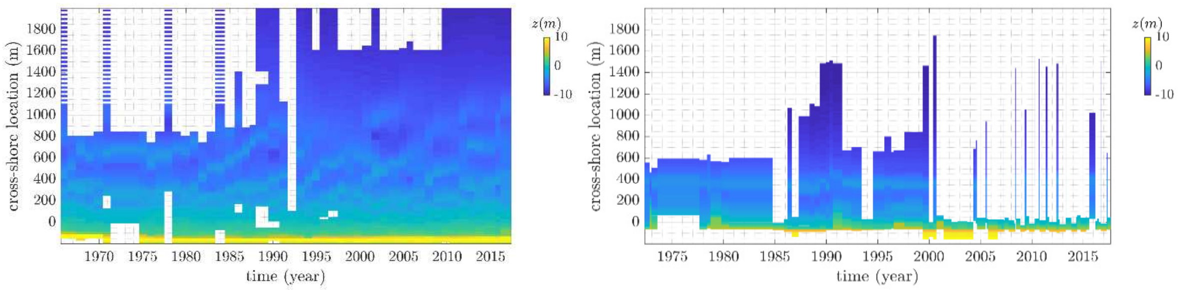


Figure 4: Typical data set characteristics of a coastal profile along Ameland (left) and Sylt (right) (Gijsman et al. 2018).

The coastal profiles of Ameland were collected with $f_s$ = 1 yr$^{-1}$. In time, $\partial$x decreased from 20 m to 5 m and $\Delta$x increased from 800 m to 3000 m from the shoreline (cross-shore coordinate x = 0 m). The transects in Ameland were separated by $\partial$y = 200 m (Gijsman et al. 2019c). In Sylt the monitoring frequency was higher, between 1 and 4 measurements were performed per year. $\partial$x typically ranged between 20 m and 1 m while the maximum $\Delta$x was 1700 m, often depending on the nourishment that was measured (Gijsman et al. 2018). Transects on Sylt were separated by $\partial$y = 50 m (LKN.SH 2015). In general, the Ameland data set was measured to evaluate the long-term morphological development of the shoreline, while the Sylt data set was collected to assess the direct necessity of a beach and/or shoreface nourishment as well as their behavior.

Table 3: The nourishment strategies at the two field study sites between 1965 and 2017 (adapted from Gijsman et al. 2018), showing nourishment lifetime $L_n$, nourishment density $c_n$ and the horizontal alongshore coordinate $y_n$ of the nourishments.

| | Beach nourishments | | | | Shoreface nourishments | | | |
|---|---|---|---|---|---|---|---|---|
| | No. | $L_n$ (a) | $c_n$ (m$^3$/m) | $y_n$ (km) | No. | $L_n$ (a) | $c_n$ (m$^3$/m) | $y_n$ (km) |
| Ameland | 5 | 4-5 | 180-240 | 4-8 | 7 | 3-5 | 250-560 | 2-8 |
| Sylt | 147 | 1 | 50-675 | 0.2-5 | 10 | 1-2 | 150-450 | 0.5-2 |

Beach nourishments were placed on the emerged part of the profile between z = 2 m and z = 5 m above mean sea level (z = 0 m). Shoreface nourishments were placed on the submerged part of the profile between z = −3 m and z = −6 m. The general ranges of the design characteristics of these nourishments are summarized in Table 3.

Different design strategies were used in the two nourishment areas (Wilmink et al. 2017). While a total of 5 beach nourishments and 7 shoreface nourishments were constructed on Ameland until 2017, 147 beach nourishments and 10 shoreface nourishments were constructed on Sylt. These different approaches allowed for studying the effects of different nourishment types.

## 3.2 Natural morphological processes

Before reporting on the morphological behavior and effects of the nourishments, the natural long-term morphological behavior of the coastal profiles at the two study sites is presented. Two characteristic profiles in Ameland (transect 1600) and Sylt (transect 0+205) indicated significant differences in morphological evolution and long-term equilibrium. Figure 5 presents the individual measurements between the beginning of the data set and 2017 (grey lines), as well as the average profile (thick black line) and standard deviation (thin black line).

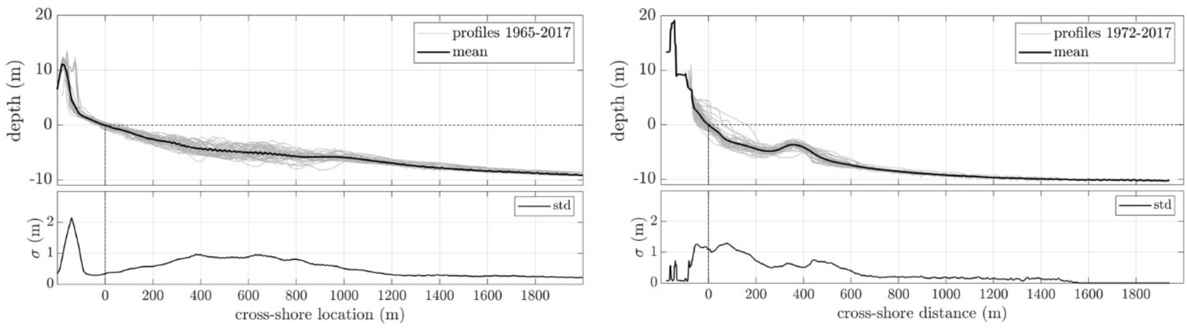


Figure 5: Long-term morphological behavior at two transects on Ameland (left) and Sylt (right) (Gijsman et al. 2018).

Between elevations of z = 0 m and z = 2 m, the average Ameland profile was more gently sloped (±1:50) than the Sylt profile (±1:17.5) (Gijsman et al. 2018). Furthermore, the Ameland profile is characterized by 2-3 consecutive sandbars that migrate in offshore direction on the timescale of years (Ruessink et al. 2003). This can also be observed in Figure 4, as well as by the increasing standard deviation in the submerged part of the profile. In Sylt, the present sandbar was relatively stable and the deviations in the emerged and submerged parts of the profile were the direct result of beach and shoreface nourishments and the preceding beach erosion. The Wright and Short (1984) beach state model classifies the beaches as intermediate-dissipative (Ameland) and intermediate-reflective (Sylt). The difference in natural morphology affects both the analysis of nourishment interventions as well as the behavior of the nourishments.

## 3.3 Data-driven methodology to assess nourishment lifetimes

In the recent design practice of nourishments, usually a certain nourishment lifetime ($L_n$) was projected (Hanson et al. 2002). The nourishment lifetime indicates the period of time in which the nourishment is expected to have a positive contribution to the beach profile. For beach nourishments, this means that the nourishment leads to a larger volume of sand on the emerged part of the beach, either compared to the pre-nourishment volume or to a defined reference volume. For shoreface nourishments, this means a larger volume of sand in the submerged part of the profile. However, since shoreface nourishments are placed in the morphologically active part of beach profile, they continuously interact with ongoing natural morphological processes such as the migration of nearshore sandbars (Gijsman et al. 2019c). Therefore, the period of time that the nourishment interferes with the natural behavior of the nearshore sandbar has also been used as a definition of the shoreface nourishment lifetime (Ojeda et al. 2008).

Present design guidelines (Verhagen 1992, Dette et al. 1994) indicated that the nourishment lifetime mainly depends on the nourished volume ($v_n$) in combination with the local erosion rates (e). According to Verhagen (1992) the local erosion rates can be assumed constant and independent of the nourished volume. His approach, however, applies a 40 % reduction of the nourishment lifetime to include possible increased erosion rates and other uncertainties. The approach by Dette et al. (1994) describes that local erosions rates may initially increase with increasing nourishment volume. Apart from the nourishment volume, effects of other design parameters such as the alongshore length ($y_n$) and the nourishment elevation ($z_n$) have not been included. Furthermore, the intra-nourishment lifetimes have not been reported, thereby incorrectly assuming a constant nourishment lifetime in the alongshore dimension (Dean 2002). The objective of this study was therefore to develop a data-driven methodology 1) to quantify nourishment lifetimes in different data sets and different morphological systems and 2) to relate the nourishment lifetimes to the nourishment design parameters. The aim is to provide additional design support for nourishments by an improved understanding of the most influential design parameters for the nourishment lifetime. In order to quantify the nourishment lifetime, the study applied/developed:

1. A profile-based approach, to study nourishment lifetimes based on their effect on the natural morphological processes
2. A volume-based approach, to study nourishment lifetimes based on the sand volume

## 3.4 Profile-based approach to study shoreface nourishment lifetimes

The profile-based approach was developed to study the interferences of nourishments on natural morphological processes, more specifically the interference of shoreface nourishments on the migration of nearshore sandbars, as is present on Ameland (see also Table 3, Figure 4 and Figure 5).

A first step in the approach was the application of the Complex Principal Component Analysis (Joliffe 2002) to the data sets of, amongst others, the Ameland coastal profiles (Gijsman et al. 2018, 2019c). The statistical methodology aims to describe the complete data set of 52 coastal profiles (i.e. one per year between 1965 and 2017) with single 'components.' These components aim to describe most of the variance in the coastal profiles and consist of a spatial function ($f_k$) and temporal weights ($w_k$). The spatial function describes the 'statistically most-present' coastal profile in the data set, the weights its variation in time. Since the method was performed complexly (i.e. on the Hilbert transformed profiles), the components can be used to describe migrating patters. While the natural migration of the sandbars characterized most of the profiles in the data set, the first component was able to describe this natural behavior. Since the shoreface nourishments interrupted this behavior, the first component did not describe these effects. Hence, the periods of interruption (i.e. the shoreface nourishments lifetimes) were filtered from the data sets and could be quantified.

The methodology was applied to the coastal profiles in which nourishments were constructed on Ameland and other parts of the Dutch shoreline (Gijsman et al. 2019c). A relation between the design of 21 shoreface nourishments and their lifetime could not be identified. When the shoreface nourishment lifetimes were related to both nourishment design and the characteristics of the sandbar migration itself, using a linear system, a goodness-of-fit of $r^2 = 0.67$ could be reached. These findings indicate the influence of both natural processes and nourishment design on the shoreface nourishment lifetime. The most influential parameters in the relation were the nourishment density ($c_n$) and the nourishment depth location ($d_n$), which both increased the shoreface nourishment lifetime. The density of the migrating sandbars ($c_b$) and the bar cycle return period ($T_r$) (i.e. the timescale on which the sandbar migrate) affected the nourishment lifetime negatively (Gijsman et al. 2019c).

