# APPENDIX 1C

# NAMIBIAN MARINE PHOSPHATE (PTY) LTD

BENTHOS

## Sandpiper Project

**Proposed recovery of phosphate enriched sediments from the Marine Mining Licence Area No.170 off Walvis Bay Namibia.**

**Environmental Impact Assessment Report for the Marine Component**


***Prepared by:***
*Mr. Jeremy Midgley (Pr.Sci.Nat)*
*J Midgley & Associates*
*In association with:*
*Enviro Dynamics*
*The CSIR*



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**March 2012**

# Dr. N Steffani: Benthos


Steffani Marine Environmental Consultant

21 Skippers End
Zeekoevlei 7941
South Africa

# SPECIALIST STUDY NO. 1C:

*Marine Benthic Specialist Study for a Proposed Development of Phosphate Deposits in the Sandpiper Phosphate Licence Area off the Coast of Central Namibia*

## *Project:*

*The Dredging of marine phosphate enriched sediments from Mining Licence Area No. 170*

## *Date:*

*March 2012*

## *Prepared for:*

*Namibian Marine Phosphate (Pty) Ltd.*

## *Contact details:*

*Dr Nina Steffani*
*Steffani Marine Environmental Consultant*
*Mobile: +27 72 217 2060*
*Email: nina@steffanienviro.co.za*

## *Declaration:*

I, *C. Nina Steffani* of Steffani Marine Environmental Consultant, do not have and will not have any vested interest (either business, financial, personal or other) in the proposed activity proceeding other than remuneration for work performed in terms of the South African Environmental Impact Assessment Regulations, 2010

# summary

Namibian Marine Phosphate (Pty) Ltd (NMP) has been awarded a 20-year mining licence (ML170), which is located on the Namibian continental shelf offshore Conception Bay in water depths ranging from 180 to 300 m covering a total area of 2233 km$^2$. Within the mineralized resource zones of the licence area, (also named Sandpiper licence area), three target areas have been identified, i.e. Sandpiper-1 (SP-1), 2 (SP-2), and 3 (SP-3), which are the focus of this EIA. SP-1 is in the north of ML170 in water depth from 190-235 m, SP-2 is in the center in depth 245-285 m and SP-3 is in the south at 235-270 m depth. Both SP-1 and SP-2 target areas are 22 km long x 8 km wide, while SP-3 is 11 km long x 6 km wide. NMP is proposing to dredge the uppermost 1-2.5 m (up to 3 m) of the seafloor in these target areas to recover phosphate rich material for use as fertilizer. The export target for the Sandpiper Phosphate project is 3 million tonnes (Mt) of 'rock phosphate' per annum, which requires the mining of 5.5 Mt of marine sediments.

During the public scoping process for this project, the need for a specialist report describing and evaluating the potential impacts of the dredging activities on the benthic communities inhabiting unconsolidated sediments was recognised. Dredging is destructive by nature and the removal of the sediment and with it the associated benthic fauna has been identified as one of the main impacts. This specialist study assesses the impacts associated with dredging in SP-1, SP-2 and SP-3 only and with the technical specifications provided. If at a later stage mining is planned to be undertaken outside these areas or other technologies are considered, this needs to be assessed in a separate study. As specified by the terms of references, a 'desktop' approach was adopted for this study. Besides an 18 station macrobenthic baseline study conducted in the target area SP-1 including a single station from SP-3, no further data on biotic communities or the physical environment (e.g. $H_2S$ concentrations, oxygen conditions, etc.) exist or were available at the time of writing this report. The study thus relies almost entirely on data from publicly accessible literature based on studies conducted in the larger general region. The only site-specific data are those from the above-mentioned macrobenthic survey and associated sediment analysis conducted in 2010.

## PROJECT DESCRIPTION

The preferred method of sediment recovery is the use of a Trailing Suction Hopper Dredge (TSHD). Dredging will be primarily on a north (N) or south (S) heading (swell dependent), recovering sediment in a ~ 3.0 m wide x ~ 0.75 m deep cut along a mine block (4 km length) of the larger target mine area. Mining will commence in SP-1, with subsequent mining in SP-2 and later possibly SP-3. To fill the hopper of 46,000 m$^3$ capacity, up to 6 parallel and adjacent 4 km long cuts will be made. During dredging, excess water with some fines will be discharged through an overflow pipe at 10-15 m below the surface. Once the vessel hopper is filled, the sediment will be pumped ashore. A typical dredge cycle is assumed to consist of 16-17 hours dredging to fill the hopper, and on average 20 hours to sail to shore for unloading and sail back to the operational location to initiate dredging again. Once back at the location, the vessel will continue to dredge within the particular 'cut' zone. Depending on the resource this will be 1-2.5 m deep (possibly up to a depth of 3 m). Underlying the phosphate resource is a stiff clay footwall, and the operational requirement is not to cut (dredge) into the footwall, but to rather leave a residual thickness of phosphatic marine sediments over the footwall of

0.3 m (approximately 10 % of the particular mine block thickness). As a result, the surface of the dredged area will be locally uneven or hummocked. Determined by the average resource thickness, the maximum annual mining rate is 3 $km^2$, which at a life of mine of 20 years amounts to a maximum total mined-out area of 60 $km^2$ or 2.7% of the total ML170 licence area, with a maximum potential dredged-depth of 2.5 m (possibly up to 3m) below the original seafloor.