The developed methodology cannot interpret underlying physical processes, owing to its purely statistical character. The quantification of the shoreface nourishment lifetime in data sets with in-stationary morphological evolution was therefore hampered. The same holds for the data sets in which the behavior of different shoreface nourishments interacted with each other.

## 3.5 Beach nourishment lifetimes on Sylt

In the study of Gijsman et al. (2018) the profile based approach was applied to study the lifetime of beach nourishments placed in a single coastal transect on Sylt (transect 0+205, see Figure 5). Since no cross-shore migrating patters were present in the Sylt profile, the profile deviation from the mean was described with a standing wave pattern in the first spatial component. The first component described 88 % of the variance in the profiles and indicated that the profile deviation from the mean was similar in the cross-shore direction. Figure 6 presents the temporal weights of the first component, which indicates how much the profile deviated from the mean in meters. Temporal weights larger than zero indicate above-average profiles, and vice versa.

In this study, the beach nourishment lifetime is defined relative to a long-term average reference profile. Indicated nourishment lifetimes (grey areas in Figure 6) end at the zero-downcrossing of this reference. By relating these lifetimes with the nourishment design characteristics 1) density, 2) alongshore length, 3) elevation and 4) relative alongshore location ($rl_n$) for the 14 beach nourishments that were placed in this profile, with a linear system, a goodness-of-fit of $r^2$=0.89 was reached. The system, although based on a limited number of beach nourishments, identified that the nourishments with larger density, larger alongshore length and higher elevation increased the beach nourishment lifetime, while an increasing relative location reduced the beach nourishment lifetime. This shows that, for a certain nourishment, the nourishment lifetime decreases in the direction of alongshore sediment transport (Gijsman et al. 2018).

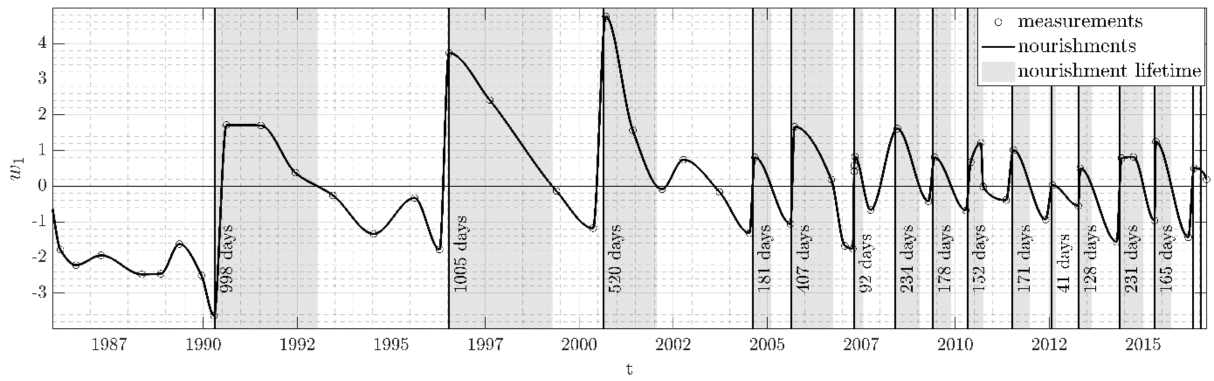


Figure 6: Temporal weights (Gijsman et al. 2018).

To increase the reliability of the abovementioned findings, the methodology was applied to the 691 coastal transects along Sylt (Gijsman et al. 2019b). Figure 6 shows the 691 coastal transects along Sylt (y) in combination with the timing of the measurements (grey crosses) and the beach nourishments (black lines). In total, 3290 transect-nourishment interactions were present in the data set. Since the beach nourishments on Sylt did not affect processes of natural migration, the volume-based approach was used. Hence, beach volumes were calculated for each measurement instead of temporal weights.

The beach volumes were calculated between horizontal limits, where the long-term average profile was equal to z = 0 m (mean sea level) and z = +4 m (dune foot). Values were interpolated linearly spatiotemporally on a grid with resolution of $\partial t = 7$ days and $\partial y = 50$ m. As a result of the spatiotemporal variability of the measurements in relation to the nourishments, the computation of the beach nourishment lifetimes was affected. Therefore, the following requirements were imposed for which the calculated lifetime was disregarded in the following cases:

1. If another nourishment was constructed in the same transect before the end of the nourishment lifetime.
2. If the calculated nourishment lifetime was shorter than 1 month or longer than 4 years.
3. If the lifetime-averaged erosion rate exceeded 150 $m^3$/m/year.

For the complete data set of Sylt, 899 nourishment lifetimes were calculated in this way. With the linear system a goodness-of-fit of $r^2 = 0.45$ could be reached, indicating that not all of the variability in nourishment lifetimes was described by the nourishment design parameters. With the focus on the Westerland coastal area alone, a linear system could describe the 153 nourishments lifetimes with goodness-of-fit of $r^2 = 0.70$. This indicates that the nourishment lifetime calculation is affected by 1) the nourishment design 2) the local morphological behavior and 3) the quality of the data set. Not only the quality of the data set varies along Sylt, but also the morphological evolution of the shoreline and nearshore sandbar varies between the north and the south (Gijsman et al. 2019a). This morphological variation was not included in the presented approach and reduces the capability to project the nourishment lifetimes. For the complete island as well as for Westerland, however, it was found that the nourishment density, alongshore length and profile elevation increase the nourishment lifetime, while the intra-nourishment lifetime decreases in the direction of alongshore sediment transport. For more details reference is made to Gijsman et al. (2019b).

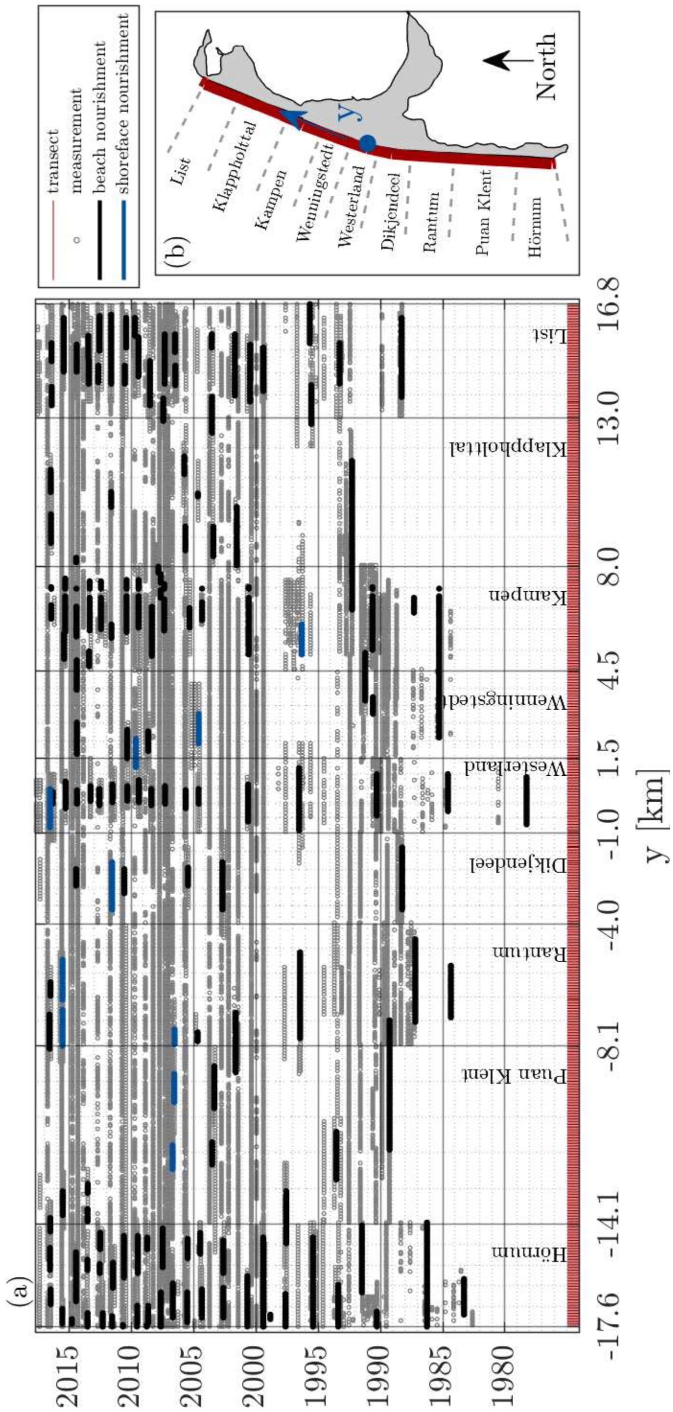

Figure 7: Spatiotemporal variability in measurements and nourishments along Sylt (Gijsman et al. 2019b).

Table 4: Number of computed nourishment lifetimes for Sylt and the Westerland coastal section, in combination with the fitted values of the linear system (Gijsman et al. 2019b).

| | Linear system characteristics | | | | |
|---|---|---|---|---|---|
| | no. | $k_1$ (d/m$^3$) | $k_2$ (d/m) | $k_3$ (d/m$^3$) | $k_4$ (d) |
| Sylt | 899 | 1.05 | 0.0673 | 3.57 | -69.4 |
| Westerland | 153 | 0.58 | 0.171 | 2.72 | -139 |

## 4 Improved methods for morphological changes under storm surge conditions

Existing models for coastal hydro- and morphodynamics (cf. chapters 2 and 3) use various parameterizations to estimate sediment erosion. For the simulation of morphodynamics, these models usually assume a uniform grain-size distribution, i.e. the erosion threshold and transport behavior of the sediment is governed by the median grain size $D_{50}$. In contrast, natural coastal sediment is usually widely graded (Holland and Elmore 2008), leading to complex interactions between different grain sizes. Several models have incorporated simple hiding-exposure coefficients, which lead to a modification of the erosion threshold in a bimodal mixture: Fine grains (hiding between larger grains) are harder to entrain from a mixed bed than from a uniform bed, whereas coarse grains (protruding into the flow) are easier to entrain than from a uniform bed. However, the interactions between different grain sizes have a variety of effects on the sediment matrix, near-bed flow velocities and erosion behavior (e.g. van Ledden et al. 2004, Venditti et al. 2010, Houssais and Lajeunesse 2012, Staudt et al. 2017, 2019b), and can subsequently affect the erosion stability of the bed. The processes underlying these effects are not fully understood, and the accuracy of existing sediment transport models remains limited.

Due to the complexity of the coastal environment, sediment transport under full-scale wave action is difficult to investigate in the field. Many studies have investigated sediment transport in the laboratory, especially in so-called oscillatory flow tunnels (OFT): These facilities mimic the oscillatory horizontal flow above the sediment bed, but neglect e.g. the variation of the flow profile with depth, the vertical flow component and undertow, i.e. flow processes which are induced by full-scale free-surface waves. Van der Werf et al. (2009) provide an overview of large-scale laboratory experiments that have investigated wave-driven sand transport until 2009 (the so-called SANTOSS database). Only 4 of the 26 studies covered in the database have investigated the transport of mixed sand (Inui et al. 1995, Hamm et al. 1998, Hassan 2003, O'Donoghue and Wright 2004a, b, Hassan and Ribberink 2005) – all of them conducted in oscillatory flow tunnels (OFTs). To account for all wave-induced processes and to avoid scale effects, experiments need to be carried out in large-scale wave flumes. Previous experiments with uniform sediment in the Large Wave Flume (GWK) in Hannover have shown that transport rates under free-surface waves can be 2–2.5 times higher than those measured in OFTs (Ribberink et al. 2000, Dohmen-Janssen and Hanes 2002). However, up to date, no experimental data for sand mixtures under full-scale free-surface waves is available.

If fine sediment is entrained under wave action or in a current it tends to develop the so-called sheet flow, a thin layer of moving grains above the bed. Coarser sediment tends to develop bed forms or ripples, which are washed out at very high flow velocities. It is assumed that sheet-flow conditions, e.g. caused by large waves and orbital flow velocities during storm-surge conditions, lead to significant erosion of the coastline. While the transition between ripple and sheet-flow regime and the resulting sediment transport rates can be determined with some accuracy for unimodal sediment, they are unknown for mixed sediment.

## 4.1 Laboratory experiments with mixed sand under full-scale wave action

To broaden the database and to systematically investigate the transport behavior of different sand mixtures in a controlled environment, experiments under full-scale free-surface waves were conducted in the Large Wave Flume (GWK) in Hannover in spring 2018. Fine ($D_{50}$ = 0.21 mm) and coarse sand ($D_{50}$ = 0.58 mm) was mixed in four different ratios (100:0, 68:32, 46:54, 26:74 %) and subjected to two different, regular wave conditions (WC1: $H_1$ = 1.5 m; WC2: $H_2$ = 1.0 m; T = 7 s; h = 3.5 m). Each experiment comprised five intervals with 200 waves each. A range of instrumentation was used to investigate near-bed flow velocities, sediment transport processes and bed morphology in the middle of the 30 m long test section. The overall changes in bed morphology, as recorded by an echo-sounder before and after each wave interval, were used to derive the net transport rates. Table 5 shows that the net transport rate generally decreased with a decrease of fines (i.e. an increase of $D_{50,mix}$) and, as expected, transport rates under the high waves (1.5 m) were larger than under the low waves (1.0 m). The experimental procedure and first results are outlined in further detail in Van Der Werf et al. (2019).

Table 5: Experimental conditions, resulting net transport rates and net transport rates as calculated by the SANTOSS model (van der A et al. 2013).

| Experiment No. | Sand | Fine fraction (%) | $D_{50,mix}$ (mm) | H (m) | T (s) | Net transport rate $q_s$ (mm²/s) | |
|---|---|---|---|---|---|---|---|
| | | | | | | Experiment | SANTOSS |
| 0516 | A | 100 | 0.217 | 1.5 | 7.0 | 76 ± 11 | 67 |
| 0518 | | | 0.217 | 1.0 | 7.0 | 38 ± 12 | 23 |
| 0525 | B | 68 | 0.238 | 1.5 | 7.0 | 101 ± 22 | 77 |
| 0530 | | | 0.238 | 1.0 | 7.0 | 31 ± 27 | 26 |
| 0605 | C | 46 | 0.439 | 1.5 | 7.0 | 64 ± 24 | 64 |
| 0608 | | | 0.439 | 1.0 | 7.0 | 13 ± 16 | 69 |
| 0615 | D | 26 | 0.566 | 1.5 | 7.0 | 48 ± 6 | 95 |
| 0620 | | | 0.566 | 1.0 | 7.0 | 8 ± 10 | 5 |

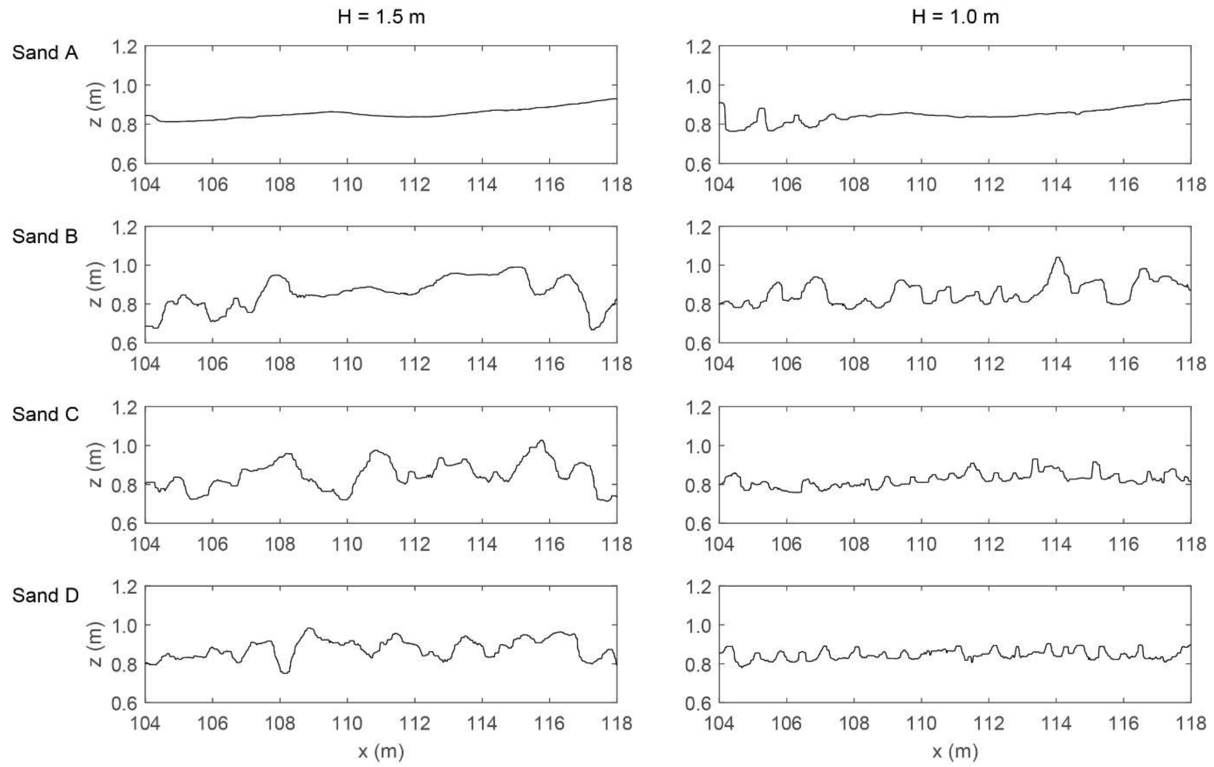


Figure 8: Final bed profiles (after 1000 waves) at transect y = 3 m for all experiments (mod. after Van Der Werf et al. 2019). The Large Wave Flume is 5 m wide.

As can be seen in Figure 8, the bed morphology changed depending on the sediment mixture: The addition of coarser sediment resulted in the development of bed forms, which became smaller and more regular with increasing content of coarse sand. Sand A and B (100 and 68 % fines) were mostly transported in the sheet-flow and transitional regime (where sheet flow develops on top of large, irregular bed forms). Sand C and D (46 and 26 % fines) showed ripple development; sheet flow could no longer be detected by the instrumentation in the middle of the test section.

## 4.2 Improvement of a sediment transport model

The net transport rates from the wave-flume experiments were used to test and optimize the existing SANTOSS model (van der A et al. 2013) for sediment transport under full-scale wave action. The model is based on OFT experiments and two studies with unimodal sediment that were conducted in the GWK. In contrast to other equations which average the sediment transport over the wave cycle, the SANTOSS model resolves the transport for each half-cycle (crest and trough) and can account for the phase-lag, which describes the exchange of suspended sediment from crest to trough half-cycle and vice versa. If required, the model uses a simple hiding-exposure formulation to account for changes in grain entrainment for sediment mixtures.

In a first step, the experimental parameters and measurements (e.g. wave height, orbital flow velocities, grain sizes) were used as model input in an attempt to reproduce the net transport rates of the experiments (Van der Werf et al. 2019). Both hiding-exposure as well as the phase-lag effect was used in this calculation. A comparison of the model results with the experimental observations shows several shortcomings (Figure 9a).