## BASELINE ENVIRONMENT

Typical of coastal upwelling systems, the central Namibian shelf is characterised by the occurrence of natural shelf hypoxia, which is referred to as the oxygen minimum zone (OMZ). On the Walvis Bay margin, there are two shelf breaks at about 150 m and 300-400 m depths, which effectively divide the shelf into an inner and outer shelf. A significant feature of the central Namibian inner shelf is an extensive mud belt comprising organically rich diatomaceous oozes originating from planktonic detritus, which extends over 700 km in an N-S direction in approximately 50-150 water depth. The mud belt is characterised by severe hypoxic and often anoxic conditions and high toxic hydrogen sulphide ($H_2S$) concentrations in the upper sediment layers that support extensive mats of large sulphur-oxidising bacteria that reduce the flux of $H_2S$ into the water column by oxidising sulphide to sulphur with nitrate to obtain energy. Occasional $H_2S$ eruptions from gas pockets contained in the thickest parts of the mud belt (>8 m) can spread over large areas with disastrous effects on fish and other marine life.

Put into a regional context, ML170 and specifically the three target areas, are located in a generally sandy environment on the outer shelf beyond the inner shelf break, and thus offshore of the diatomaceous mud belt and south of a mid-shelf belt high in organic matter. As ascertained from the available literature, organic matter as well as nutrient concentrations in the sediments of the target areas are likely to be relatively low, which is a result of relatively strong bottom currents in this region, preventing the deposition of fine material. The target phosphorite deposits in the licence area are pelletal phosphate sands of Miocene age that are geographically distinct and have a different origin than the concretionary phosphorite that presently forms in the diatomaceous mud belt. Furthermore, the licence area lies at the southern offshore fringe of the OMZ, with perennial low dissolved oxygen levels (<0.5 ml/ℓ) at the bottom but typically not anoxic conditions. Hydrogen sulphide pore water concentrations, $H_2S$ fluxes from the sediments and $H_2S$ bottom water concentrations are likely to be very low, but it cannot be excluded that $H_2S$ concentrations in deeper sediments (>50 cm) may be higher.

Despite oxygen depletion, specialised benthic assemblages can thrive in OMZs and many organisms have adapted to low oxygen conditions by developing highly efficient ways to extract oxygen from depleted water. Within OMZs, benthic foraminiferans, meiofauna (animals between 0.1-1 mm), and macrofauna (>1 mm) typically exhibit high dominance and relatively low species richness. Macrofauna and megafauna (>10 cm) often have depressed densities and low diversity in the OMZ core, where oxygen concentration is lowest, but they can form dense aggregations at OMZ edges. Body size seems to be very important as small organisms are best able to cover their metabolic demands in the OMZ, and besides adaptation to low oxygen often have a capability to conduct anaerobic metabolism. Meiofauna may thus increase in dominance in relation to macro- and megafauna. Nonetheless, although small organisms prevail, the species inventory of OMZs comprises the whole range between micro- (<0.1 mm such as bacteria) and megafauna. Very little is known about the benthic fauna specific to the Namibian OMZ. Data from a macrofauna baseline survey in SP-1 have shown that overall species richness of the benthic macrofauna assemblages was relatively low and strongly dominated by polychaetes particularly the spionid polychaete *Paraprionospio pinnata*, which is the dominant species found worldwide in oxygen-constrained environments. Crustaceans, on the other hand, were both in terms of abundance and biomass very poorly represented. The phyla distribution is generally in common with other OMZs around the world. Most species found in the study area have a larger geographical distribution and/or have been recorded elsewhere from the Namibian and/or South African west coast. No data exist on meio- or microfauna (bacteria) composition in the target areas, but evidence from published data strongly suggests that concentrations of large sulphur-oxidising bacteria in the target areas are likely to be very low, if present at all.

## IMPACT ASSESSMENT

As a result of the dredging operations to recover marine phosphate resources in ML170, trenches will be excavated in the seabed and the benthic biota associated with the sediments will be removed. The mining licence is issued for a period of 20 years, and at a maximum dredging rate of 3 $km^2$ per annum, this will lead to a dredged area of 60 $km^2$, divided over the three target areas SP-1, SP-2 and SP-3. The total dredge depth will be on average 1.69 m, 1.70 m and 1.30 m for SP-1, SP-2, and SP-3 respectively. The maximum resource depths are 2.0 m, 2.25 m and 1.85 m for SP-1, SP-2 and SP-3 respectively (possibly up to 3 m). (There may be variability in the actual depth to which the resource will be mined in comparison with the thickness of the resource as determined from exploration). The most immediate effect of dredging is the loss of benthic organisms by the removal of the substratum, but a typical by-product of dredging activities is the re-suspension of sediments into the water column and the eventual re-deposition of this material. More specifically, aspects of the dredging activities that are considered include:

- The loss of benthic communities through removal of sediment during the dredging process;
- The effects of sediment removal on (re-) colonisation and recovery rates of impacted communities;
- Change in sediment characteristics due to dredging;
- The potential indirect effects of the loss of benthic communities on demersal fish in the area;
- The effects of re-deposition of suspended material; and
- Release of nutrients by dredging and its direct/indirect effect on benthic communities, and release of hydrogen sulfide from sediments during dredging.

Other specific concerns voiced during the Public Participation Process and summarised in the Scoping Report are:

- The removal of mats of large sulphur-oxidising bacteria and associated recovery rates;
- The possible proliferation of bacteria in an anaerobic environment, specifically the botulism causing bacterium *Clostridium botulinum*, and its subsequent contamination of fish and other wildlife (and possibly humans); and
- The possible release of hydrogen sulphide from the sediments by dredging.

This specialist report focuses on the benthic soft-bottom environment and therefore only discusses impacts relating to this habitat. Sources of risks to the pelagic environment, the water quality (e.g. turbid plumes, re-suspension of contaminants), and fish communities and associated fisheries are described in detail in other specialist studies (this EIA).

An assessment of the risks associated with the dredging activity identified nine potential negative impacts on the benthic biota in the three target areas or beyond. Of these, two impacts are considered to be of medium significance, six of low significance, and one is assessed as having no significance. The impacts of medium significance are:

***Impact*** - The removal of the upper 1-2.5 m (up to 3 m) of sediment by dredging will result in the loss of the benthic biota associated with the sediment. The exposed sediments are likely to be different to the original superficial deposits, and sediment refill rates at this depth are likely to be very slow. Colonising assemblages are likely to differ from those present prior to the dredging activity.

***Significance*** - Medium as the duration of the impact is permanent (exceeds life of mine) in view of recovery to original community but recovery to a different community but providing similar ecosystem services may occur sooner. The intensity is moderate to serious but the extent is confined to the actual mine sites, with a maximum of 60 $km^2$ after 20 years of dredging.

***Mitigation*** - Leave behind a residual sediment layer over the clay footwall of at least 30 cm to cover the clay footwall. Leave behind undredged areas to enable migration of mobile organisms from these areas into the mined areas.

***Impact*** - The depth of the dredged area might change local near-bottom hydrographical conditions and thus act as trap for very fine material. This could lead to high decomposition rates and consequently anoxic conditions and $H_2S$ concentrations in the sediments.

***Significance*** - Medium as the duration is permanent (beyond life of mine) and intensity moderate to serious, but extent is restricted to the mine sites and large areas of the inner shelf are naturally subjected to anoxic conditions.

***Mitigation*** - Leave behind a residual sediment layer over the clay footwall of at least 30 cm, which will reduce the depth of the dredged-out area.

Although the mitigation measures will facilitate the colonising of the newly exposed sediments, and may reduce the risk of large areas of the dredged sites becoming anoxic, the significance will remain medium after mitigation. This is due to the very long time scales anticipated for the recovery of the disturbed biota to its original status and the expected low infilling rates at this water depth. Functional recovery, defined as recovery to a community that provides similar ecosystem functions to those of the original community despite being different in composition, is, however, likely to occur sooner.

In general, the confidence level in the assessments is medium, as most of the impact evaluations are based on assumptions that are derived from publicly available literature data, whereas data from ML170 itself are very limited. A verification survey is therefore critical to confirm these assumptions, which should include sampling of meio- and macrofauna, surveying for presence of large sulphur-oxidising bacteria, measurements of sediment characteristics, organic matter and nutrient concentrations as well as $H_2S$ and oxygen concentrations. In case that the verification data reveal a substantially different habitat to that discussed in the environmental baseline description, the impacts will need to be re-assessed.

Subsequent to the verification survey, a monitoring programme needs to be established. Sampling of benthic macrofauna (using a 500- or 300-micron sieve) should be undertaken both before the start of operations, as well as at regular intervals after completion of dredging to determine the (functional) recovery rates of benthic communities. The sampling interval can best be determined after the first post-dredging sampling campaign, approximately three years after dredging of the first area. Sampling stations should include dredged and undredged reference stations in comparable environmental habitats (e.g. similar depth and sediment characteristics prior to dredging). Included in the sampling procedure should be the sampling, for at least, sediment properties (i.e. grain size analysis) as well as near-bottom dissolved oxygen concentrations and organic matter content.

# glossary of terms and abbreviations

| | |
|---|---|
| Anoxia/anoxic | Containing no oxygen. |
| Authigenic deposits | An authigenic mineral or sedimentary rock deposit is one that was generated where it is found or observed. Authigenic sedimentary minerals form during sedimentation by precipitation or recrystallization instead of being transported from elsewhere. |
| Benthic | Referring to organisms living in or on the sediments of aquatic habitats (lakes, rivers, ponds, etc.). |
| Benthic organisms | Organisms living in or on sediments of aquatic habitats. |
| Benthos | The sum total of organisms living in, or on, the sediments of aquatic habitats. |
| Biodiversity | The variety of life forms, including the plants, animals and micro-organisms, the genes they contain and the ecosystems and ecological processes of which they are a part. |
| Biomass | The living weight of a plant or animal population, usually expressed on a unit area basis. |
| Biota | The sum total of the living organisms of any designated area. |
| Bottom nepheloid layer | A layer above the ocean floor that contains significant amounts of suspended sediment. |
| Chemolithotroph | An organism that obtains its energy from the oxidation of inorganic compounds. |
| Community | An assemblage of organisms characterized by a distinctive combination of species occupying a common environment and interacting with one another. |
| Diagenesis/diagenetic | The process of chemical and physical change in deposited sediment during its conversion to rock. |
| Dissolved oxygen (DO) | Oxygen dissolved in a liquid, the solubility depending upon temperature, partial pressure and salinity, expressed in milligrams/litre or millilitres/litre. |
| Epifauna | Organisms, which live at or on the sediment surface being either attached (sessile) or capable of movement. |
| Eukaryote/eukaryotic | A single-celled or multicellular organism whose cells contain a distinct membrane-bound nucleus. |
| Habitat | The place where a population (e.g. animal, plant, micro-organism) lives and its surroundings, both living and non-living. |
| Hypoxia/hypoxic | Low oxygen conditions (<0.5 ml/ℓ dissolved O2) that are physiologically stressful to marine organisms. |
| Infauna | Animals of any size living within the sediment. They move freely through interstitial spaces between sediment particles or they build burrows or tubes. |
| Macrofauna | Benthic invertebrate animals >1 mm. |