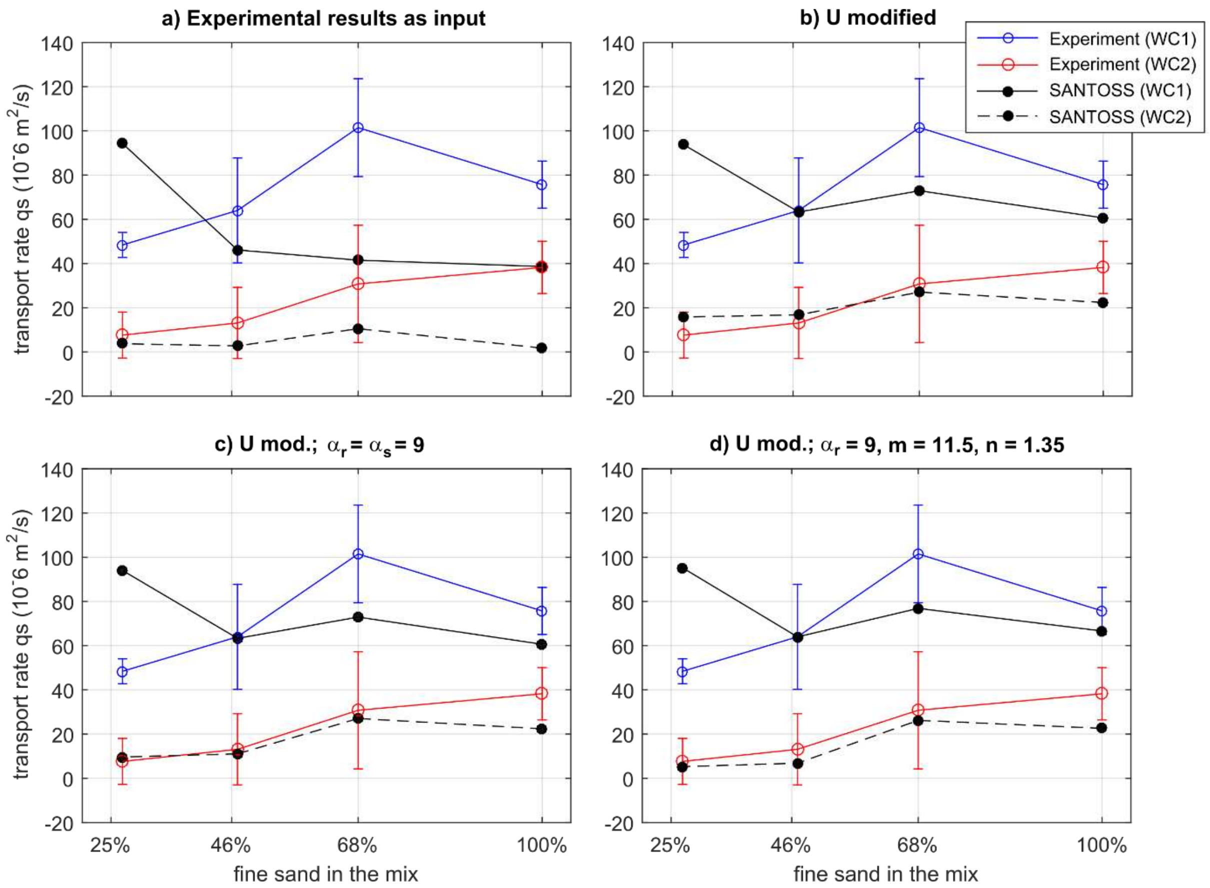


Figure 9: Measured (incl. standard deviation) vs. calculated net transport rates using different input velocities and calibration coefficients: a) original model with ADV velocities as input (cf. Van der Werf et al. 2019), b) modified $U_{on}$ and $U_{off}$ (cf. text), modified $U_{on}/U_{off}$ and calibration coefficients $\alpha_r$ and $\alpha_s$ (phase-lag), d) modified $U_{on}/U_{off}$ and calibration coefficients $\alpha_r$, $\alpha_s$, m and n.

The general trend of increasing net transport with an increase in fine sediment is not reproduced by the model. Instead the model results for H = 1.5 m show an almost exponential decrease in net transport with increasing fine content. For H = 1.0 m the model underestimates the measured transport rates in most cases and shows no clear tendency.

For this first estimate of the model performance (Figure 9a) the orbital flow velocities from a point ADV at 1.30 m above the bed were used as input parameters, as the near-bed velocity profiles (which were recorded using an ACVP; Hurther et al., 2011) were still being processed. Due to boundary-layer streaming, the near-bed flow velocity is expected to be slightly larger than the free-stream velocity. In a second step, the model calculations were therefore repeated with higher input velocities, which were estimated according to the initial ACVP results for Sand A (1.5 m). The modified input velocities ($U_{1,on}$ = 1.44 m/s, $U_{1,off}$ = 1.07 m/s; $U_{2,on}$ = 1.09 m/s, $U_{2,off}$ = 0.78 m/s) yield a better fit between model and experimental result (Figure 9b). The transport rate is increased and shows a trend that is similar to the experimental observations. Different indicators for model performance and thus the reliability of the model results (e.g. the number of model results that lie within a factor 2, 5 and 10 of the measurements, coefficient of determination $R^2$, Brier Skill Score BSS, root-mean-square error RMSE) are shown in the statistical analysis (Table 6).

Table 6: Statistical analysis for SANTOSS model results and experimental results depending on input velocity and calibration coefficients $\alpha_r$, $\alpha_s$, m and n.

| $U_{input}$ | $\alpha_r = \alpha_s$ | m | n | fac2 | fac5 | fac10 | $R^2$ | BSS | RMSE |
|---|---|---|---|---|---|---|---|---|---|
| ADV1 | 8.2 | 10.9 | 1.2 | 37.5 | 87.5 | 87.5 | 0.276 | 0.631 | 34.026 |
| modified | 8.2 | 10.9 | 1.2 | 87.5 | 100 | 100 | 0.553 | 0.863 | 20.746 |
| modified | 9 | 10.9 | 1.2 | 100 | 100 | 100 | 0.581 | 0.865 | 20.605 |
| modified | 9 | 11.5 | 1.35 | 100 | 100 | 100 | 0.635 | 0.873 | 19.966 |
| modified | 9 | 12.5 | 1.35 | 87.5 | 100 | 100 | 0.635 | 0.856 | 21.287 |

The SANTOSS model can be calibrated with a variety of coefficients, which have been determined using the SANTOSS database (van der A et al. 2013). As this database contains mostly OFT experiments, it is expected that the original calibration tends to underestimate sediment transport under full-scale waves. The calibration coefficients $\alpha_r$ and $\alpha_s$ affect the phase-lag parameter: if this parameter exceeds 1, entrained sediment is exchanged between crest and trough half-cycle. Subsequently, if $\alpha$ is increased, the sediment exchange between half-cycles rises. If there is a lot of suspended material in the water column, the phase-lag can lead to an offshore sediment transport, as fine material which has been entrained under the wave crest is transported seawards under the wave trough. This negative transport leads to a lower total transport rate. A re-calibration of $\alpha_r = \alpha_s = 9$ leads to very good fit between model results and experiment measurements of H = 1.0 m (Figure 9c). Using this calibration coefficient, 100 % of the model results lie within a factor 2 (fac2) of the measurements (Table 6).

In the SANTOSS model, the sediment load $\Omega_i$ during each half-cycle i is proportional to the excess shear stress at the sediment bed:

$$\Omega_i = m(|\theta_i| - \theta_{cr})^n \tag{4.1}$$

where $\theta_i$ is the effective Shields number at the bed, $\theta_{cr}$ is the critical Shields number of the sediment in question, and m and n are calibration coefficients (m = 10.97, n = 1.2 in the original model). If the coefficients are adjusted slightly to m = 11.5 and n = 1.35, the performance of the model for mixed sediment under full-scale waves can be further improved (Figure 9d). For the larger wave condition, 50 % of the model results are within the standard deviation of the measurements. For the smaller waves, the model yields a result that lies within the standard deviation in 75 % of the cases. However, the calculated transport rate for Sand D (26 % fines) under 1.5 m waves still exceeds the experimental result by far. If m is increased to > 11.5, the performance for the other scenarios can be further improved; however, also the discrepancy for Sand D becomes stronger.

While the re-calibrated SANTOSS model yields plausible results for sheet-flow conditions (68, 100 % fines) and low orbital flow velocities, there are still strong deviations for high orbital flow velocities (H = 1.5 m). The ongoing analysis of the other experimental data will provide additional input for the improvement of the model. The database used for the calibration of the SANTOSS model contains only two other experiments with progressive surface waves, both conducted with uniform sand (Dohmen-Janssen and Hanes 2002, Schretlen 2012). In the last step of the project, the transport rates from these experiments were reproduced using the improved model (Figure 10). The transport rates for the fine sand experiments by Schretlen (2012) are systematically underestimated, while the transport rates for the medium sand experiments are overestimated. The transport rates for Dohmen-Janssen and Hanes's (2002) experiments are slightly underestimated, but are within a factor 2 of the measured results. The ongoing data analysis will provide further results to improve the calibration of the SANTOSS equation.

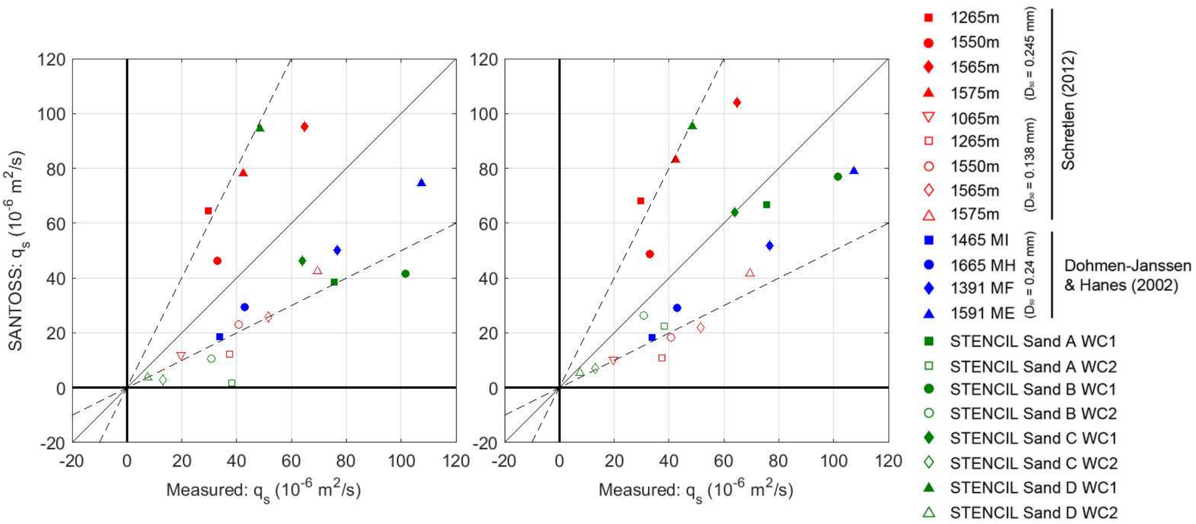


Figure 10: Comparison of STENCIL results with other experimental results with full-scale waves (Dohmen-Janssen and Hanes 2002, Schretlen 2012).

## 5 Monitoring sedimentary behavior of benthic habitats at a marine extraction site: Westerland II/III (German Bight, North Sea)

Information regarding the sustainability and the impact of sand mining in the North Sea off the West Coast of Sylt are sparse. Zeiler et al. (2004) and Mielck et al. (2018) revealed that the dredging pits serve as sediment traps where fine-grained material (mud) accumulates. As the fine material is unsuitable for beach nourishment, no further extraction activity at the upper layers is possible in these areas. In order to get more precise information about the impacts of sand extraction activities on benthic habitats, the largest marine sediment extraction site in the German Bight, "Westerland II & III", was monitored over a period of three years using hydroacoustics as well as direct measurements such as underwater videos and grab samples.