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|---|---|
| Meiofauna | Benthic invertebrate animals <1 mm. |
| Microfauna | Animals <0.1 mm, including bacteria and protists. |
| Megafauna | Benthic invertebrate animals >10 mm. |
| Oxygen minimum zone (OMZ) | Zone in the ocean where oxygen saturation is perennially at hypoxic (<0.5ml/ℓ) conditions. |
| Pelagic | Living in the water column as opposed to benthic. |
| Photic zone | Region of the ocean through which light penetrates and where photosynthetic marine organisms live. |
| Phylum | The major taxonomic group of animals and plants; contains classes |
| Population | Population is defined as the total number of individuals of the species or taxon. |
| Prokaryote/prokaryotic | An organism whose cell lacks a true nucleus, or any other membrane-bound organelles, such as bacteria and archea. |
| Protozoa | Eukaryotic organisms belonging to a group characterized for being single-celled, most of them motile and heterotrophic such as flagellates, ciliates, sporozoans, amoebas, and foraminifers. |
| Species | A group of organisms that resemble each other to a greater degree than members of other groups and that form a reproductively isolated group that will not produce viable offspring if bred with members of another group. |
| Taxon (Taxa) | Any group of organisms considered to be sufficiently distinct from other such groups to be treated as a separate unit (e.g. species, genera, families). |
| | |

**Abbreviations**

| | |
|---|---|
| BID | Background Information Document |
| EIA | Environmental Impact Assessment |
| EPL | Exclusive Prospecting Licence |
| $H_2S$ | Hydrogen sulphide |
| I&APs | Interested and Affected Parties |
| MFMR | Ministry of Fisheries and Marine Resources, Namibia |
| ML | Mining Licence |
| MLA | Mining Licence Area |
| Mt | million tonnes |
| NMP | Namibian Marine Phosphate (Pty) Ltd |
| POM | Particulate Organic Matter |
| SP-1, -2, -3 | Sandpiper-1, 2, 3; the target areas within ML 170 |
| TSHD | Trailing Suction Hopper Dredge |

# contents

## Figures

# 1 INTRODUCTION

Namibian Marine Phosphate (Pty) Ltd (NMP) has Exclusive Prospecting Licences (EPLs) for the marine licence areas 3323, 3414 and 3415 (and four more EPLs), and exploration confirmed world-class phosphate resources in parts of the three lease areas. Subsequently, a mining licence (ML170) was awarded to NMP in July 2011, which incorporates the whole of EPL 3414 and parts of EPL's 3415 and 3323. ML170, also named Sandpiper licence area, is located on the Namibian continental shelf approximately 120 km south, south west of Walvis Bay in water depths ranging from 180 to 300 m covering a total area of 2233 km$^2$ (Figure 1).

NMP is proposing to dredge a 5.5 Mt amount of sediment on an annual basis from the uppermost 1-2.5 m (possibly to 3.0 m) of the seafloor in ML170 to recover phosphate rich material for use as fertilizer. The preferred method of sediment recovery is the use of a Trailing Suction Hopper Dredge (TSHD). Three target areas from within the mineralized resource area, have been identified in ML170, i.e. Sandpiper-1 (SP-1), -2 (SP-2), and -3 (SP-3), of which SP-1 and SP-2 are the initial dredging areas.

With respect to the proposed project, NMP is required to undertake an Environmental Impact Assessment (EIA) based on the Namibian Environmental Management Act (Act. No. 7 of 2007). During the scoping process for this EIA, the removal of sediments from the seabed has been identified as a primary impact of the proposed project (see Background Information Document (BID)). Dredging will disrupt the sediment structure and profile of the dredged area, and with it the associated benthic fauna. Potential impacts associated with this, include:

- Loss of soft-bottom habitat;
- Effects on the associated marine benthic fauna;
- Impairment of food chain functionality; and
- Creation of new habitat colonised by as yet unknown fauna.

To assess the significance of these impacts, NMP requires a Marine Benthic Specialist Study, and has appointed Steffani Marine Environmental Consultant for this task. This Specialist Study evaluates the impacts associated with dredging in SP-1, SP-2, and SP-3 only and with the technical specifications listed in the detailed project description of the EIA. If at a later stage mining is planned to be undertaken outside these areas or other technologies are considered, the findings and conclusions of this study will have to be re-assessed.

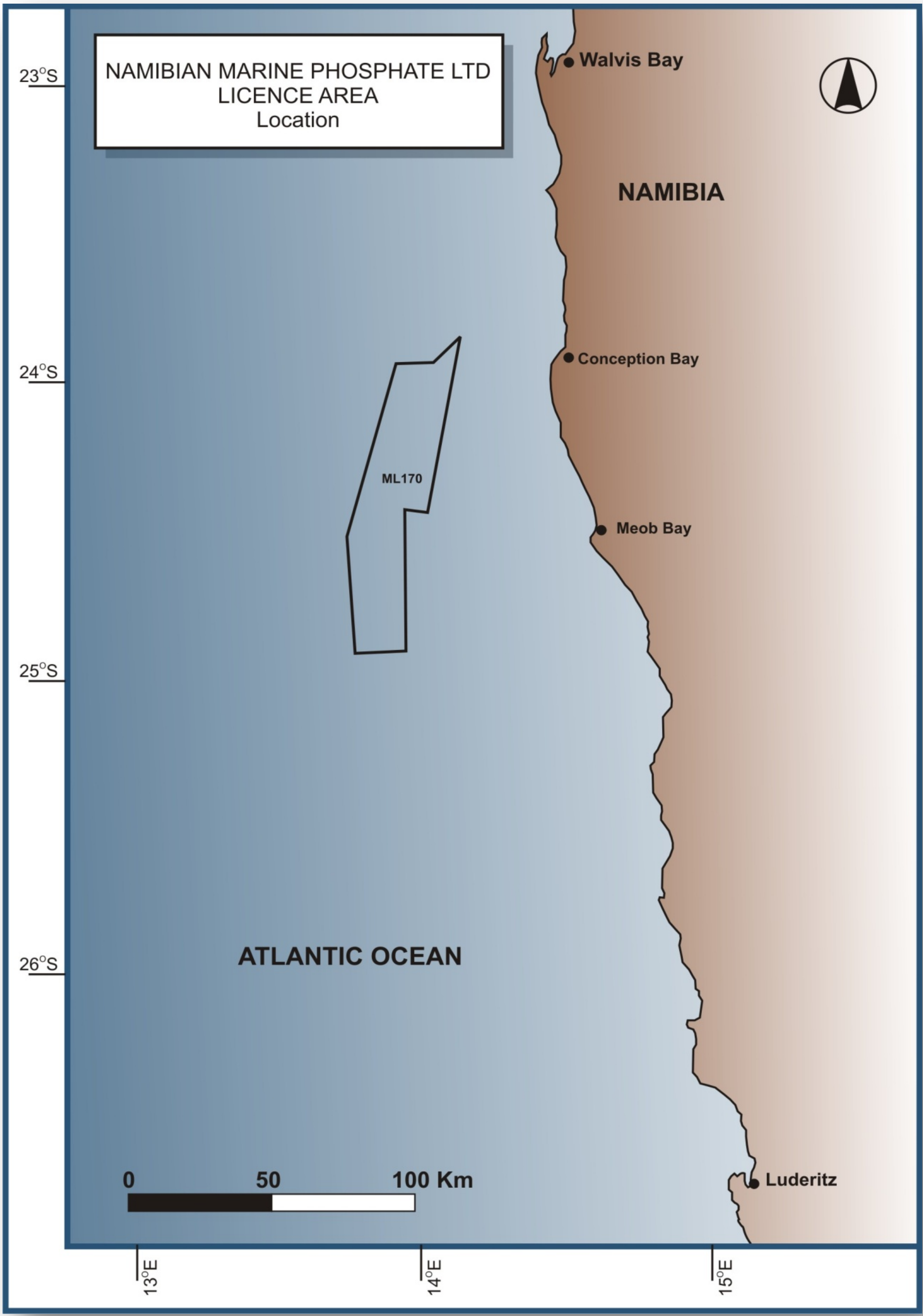


***Figure 1: Location of the Sandpiper Phosphate Licence Area off the coast of central Namibia.***

## 1.1 TERMS OF REFERENCE

The Terms of Reference (ToR) for the Marine Benthic Specialist Study are to provide:

- An introduction with a brief project overview, study approach, methodology, and assumptions and limitations;
- A description of the marine environment of the project area, focusing on the benthic invertebrate communities with special reference to the Oxygen Minimum Zone, utilising the information gathered during the macrofaunal baseline survey (it is understood that the physical environment, the water column and associated fauna, as well as fish and fisheries, mammals, and coastal birds are covered by other Specialist Studies);
- A description of the potential impacts of the project on the benthic invertebrate fauna, followed by an assessment of the significance of these impacts. The assessment will take into account the spatial scale, intensity, duration, etc. of the impacts, presented in table format; and
- Recommendations for mitigation measures and monitoring requirements, which may include the development of a macrofauna and/or meiofauna monitoring program.

Within the ML170 Licence Area, there are three general target areas, Sandpiper-1 (SP-1), -2 (SP-2) and -3 (SP-3). Mining is only proposed for these target areas and the EIA and consequently this Benthic Specialist Study is thus concerned only with the proposed phosphate mining by dredging in these three target areas for a period of 20 years, as well as the collection of exploration and environmental samples (core samples and grab samples) in the larger ML170. Any mining outside of these target areas, the introduction of other mining technology, and/or an extension to the life of mine are not covered in this Specialist Study and would need to be assessed in a separate study.

## 1.2 RATIONALE AND APPROACH TO THE STUDY

As determined by the terms of reference for the NMP project, this specialist report adopts a ‘desktop’ approach, and no new data will be collected for the purpose of the EIA. Relevant macrofauna and sediment character distribution data were, however, collected from target area SP-1 (18 stations) and SP-3 (1 station) during a benthic baseline survey in May 2010 (Steffani 2010a). Existing information from available published scientific literature was gathered and reviewed, as were unpublished reports from several sources. The assessment is structured on:

- A short description of the project background and design;
- A brief account of the affected benthic environment and existing impacts;
- A summary of the effects of sediment removal and sedimentation on benthic communities, and identification of potentially threatened habitats and/or species;
- Identification of the potential direct and indirect environmental impacts on benthic communities associated with the dredging in the Sandpiper Licence Area, and an assessment of their extent, duration, and intensity;
- Recommendation of mitigation measures and management actions; and
- Identification of monitoring requirements.

## 1.3 ASSUMPTIONS AND LIMITATIONS

The specialist study is limited to a 'desktop' approach and thus relies almost entirely on existing information only. Apart from the NMP macrofauna baseline study (Steffani 2010a) in the northern part of the Sandpiper Licence Area, and one single station in the southern part, no data on benthic communities or the physical environment (e.g. $H_2S$ sediment concentrations, near bottom oxygen concentrations, etc.) are available, and the extent and severity of the cumulative biological impacts, or the recovery period that may be expected, thus remain largely unknown. Furthermore, the frequency of natural perturbations along the southern Namibian coastline (e.g. low oxygen events, sulphur eruptions, 'berg' winds etc.), make it difficult to separate natural from human-induced impacts.