### 5.1 Methods and study site

The study site is located approx. 7 km off the western coast of the island of Sylt and has a size of ca. 15 km² including both the dredging area itself and the adjacent unaffected seafloor as a reference for pre-dredging conditions (Figure 11).

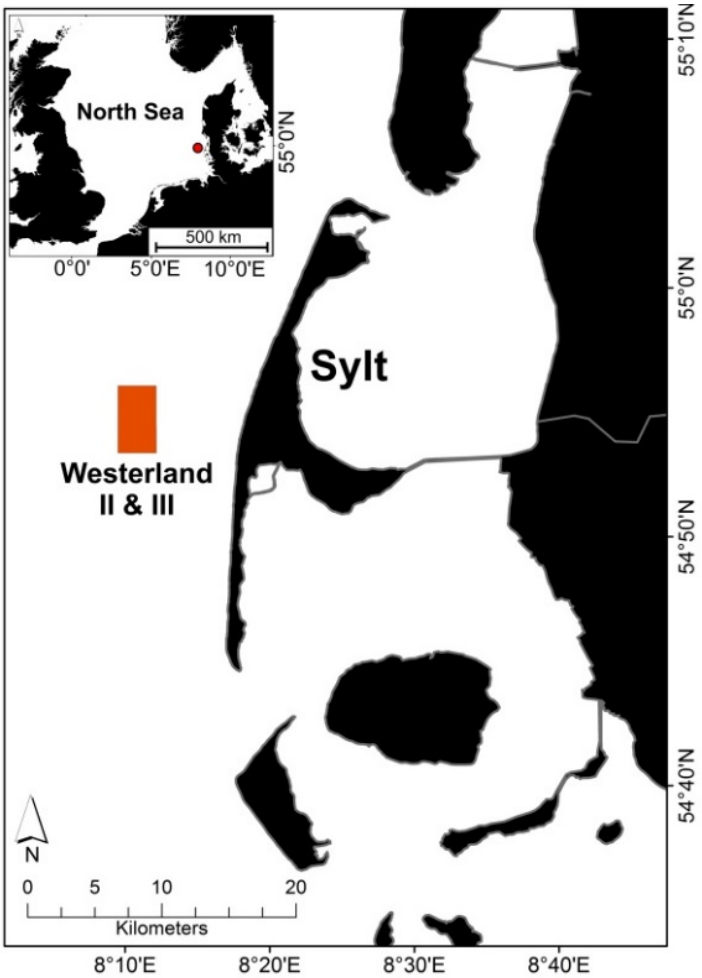


Figure 11: The study site "Westerland II & III" located west of the North Frisian Island of Sylt.

The research vessels "Mya II", "Heincke" and "Alkor" were used to collect hydroacoustic data as well as sediment samples and underwater videos in the study area. Table 7 shows all used devices and their specifications.

During the surveys, sidescan sonars with different frequencies and multibeam echo-sounders were used to map the prevailing seafloor conditions. For ground truth purposes, sediment samples and underwater videos were taken to obtain more detailed information about the habitat characteristics. In order to investigate short-term back-filling processes at the fresh excavation pits, gapless bathymetric mapping was conducted approximately semiannually between 2016 and 2019.

Table 7: Data acquisition in the years 2016 to 2019.

| Vessel | Date | Devices | Frequency | Grab Samples (Van Veen) | Underwater videos |
|---|---|---|---|---|---|
| RV *Mya II* | Sep. 2016 | SB<br>YEL*<br>STA* | 180 kHz<br>330 kHz<br>1000 kHz | 10 | 4 |
| RV *Mya II* | Apr. 2017 | SB | 180 kHz | - | - |
| RV *Mya II* | Aug. 2017 | - | - | 8 | 2 |
| RV *Heincke* | Dec. 2017 | KON | 100 kHz | - | - |
| RV *Mya II* | Mar. 2018 | SB | 180 kHz | - | - |
| RV *Mya II* | Nov. 2018 | SB | 180 kHz | - | - |
| *RV Alkor* | Jan. 2019 | SB<br>YEL*<br>STA* | 180 kHz<br>330 kHz<br>1000 kHz | 53 | - |

**SB** = SeaBeam 1180 multibeam echosounder; **KON** = Kongsberg EM710 multibeam echosounder; **YEL** = Imagenex YellowFin 872 sidescan sonar; **STA** = StarFish 990F sidescan sonar.
*raw data are not presented in this article

## 5.2 Results

The results of our bathymetric mapping campaign within the framework of STENCIL are shown in Figure 12. Backfilling processes within the freshly dredged pits are clearly visible at 54°54'N und 8°10'E. Mining in these pits was terminated in September 2016. The excavation of a new pit started in 2017 in the center of the study site (54°54'N und 8°11'E). Over a 2-year period, a depression of approx. 500 m width and 20 m depth (below seafloor) was created.

The investigation revealed a slight backfilling, which occurred a few weeks after the sand mining had started. This was mainly caused by slope failures, which appeared at the steep slopes of the pits in this early stage (Mielck et al. 2018, 2021). The material involved generally consisted of fine sand. After the pit's walls had stabilized, mud accumulated within the pit due to the decreased current speed (resulting from the greater water depth, i.e. a greater hydraulic diameter). Sand transport ceased at the rims of the depressions. Our monitoring activities confirmed that the mud accumulation within the pits is very slow. For example, in the excavation pit in the north of the investigated area (54°55'N und 8°10'E), which was exploited in the 1980s and 1990s, no backfilling could be detected in our bathymetric data. The same applies for the dredging area in the south of the study area where sand mining was discontinued in 2008.

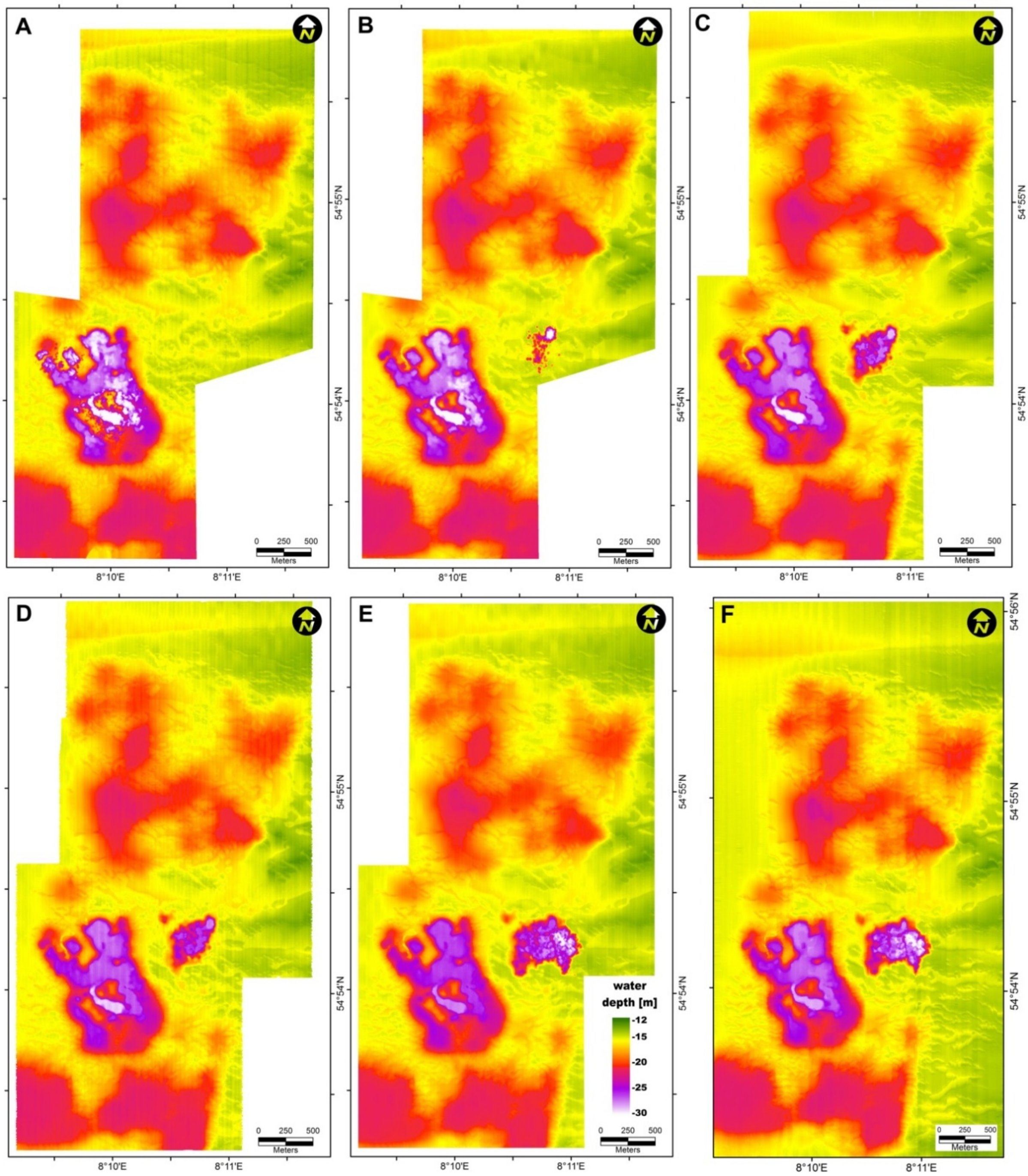


Figure 12: Bathymetry of the study site. All visible depressions are caused by sand mining since 1984. Data was collected in September 2016 (A); April 2017 (B); December 2017 (C); March 2018 (D); November 2018 (E) und January 2019 (F). Changed after Mielck et al. (2021).

## 5.3 Discussion

Using sidescan sonars, underwater footage and sediment samples for grain-size and benthos-analyses, it was possible to create habitat maps of the investigated area. These maps show no more coarse material in the deeper areas of the pits. The removal of coarse sand from the dredging sites triggered a long-term or maybe also permanent change in sediment composition and the associated benthic communities (Mielck et al. 2021). While the dredging process is still active, turbulent conditions prevent the establishment of a lasting habitat and fauna. After termination of the activities, slope failures maintain a sandy environment; however, soon after, sluggish mud accumulation starts which significantly changes the habitat. Since the coarse material is rather immobile on the seafloor (Tabat 1979, Werner 2004, Mielck et al. 2015), a natural regeneration is deemed unlikely in the coming decades or centuries. This is in accordance with Zeiler et al. 2004, who conclude that the benthic communities will not recover before the native sediment characteristics are restored. However, when the habitat disturbance continues, domination of opportunistic species and predators may occur resulting in a complete change of the habitat (Greene 2002, de Jong et al. 2015).