The Ministry of Fisheries and Marine Resources (MFMR) Namibia through the Permanent Secretary has been approached to grant permission to engage with MFMR scientists to obtain access to information and/or data (if available) that have relevance to the project location. At the time of writing of this report, such information relating to the benthic environment was, however, not available/provided, and the report therefore relies on publicly accessible data only. Scientific discussions were, however, conducted with, and limited unpublished data sourced from other international scientists that have worked or are currently working in the area.

## 1.4 LEGISLATION AND STANDARDS OF RELEVANCE

The legislation, standards, and codes that have relevance to the Sandpiper phosphate project are summarised in Chapter 2 of the EIA. These where relevant are applicable to this Benthic Specialist Study.

# 2 PROJECT DESCRIPTION

A detailed project description with all technical data tabulated is provided in the general EIA. Below is a brief summary of the proposed project.

The aim of the Sandpiper Phosphate project is to export 3 million tonnes (Mt) of 'rock phosphate' per annum, which requires the mining of 5.5 million tonnes of marine sediments. The mining licence has been issued for a period of 20 years. In order to accommodate product supplies to the market place, as well as building a stockpile of exportable phosphate material a three-year ramp up of production, is envisaged.

Within the ML170 Licence Area, there are three general target areas, one in the north (SP-1), one in the centre (SP-2) and one in the south of the licence area (SP-3) (Figure 2). SP-1 is 22 km long and 8 km wide (176 $km^2$) in water depth of 190 to 235 m (Table 1). This is the primary target site where dredging will commence. SP-2 has the same size but is in deeper water ranging from 245 to 285 m. SP-3 is 11 km long and 6 km wide (66 $km^2$), in water depth ranging from 235 to 270 m.

After consideration of several alternatives for the proposed recovery of the phosphate rich deep-water sediments, the use of a Trailing Suction Hopper Dredger (TSHD) has been identified as the preferred method. The proposed TSHD has a hopper capacity of 46,000 $m^3$ of slurry. The mining operation is a four part process: the first step is the dredging on a north (N) or south (S) heading (swell dependent), with the continual engagement of the dredge arm and draghead, recovering sediment in a ~3.0 m wide x ~0.75 m deep cut. The draghead needs to be engaged with the seafloor for up to ~16 km to fill the hopper (the actul distance is variable with vessel speed and cut depth); as the mine blocks within the target mine areas are only 4 km long, this implies up to 4 parallel cuts in a particular mine block for a 0.75 m deep cut. During dredging, excess water with some fines will be discharged through the overflow pipe at 10-15 m below the surface. Once the vessel hopper is filled, the dredger will sail to a single point mooring to pump the slurry ashore to a holding pond. It is assumed that it will take 16-17 hours to fill the hopper dredge, and on average 20 hours to sail to shore for unloading and sail back to the operational location to initiate dredging again.

Once back at the location, the vessel will continue to dredge within the particular 'cut' zone. Depending on the resource this will be 1-2.5 m (possibly up to 3 m) deep (Table 1). The operational requirement is not to cut (dredge) into the footwall clay, but to rather leave a residual thickness of marine sediments over the footwall of 0.3 m. Although the primary dredge direction is N-S (or S-N), dredging may also occur W-E (or E-W) to reach the desired extraction coverage. The residual layer will be thus of irregular thickness ("hummocked") due to cut width and depth, and cut numbers, and will constitute between 10 and 15% of the original thickness of the deposit at that location.

125,500 t of solids will be transported per campaign week to the shore for offloading, amounting to a dredged deposit of 5.5 Mt per annum. The levels of production are expected to require approximately 43 weeks of continuous dredging per year when the operation is in full production, which is expected to be in the third year.

Dredging will initially take place within the SP-1 deposits, which are accessible by the dredger using a single extension arm. Subsequent dredging in the deeper SP-2 and SP-3 will require a double extension arm to be installed on the dredger to be able to dredge in water depth of up to 275 m. The actual timing of this is dependent on the successful testing of the extension arm, and other factors such as the phosphate market, grade of the phosphate in the sediments, and other sediment properties (e.g. the volumes of shell in the upper layer).

***Table 1: Parameters describing the three target areas in the ML170 Licence Area***

| Detail | Sandpiper-1 | Sandpiper-2 | Sandpiper-3 |
|---|---|---|---|
| Approximate width (km) | 8 | 8 | 6 |
| Approximate length (km) | 22 | 22 | 11 |
| Area ($km^2$) | 176 | 176 | 66 |
| Water depth range (m) | 190 - 235 | 245 - 285 | 235 – 270 |
| Deposit thickness Avg (m) | 1.69 | 1.70 | 1.30 |
| Deposit thickness Max (m) | 2.5 | 2.25 | 1.80 |
| Deposit thickness Min (m) | 0.5 | 0.50 | 0.60 |
| Area mineable ($km^2$) | 160 | 162 | 54 |
| Average area mined/annum | 2.44 | 2.42 | 3.17 |

The scale of mining is primarily controlled by the annual mining target of 5.5 Mt of marine sediments to achieve the export target of 3 Mt rock phosphates. This is based on the average tonne of marine sediment containing 60% of minable grade phosphate. The area that needs to be mined in a year depends on the thickness of the deposit. The thickness of the deposit varies, but on average the resource thicknesses are (Table 1):

- 1.69 m for the SP-1 deposit, which at full production will result in area of 2.44 $km^2$ being mined,
- 1.7 m for the SP-2 deposit, which at full production will result in area of 2.42 $km^2$ being mined, and
- 1.3 m for the SP-3 deposit, which at full production will result in area of 3.17 $km^2$ being mined.

In a licence period of 20 years and a maximum annual dredging rate of 3 $km^2$, this amounts to a maximum total mined area of 60 $km^2$ or 2.7% of the total ML170 licence area, with a maximum dredge-depth of ~2.5 m (possibly up to 3 m) below the original seafloor. The dredged areas are expected to predominantly from SP-1 in the initial years and from SP-2 in the mid and later years, potentially supported by material from SP-3.

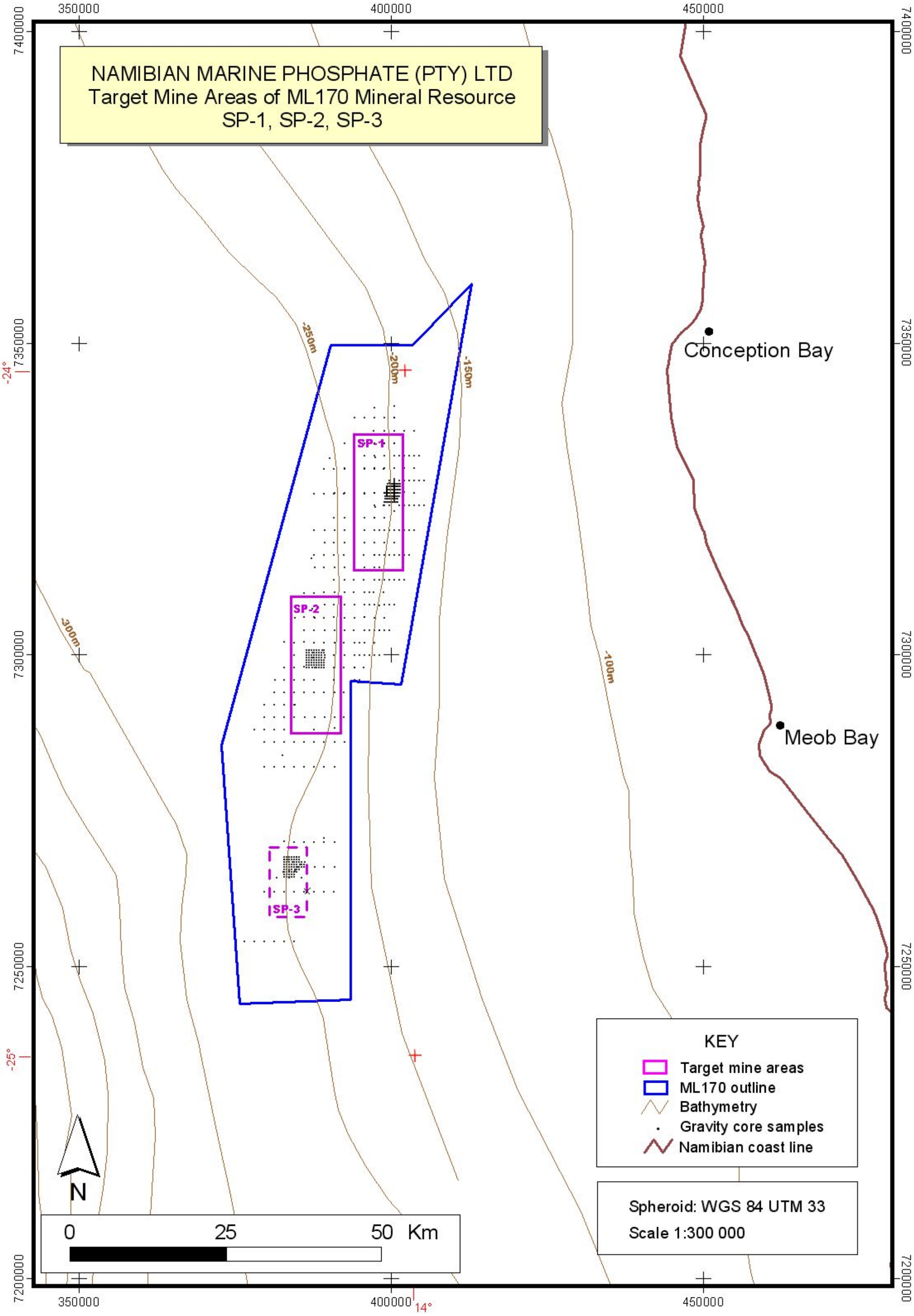


***Figure 2:*** ***The Sandpiper Phosphate Licence Area showing the three resource areas SP-1, SP-2, and SP-3.***

# 3 DESCRIPTION OF THE RECEIVING BENTHIC ENVIRONMENT

Descriptions of the regional bathymetry, geology, and oceanographic and physical processes as well as fish and pelagic communities off the central Namibian coast are presented in the EIA report and/or other specialist studies. Outlined below is a summary of the general regional setting, and a description of the benthic communities of the study area.

## 3.1 REGIONAL SETTING

*Bathymetry and Regional Sediment Properties*

The continental shelf along the west coast of southern Africa is variable in width and depth, with the edge of the continental shelf, termed shelf break, at about 200 m depth but may vary in places between about 200 and 500 m (Shannon 1985, Shannon & Nelson 1996). Double shelf breaks are common, and near Walvis Bay (23°S) there are inner and outer shelf breaks at depths varying from about 130-180 m and 300-450 m respectively. Following from the shelf break is a steep continental slope, which descends to a depth of about 5,000 m where it meets the abyssal plain of the South-east Atlantic Ocean.