Our analyses revealed that the sand extraction significantly reduced both the macrozoobenthic abundance and the species diversity. Moreover, a change in the community composition was observed: mud-preferring species profited from the habitat change while sand-preferring species became less abundant or even disappeared. A deeper analysis of these effects is was conducted in Mielck et al. (2021).

From a holistic point of view, the regeneration potential of the study area towards the pre-dredging conditions is difficult to assess. As a matter of fact, the first sand mining activities are still visible on the seafloor after 35 years. If the sedimentation rates within the pits were similar to e.g. those of depressions east of Helgoland (2–18 mm/year, ca. 80 km south of the study area), a complete refill would take many decades to centuries (Dominik et al. 1978, von Haugwitz et al. 1988) while the coarse-sand characteristics are lost. However, the dredging area offshore of Sylt is quite small and a wider-ranging disturbance of the surrounding environment was not observed.

The bathymetric maps showed that there is still mineable material between the already exploited dredging pits (Figure 12). Hence, it might be possible to continue dredging in this designated area in order to keep the extraction area as small as possible. Based on estimations by Temmler (1983, 1994), approx. 1500 $Mm^3$ of suitable sediment for beach nourishment is available in an area of ca. 100 $km^2$ around the study site. This is enough material to protect the island for the next centuries and to budge from the current approach might not be recommendable. However, facing a near-future sea-level rise with increasing erosion rates, to protect the coastline with nourishments in a sufficient and cost-effective way may become difficult and only hard coastal protection measures (such as dikes, sea walls or concrete units) are an alternative.

## 6 Experimental assessment of hydrotoxicological impacts of dredging activities and shore nourishments

Sand extractions and sand nourishments and the associated relocation of sediments can lead to morphodynamic, sedimentological and ecotoxicological processes and interactions. Therefore an interdisciplinary approach, based on the hydrotoxicology method developed by Cofalla (2015), was applied to assess the combined effects of sediment dynamics and ecotoxicological processes due to anthropogenic interventions of sand nourishments and dredging. Especially the extraction sites are prone to ecotoxicological effects due to the deposition of fine sediments, which can bind organic as well as inorganic pollutants.

To investigate possible hydrotoxicological effects of sand mining activities, extensive sampling was conducted at the extraction area Westerland II in cooperation with AWI during two measurement campaigns in August 2016 and February 2019. A Van Veen grab sampler was used to take the sediment samples at water depths between 15 and 20 meters. More than 1000 kg of sediment were collected during both measurement campaigns. The sediment samples were taken at locations where dredging has been carried out as well as at undisturbed locations for reference. In addition, comparative investigations were carried out with ecotoxicologically relevant sediment samples from the port of Hamburg and fluvial reference sediments from the river Rhine, which has representative sediment properties for natural fine sediments (Höss et al. 2010, Hudjetz et al. 2014). The sampled sediments were filled into barrels with a capacity of 80 l. To reduce the biological activity, the barrels were stored at 4°C in a cooling chamber at the Institute of Hydraulic Engineering of RWTH Aachen University (IWW).

The interactions between suspended and deposited sediments, pollutants and the surrounding water are influenced by a wide range of different physical-chemical parameters. To determine the interactions, a comprehensive characterization of the sediments with regard to components, mineralogical composition and particle size distribution is necessary (Bryan and Langston 1992, Mitchener and Torfs 1996, Black et al. 2002). Therefore, extensive sediment analyses were carried out to characterize the considered sediments.

The water content was determined by oven drying the samples at 105 °C until mass consistency was reached (DIN EN ISO 17892-1) and ranged between 60 and 75 %. The particle size distribution was determined by laser diffraction according to ISO 13320. The results confirmed previous investigations by Mielck et al. (2018) and Zeiler et al. (2004), which showed that the morphological changes due to sand dredging at the extraction sites have led to a deposition of fine materials in the extraction pits. These fine sediments affect the consolidation behavior of the sampled sediments, which was measured by means of an Ultra High Concentration Meter (UHCM). While the sediment concentration of the reference sediments show a constant concentration with increasing depth of the sediment bed, the fine sediments sampled inside the extraction pits exhibit an increasing compaction over depth. This in turn affects the resuspension behavior and simultaneously influences transport processes at the sediment-water interface and inside the seabed. Together with altered digging and burrowing properties of the seafloor, this leads to effects on benthic communities. Another important parameter for sediment transport behavior and the calculation of erosion rates is the critical bottom shear stress, $\sigma_{crit}$, which defines incipient motion for sediment beds and was determined with the help of the EROSIMESS measurement system. A motor-driven propeller generates a flow in a test cylinder, which induces a defined shear stress on the sediment bed depending on the rotational speed of the propeller. The critical bottom shear stress is determined by gradually increasing the bottom shear stress levels until sediment motion sets in.

The results for particle size distribution, critical bottom shear stress as well as total organic carbon and sulfur content of the sediment samples taken from a dredging pit and four reference sites at extraction area Westerland II in August 2016 are given in Table 8 together with results for samples from the port of Hamburg and river Rhine, which were used in previous studies (e.g. Höss et al. 2010, Hudjetz et al. 2014) and therefore represented well examined reference sediments. All experiments in the flume were performed using salinated tap water.

Table 8: Clay, silt and sand content, critical shear stress, total organic carbon and sulfur content of investigated sediments from Westerland II (August 2016), port of Hamburg and river Rhine.

| Sediment | Clay (%) | Silt (%) | Sand (%) | $\sigma_{crit}$ ($N/m^2$) | TOC (%) | Sulfur (%) |
|---|---|---|---|---|---|---|
| Westerland II | 16.17 | 50.19 | 33.64 | 0.3 | 3.04 | 0.498 |
| Westerland II R1 | 1.6 | 13.1 | 85.3 | 0.156 | 0.12 | 0.028 |
| Westerland II R2 | 1.2 | 8.2 | 90.7 | 0.23 | 0.04 | 0.027 |
| Westerland II R3 | 12.4 | 72.4 | 15.2 | 0.15 | 0.46 | 0.084 |
| Westerland II R4 | 0.9 | 3.5 | 95.6 | 0.21 | 0.11 | 0.037 |
| Port of Hamburg | 6.04 | 48.15 | 45.81 | 0.05 | 0.05 | 0.199 |
| River Rhine | 15.03 | 64.00 | 20.97 | 0.15 | | |

$\sigma_{crit}$: Critical bottom shear stress; TOC: Total Organic Carbon

Effects on chemical-physical water quality parameters due to resuspension of the deposited fine material at the extraction sites were investigated in an annular flume (d = 3.25 m; w = 0.25 m) where an infinite, stationary and turbulent flow is generated by opposite rotation of a lid on top of the water and the flume itself. After the sediment bed was installed in the flume, it was left for a 5-day consolidation phase before the actual experiments begun. The flow, i.e. the bottom shear stress, was gradually increased in steps of 0.1 $N/m^2$ over a total duration of 18 hours. Water turbidity was continuously measured via redundant turbidity sensors and environmental parameters (pH value, redox potential, electrical conductivity, oxygen content) were monitored in an external measuring cell which is continuously flown through. Additionally, discrete samples were taken regularly during the experiments to determine the suspended matter concentration.

All tested sediments showed beginning erosion at bottom shear stresses between 0.2–0.3 $N/m^2$, in agreement with the values of $\sigma_{crit}$ determined with the EROSIMESS measurements (cf. Table 8). The resuspension of fine sediments interacts with the surrounding fluid and already at low erosion rates, a strong reduction in light transmission was observed, which can lead to negative effects on marine flora and fauna. Also chemical-physical water quality parameters, like redox potential and dissolved oxygen content, showed high correlation with the suspended particulate matter content (SPM). With increasing bottom shear stress, i.e. increasing SPM, dissolved oxygen decreased, demonstrating the very high oxygen consumption due to sediment resuspension. Thus, if deposited sediments from the extraction pits are resuspended, effects on chemical-physical water quality parameters and therefore on the entire aquatic system have to be expected.

To assess the ecotoxicological potential of the characterized sediments the teratogenic effects on fish eggs of *Danio rerio* have been analyzed according to DIN 38415-6 (DIN 2001). *Danio rerio* is a standard organism in ecotoxilogical studies and used for freshwater as well as brackish and marine sediments (Johann 2020). Sediments were freeze-dried and analyzes were performed in sediment contact tests (Hollert et al. 2003, Zielke et al. 2011). Table 9 shows the 'no observed effect concentration' (NOEC), 'lowest observed effect concentration' (LOEC) and the lethal concentration when 50 % of the fish eggs died ($LC_{50}$), in percent sediment. Corresponding to the increase of total organic carbon (TOC), the toxic effects on fish embryos was highest at the former sand dredging area of Westerland II, compared to three different reference sites outside of this area (Table 9). This applies also compared to sediment from the Port of Hamburg, an area which is historically highly polluted by contaminant loads from the river Elbe.

Table 9: Effect values of the teratogenic toxicity on fish eggs of *Danio rerio* assessed in the sediment contact test after 96 hours of exposure.

| Sediment | NOEC (% sediment) | LOEC (% sediment) | $LC_{50}$ (% sediment) | $LC_{50}$ (mg/ml) |
|---|---|---|---|---|
| Westerland II | 10 | 12,5 | 16,18 | 8,9 |
| Westerland II R1 | ≥100 | 100 | n.t. | n.t. |
| Westerland II R2 | ≥100 | 100 | n.t. | n.t. |
| Westerland II R3 | 25 | 50 | 82,13 | 430,7 |
| Westerland II R4 | ≥100 | 100 | n.t. | n.t. |
| Port of Hamburg | 25 | 50 | 61,56 | 32,8 |

*No observed effect concentration* (NOEC), *lowest observed effect concentration* (LOEC) and lethal concentration when 50 % of the fish eggs were dead ($LC_{50}$). n.t.: No toxicity.

One criterion of validity in the fish egg test is a constant oxygen concentration of ≥ 3 mg/l of the test medium to ensure a proper development of the organisms (Strecker et al. 2011). Marine sediments and sediments with high sulfur contents in general tend to result in oxygen deficiencies in the test medium, which was also observed in the erosion tests in the annular flume with the sediment from the dredging zone Westerland II. By using an optical measuring method with a micro oxygen sensor, the oxygen levels during the sediment contact test were constantly recorded. Figure 13 shows the oxygen levels for different sediment concentrations mixed with quartz sand.

For the sediment of Westerland II a rapid decrease of the oxygen concentration for all sediment concentrations within the first 24 h was observed, with a following stabilization above the required values for the sediment contact test after approximately 24 h. The results confirmed the potential of oxygen deficiency conditions at dredging sites, which might result in negative impacts on marine organisms.

To exclude the influence of oxygen consumption on the development of fish embryos and to depict a worst-case scenario regarding the ecotoxicological potential, additional fish egg tests (FET) with *Danio rerio* were performed with sediment extracts obtained by using Pressurized Liquid Extraction (PLE). The results for the lethal concentration $LC_{50}$ (in mg sediment equivalent (SEQ) in ml test medium) are given in the last column of Table 9 and are in accordance with the results of the sediment contact tests. Since oxygen consumption could be excluded under these test conditions, toxic potential could be attributed to pollution loads in the sediments at the dredging site.

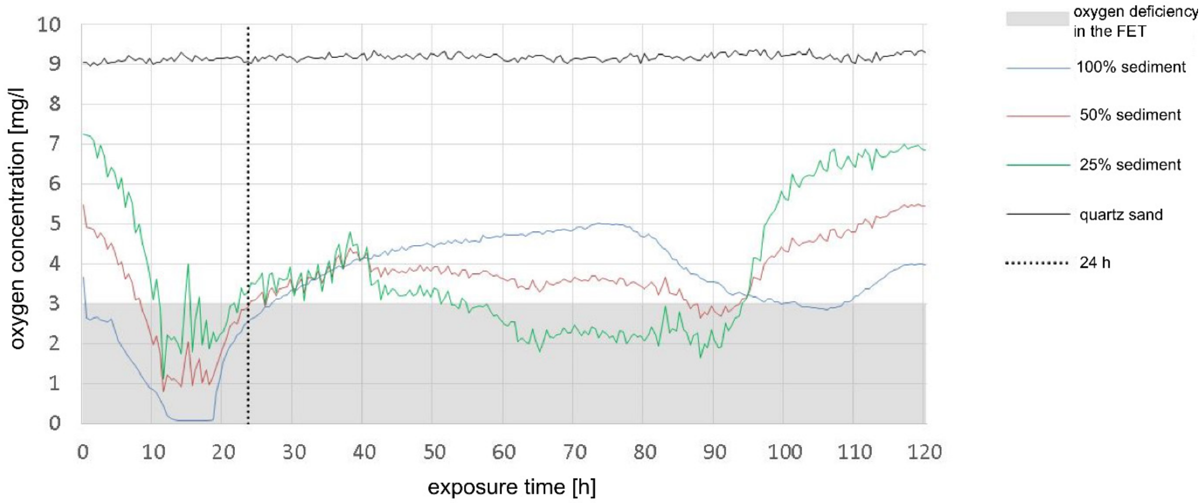


Figure 13: Oxygen concentration of the test medium (mg/l) of the sediment contact test with Danio rerio for different sediment concentrations of the Westerland II sediment (%) mixed with quartz sand over 120 h.

The results of the potential of teratogen effects on fish eggs and oxygen deficiency go hand in hand with investigations of benthic organisms of the described sediments. In contrast to the reference sediments, no benthic invertebrates were observed in the sediment of Westerland II.

The sediment campaign performed in 2019 provided an opportunity for the assessment of different extraction holes of the dredging area Westerland II, with different time points of termination of dredging activities within the last 36 years. Sediment extracts obtained via PLE were used to assess potential toxic effects on in vitro cell viability. Cytotoxicity was investigated by means of the Neutral Red Assay with permanent fish liver cells (RTL-W1) according to Borenfreund (Borenfreund and Puerner 1985, Borenfreund and Shopsis 1985) and modifications by Keiter et al. (2006) and Klee et al. (2004).

The results (EC50 values in mg SEQ/l test medium) are given in Table 10 and indicate higher cytotoxic effects for sediments of the former dredging sites compared to the reference sites. Since sampling site 2 – where dredging activities ended in 2017 – showed values comparable to the reference sites, it can be assumed that pollution pressure increases over time due to deposits of carbon bound chemicals in the dredging holes.

Table 10: Cytotoxicity (EC50 values in mg SEQ/l test medium), total organic carbon (TOC) and sulfur content (%) of the investigated sediments of the year 2019.

| **Sediment** | **$EC_{50}$ (mg/l)** | **TOC (%)** | **Sulfur (%)** |
|---|---|---|---|
| RS1 | 36.47 | 0.04 | 0.03 |
| 1 (1984) | 0.19 | 2.63 | 0.43 |
| RS2 | 2041.74 | 0.06 | 0.03 |
| 2 (2017) | 7.50 | 0.16 | 0.05 |
| 3 (2009) | 0.12 | 3.13 | 0.46 |
| 4 (2009, Depth: 30 m) | 0.21 | 3.26 | 0.50 |
| RS3 | 2.34 | 0.08 | 0.05 |
| RS4 | 54.44 | 0.10 | 0.04 |

This assumption is supported by the TOC content in the sediments (Table 10). Except for sampling site 2, the highest TOC concentrations were found in dredging areas from 1989 to 2009 (sites 1, 3 and 4). However, slightly elevated TOC values compared to reference sites indicate the beginning of organic deposit at site 2 within the following two years after the dredging. Moreover, the TOC contents positively correlate with the sulfur content. For the dredging sites the sulfur and TOC contents were comparable to the sediment of 2016 (Table 8). Therefore, embryo toxic effects and oxygen consumption conditions at sites 1, 3 and 4 are presumable.

Table 11 presents the sum concentrations of 7 indicator PCBs (PCB 28, 52, 101, 118, 138, 153 and 180) and 14 prominent PCBs (acenaphthene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benz(e)acephenanthrylene, benzo(ghi)perylene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, fluoranthene, indeno[1,2,3-cd]pyrene, phenanthrene and pyrene) obtained by chemical analysis. It can be stated that the former dredging holes accumulated organic chemicals, represented by the analyzed PCBs and PAHs, compared to reference site over time. Unexpectedly, reference site 1 showed the highest values for the sum of PCBs and high values for the sum of PAHs. The reason for such high concentrations of organic pollution at this site remains unclear. However, for site 2 the values for organic chemicals in the sediment were comparable to the chemical loads of the reference sites 2–4. Hence, chemical deposition at the dredging sites can be considered as steady process over time.

Table 11: Sum concentrations of 7 indicator PCBs and 14 PAHs in ng per g dry weight sediment [ng/g dw sediment].

| | Reference sediments | | | | Sampling sites | | | |
|---|---|---|---|---|---|---|---|---|
| | RS1 | RS2 | RS3 | RS4 | 1 (1984) | 2 (2017) | 3 (2009) | 4 (2009) |
| Total PCBs [ng/g] | 1393.58 | 746.44 | 720.68 | 690.19 | 1219.57 | 715.08 | 1070.86 | 1060.64 |
| Total PAHs [ng/g] | 2.07 | 0.76 | 0.16 | 0.21 | 2.52 | 0.15 | 2.10 | 2.27 |

Site 4 (2009) had a depth of 30 m.

In general, it has been proven that dredging activities offshore Sylt negatively impact the marine ecosystem. The consequences for aquatic organisms are manifold and range from altered benthic structures to potential ecotoxicological effects. Additionally, former dredging areas tend to severe oxygen depletion close to the bottom surface (Greene 2002). After termination of dredging activities the remained excavation holes act as a sink for organic pollution due to the accumulation of fine material with high contents of organic carbon and carbon-bound substances. This poses risk under conditions which promote remobilization of sediment and results in higher turbidities, oxygen deficiency and the potential bioavailability of environmental relevant chemicals in particular.

# 7 Towards an EAM for shore nourishments

## 7.1 Review of current practice

In the overarching, interdisciplinary part of STENCIL, the physical and ecological sustainability of nourishments as coastal protection measures was analyzed based on a comprehensive review of beach nourishments and marine sediment extractions in different countries (Staudt et al. 2020). A comparison of nourishment practice in several EU countries, the USA and Australia showed several differences in management strategies, legislative frameworks and technical aspects. Especially the countries neighboring the North Sea in mainland Europe have developed national long-term management strategies involving frequent beach and shoreface nourishments (sometimes complemented by dune nourishment/reinforcement). Other countries rely on remedial (one-off) nourishment activities or sediment recycling (i.e. transferring sediment from downdrift, accumulative areas to updrift, erosive areas) to restore the coastline for flood defense and touristic purposes.

Although beach nourishments have a number of advantages over "hard" coastal protection measures and are frequently referred to as "nature-based" or "green", they still present a significant disturbance of the environment. In order to control the disturbance of the natural environment, initial extraction or nourishment activities exceeding a critical size or volume require an Environmental Impact Assessment (EIA). The EIA should provide a comprehensive study of possible short- and long-term impacts of the project construction and operation on the natural environment and is evaluated by the responsible environmental authorities before the project is licensed (Carroll and Turpin 2002). However, the critical size/volume differs significantly between countries. As an example: nourishment activities in Spain require an EIA if they exceed a volume of 500 000 $m^3$ per proposed project, whereas nourishments in the Netherlands can be conducted without an EIA if they stay below an effective volume of 1.25 million $m^3$. In both countries the majority of nourishment activities thus do not require an EIA. In addition, regular re-nourishments of initial projects (e.g. as part of long-term nourishment strategies) are considered "maintenance" instead of new projects and can be conducted without further environmental assessment. The effectivity of the EIA as a tool to control negative environmental impacts of sediment extraction and nourishment activities therefore has to be reconsidered (Staudt et al. 2020). Several other studies have criticized EIAs due to a number of shortcomings (Peterson and Bishop 2005, Jay et al. 2007, Jha-Thakur and Fischer 2016), concluding that the EIA strategy itself is not capable of preventing environmental damage. For shore nourishments, especially the long-term environmental effects remain unknown and are not monitored except for few research projects. The gap in knowledge therefore remains, which hinders the development of long-term coastal management strategies that are in line with the EAM and a "good environmental status" as demanded by the Marine Strategy Framework Directive.

## 7.2 Analysis of current and future benefits and shortcomings

Despite the large uncertainties regarding the long-term ecological and morphological impacts of the activities, nourishments have become an integral part of coastal protection schemes across the world. Hence, they need to be designed and monitored carefully in order to understand their behavior and their possible impacts on the coastal and marine environment. As shown in the preceding chapters, several tools were developed or improved within the project STENCIL. Based on the review of nourishment practice and environmental monitoring (Staudt et al. 2020) and the work conducted within the individual subprojects (chapters 2–6), the strengths, weaknesses, opportunities and threats for various extraction and nourishment measures were identified and compared. While the strengths and weaknesses are based on the current state of knowledge, the opportunities and threats describe the possible long-term development and effects, considering other external factors. This so-called SWOT analysis can support practitioners and decision-makers to choose a suitable measure for their individual application. The SWOT analysis was conducted for a variety of extraction (i. deep extraction from the seafloor, i.e. static dredging; ii. seafloor surface dredging, i.e. dynamic dredging; iii. terrestrial sediment extraction) and nourishment measures (i. beach nourishment; ii. shoreface nourishment, iii. dune reinforcement; iv. mega-nourishment). The analysis focused on the physical (i.e. hydro- and morphodynamic) and ecological sustainability of the nourishment activities as well as on societal aspects (e.g. economy, tourism). This chapter presents a brief overview of the SWOT results for the different nourishment measures: beach nourishments, shoreface nourishments and dune nourishments (cf. Table 12 to Table 14). The full breakdown of the SWOT analysis for extraction and nourishment activities can be found in a freely-available German brochure (Staudt et al. 2019a).

*Beach nourishments* (Table 12) are placed directly on the beach, i.e. between the swash zone and the dune toe. The sand is distributed in the beach profile using e.g. loaders or bulldozers (Bird and Lewis 2015). The nourishment is usually designed to approximate the natural beach profile to maintain a natural beach slope and the associated wave breaking (Dean 2002). The sediment grain size and grading should be similar to the natural material: If the grading of the borrow material is too wide, steep cliffs can develop (McFarland et al. 1994). Both the construction and the finished nourishment can be directly seen by the public, which increases the public awareness for nourishments as coastal protection measures. As a beach nourishment widens the beach that is available to visitors, it increases the area's attractiveness. The sediment budget of the beach-dune system is directly increased, strengthening the protection function of the system as well as the function as a sandy habitat. The nourishment efficiency can be easily monitored using regular beach profiles. The environmental effects of beach nourishments vary with the location in the beach profile. The placement of sediment on the beach as well as the use of construction equipment (machines, pipelines) directly disturbs the local habitats, e.g. by burying. However, especially in the highly dynamic swash zone, several key species recover quickly from these disturbances (Menn et al. 2003, Leewis et al. 2012, Schlacher et al. 2012, Wooldridge et al. 2016). In the upper, dry parts of the beach profile, recovery rates are lower (Rakocinski et al. 1996, Janssen and Mulder 2005). The installation of pipelines and the subsequent profiling of the beach slope make beach nourishments more expensive and time-consuming than shoreface nourishments. In addition, the beach must be closed during the construction phase and cannot be used by the public. As beach nourishments are directly impacted by the waves, a strong storm event can erode large amounts of the nourishment volume, resulting in shorter lifetimes than for shoreface nourishments. In coastal management plans, beach nourishments are therefore designed for a lifetime of years to decades. The sudden erosion of a (new) nourishment can however lead to a negative public perception of the measure and lower the acceptance of nourishments as coastal protection measures.

Table 12: SWOT analysis for beach nourishments, i.e. the nourishment volume is placed on the beach.

| Strengths | Weaknesses |
|---|---|
| • Increases the sediment budget of the active coastal profile<br>• Limited impact on benthic organisms in the surf zone, as these are resilient to strong morphodynamics<br>• Immediate public perception of coastal protection measures<br>• Increasing recreational value | • No reduction of wave energy at the beach<br>• Temporary construction site required<br>• Relatively large effort, high costs<br>• Disturbances of the beach ecosystem through construction at the beach<br>• Low potential for full recovery of the ecosystem in the upper, dry parts of the beach profile |
| **Opportunities** | **Threats** |
| • Beach maintains its natural appearance<br>• Additional sediment for adjacent beaches (longshore sediment transport)<br>• Public engagement and awareness for coastal processes and protection measures<br>• Enhancement of the ecosystem as the sandy beach environment is preserved | • Wrong sediment composition can result in development of cliffs<br>• Sudden loss of nourished material, e.g. following storm events, can lead to negative public perception<br>• Changes of sediment composition and benthic habitats |

*Shoreface nourishments* (Table 13) are placed on the submerged part of the coastal profile (Mangor et al. 2017). The borrow material is usually dumped at the nourishment site directly from a dredging vessel. As construction activities at the beach are avoided, the activity is cheaper than the construction of a beach nourishment. In addition, the beach can still be used by the public during the construction phase. Shoreface nourishments reinforce offshore sandbanks, stabilize the coastal foundation and provide long-term sediment supplement to the coastal sediment budget (Sørensen et al. 2014). The nourishments facilitate offshore wave breaking, thus reducing the wave energy that arrives at the beach. The hydrodynamic effect of a shoreface nourishment is therefore similar to the effect of a breakwater (Hoekstra et al. 1997, van Duin et al. 2004, Sørensen et al. 2014). However, it is difficult to predict the morphodynamic behavior of the nourishment volume (Gijsman et al. 2019c). In addition, the hydrodynamic effects are reduced during high water levels, which dampen the effectiveness of the nourishment during extreme events like storm surges. At the same time, a shoreface nourishment is usually more resilient towards extreme events, whereas a beach nourishment can erode during a single strong storm event. Shoreface nourishments are considered an effective protection measure for tidally influenced coastlines (e.g. the North Sea), whereas they are less efficient in coastal areas with lower wave energy (Furmanczyk 2004, van Rijn 2010).

With increasing water depth, sediment grain size decreases and species abundance and diversity increase. Benthic organisms in these habitats are less resilient to strong sediment dynamics (e.g. burial) than species in the highly dynamic shoreline habitats, leading to longer recovery times after a disturbance (Rakocinski et al. 1996, Janssen and Mulder 2005).

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Habitats located in between several sandbanks or reefs are especially sensitive, as they usually offer high diversity and abundance (Janssen and Mulder 2005). Studies investigating the long-term impacts of shoreface nourishments are rare, as a monitoring of ecological processes in these areas is especially complex.

Table 13: SWOT analysis for shoreface nourishments, i.e. the nourishment volume is placed below mean water level.

| **Strengths** | **Weaknesses** |
|---|---|
| • Reduction of wave energy at the beach (during normal water levels)<br>• Lower effort than shore nourishment (construction site, vessel transit time)<br>• No construction equipment at the beach | • No wave damping during extreme water levels<br>• No additional sediment at the beach<br>• No direct increase of beach width, but possible long-term effects<br>• Benthic organisms at greater water depth are not adapted to strong sediment dynamics<br>• Low public visibility |
| **Opportunities** | **Threats** |
| • Possibility to influence local hydro- and morphodynamics<br>• In contrast to shore nourishment: Higher stability during extreme events | • Changes in morphodynamics can cause hydrodynamic side effects (e.g. rip currents)<br>• Long-term disturbance of habitats at the nourishment site |

*Dune nourishment* or dune reinforcement includes the placement of sediment in the upper beach profile or at the toe of the dune, i.e. directly increasing the width of the dune system (Mangor et al. 2017). With a larger sediment buffer, the dune system is protected from breaching during storm surges. Even in the event of erosion, the infrastructure behind the dunes is still protected from flooding. Unlike shoreface nourishments, dune reinforcements do not affect the wave energy at the beach. The morphodynamics of a dune depend merely on the wind and the related aeolian sediment transport. In contrast to beach and shoreface nourishments, designing and monitoring a dune reinforcement is comparatively simple. Planting additional vegetation (e.g. marram grass) can contribute to the stability of the dune, which safeguards the long-term coastal protection functions. However, additional vegetation also prevents the natural dynamics of a dune, which affects the ecosystem on the long term and could lead to a loss of biodiversity (de Groot et al. 2017).

The construction of a dune reinforcement is more expensive and time-consuming than for beach or shoreface nourishments. The initial influence on the beach width is low, as the nourished material only adds to the beach width through aeolian transport or through dune erosion in a severe storm event. Dune reinforcements are sometimes combined with beach or shoreface nourishments, e.g. within the coastal management concepts of Mecklenburg-Vorpommern (StALU MM 2009) or in large-scale management schemes like the Hondsbossche Duinen in the Netherlands. In contrast to the artificial stabilization of dunes, new management concepts focus on the restoration of the natural dune dynamics of the dune system, thereby facilitating biodiversity and the growth of the coastal system with sea-level rise (Löffler et al. 2013, Osswald et al. 2019).

Table 14: SWOT analysis for dune reinforcement.

| **Strengths** | **Weaknesses** |
|---|---|
| • Increased dune volume = direct protection (buffer) during flood events<br>• Easy dimensioning and monitoring | • No hydrodynamic effectiveness<br>• High effort (planting of marram grass) |
| **Opportunities** | **Threats** |
| • Public engagement and awareness for coastal processes and protection measures<br>• Change of the ecosystem through additional planting of vegetation | • Sediment not available for longshore transport<br>• Change of the ecosystem through additional planting of vegetation |

The SWOT analysis shows that the different nourishment techniques have benefits and drawbacks that need to be balanced depending on the local boundary conditions at the nourishment site and the requirements of the stakeholders (e.g. local communities or beachgoers). Future research results, especially regarding long-term ecological and morphological impacts of nourishment activities, can help turning *opportunities* and *threats* into known *strengths* and *weaknesses*, thus reducing the risk of the applications. With the remaining unknowns, coastal engineers and planners have to choose the best possible option for their respective application, considering all boundary conditions and resulting risks for the environment. It must be noted that the biological processes which are so far unknown might have additional impacts on the coastal ecosystem. Coastal protection is an active interference with the coastal environment and as such always implicates a risk for the ecosystem. The construction should therefore only be executed if it is essential for the protection of lives and property.

## 8 Conclusions

The research project STENCIL combined various methods and disciplines to provide an interdisciplinary view of the challenges faced by coastal managers and engineers when designing shore nourishments for the sustainable protection of developed coastlines. To address these challenges, STENCIL has provided improved numerical and analytical models (chapters 2, 3 and 4), laboratory procedures (chapter 6) and comprehensive datasets of laboratory experiments (chapter 4) and field measurements (chapter 5). The review of nourishment practice has revealed knowledge gaps and differences between country legislation and coastal management strategies. Based on the project results, the concluding SWOT analysis provides a tool for decision-makers and a basis for both upcoming research projects as well as for a strategy to establish an EAM for shore nourishments.

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