The central Namibian continental shelf is covered by layers of sediments primarily of biogenic (biological) origin as a result from the high productivity in the upwelled waters (see below for upwelling). A significant feature of the central Namibian middle shelf is an extensive mud belt comprising organically rich diatomaceous oozes originating from planktonic detritus. The diatomaceous mud belt with a thickness of up to 14 m extends over 700 km in an N-S direction and 100 km in an E-W direction (Bremner 1983, Figure 3 after Currie & Emeis 2009). Depending on the local bathymetry and dynamic current intensity (Mohrholz *et al.* 2008), the landward flank of the mud belt is found at 15-104 m water depth, and the seaward flank from 45-151 m (Bremner & Willis 1993). In contrast, a second southern mud belt extends from the Orange River southwards and is mainly of river origin (Rogers & Bremner 1991).

Off central Namibia, high concentrations of particulate organic matter (POM) in the seafloor sediments are confined to three well-defined longshore belts of 500-800 km length (Bremner 1978, Rogers & Bremner 1991, Monteiro *et al.* 2005, van der Plas *et al.* 2007): the inner shelf mud belt with highest POM concentrations (7-25%) at a depth of 50-140 m, a mid-shelf belt in 200-300 m depth, which ends around 23.5°S, and an outer shelf belt in 500-1400 m depth (Figure 4). Inshore and between these belts are sediments with lower concentrations of POM, i.e. the turbulent shallow waters <50 m, an inner shelf low-POM belt at 140-200 m and an outer shelf belt at 300-500 m. Analysis of sediment pore water from a cross-shelf transect at 22°S (north of Walvis Bay) has shown a similar trend with high concentrations of ammonium and phosphate in inner shelf mud belt sediments, while beyond the inner shelf break (~150 m) concentrations decreased (van der Plas *et al.* 2007). In contrast to the inshore mud belt, which is sustained by high inner shelf phytoplankton new-production fluxes (i.e. settlement of new organic matter from the overlying water column), the deeper long shore belts of organic-rich sediments are outside the upwelling front from where most new production is exported. As indicated by the higher C:N ratios there, the organic matter is probably relict particulate organic matter originating from the inshore mud belt (van der Plas *et al.* 2007).

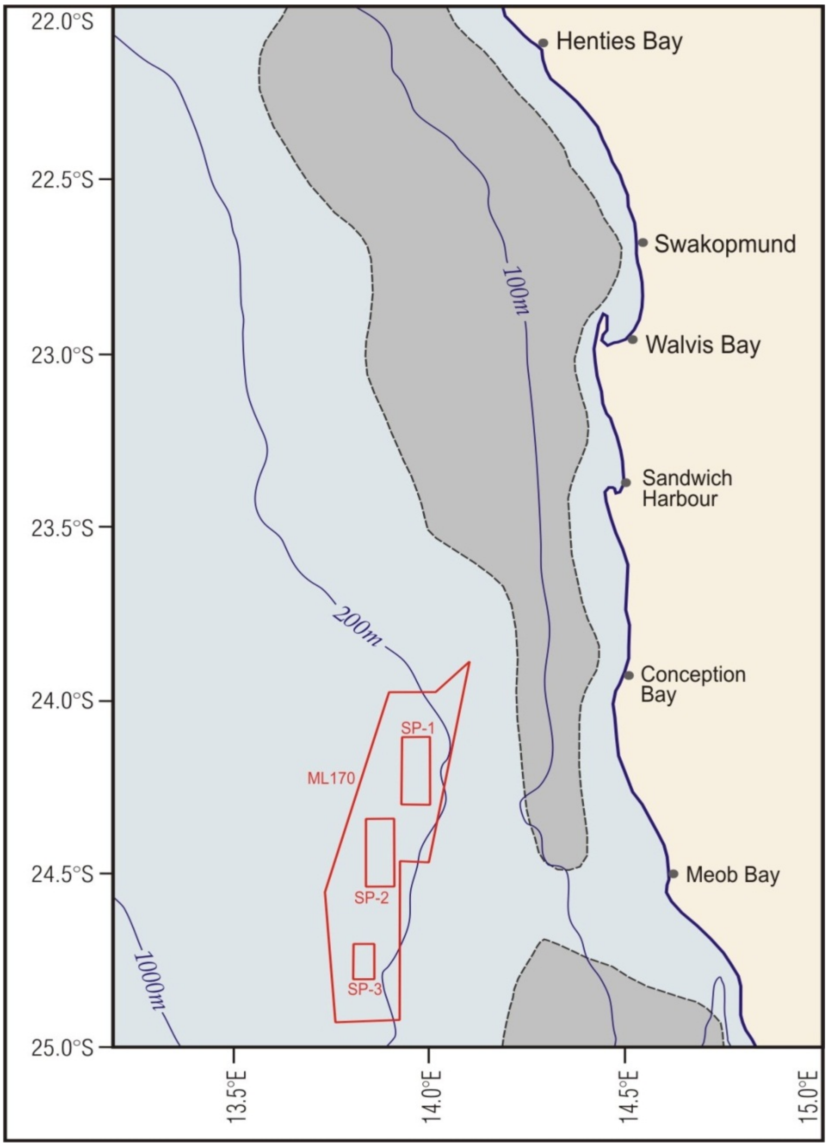


***Figure 3:*** ***The continental shelf offshore central Namibia. The grey area corresponds to the extent of the soft diatomaceous mud. The mining licence area is overlaid, showing the three target areas SP-1, SP-2, and SP-3. (Figure adapted from Currie & Emeis 2009).***

Monteiro *et al.* (2005) suggest that while Ekman transport drives the long-shore and cross-shore transport of suspended POM, the large-scale banded POM distributions in the sediments are largely governed by the interaction of internal tides with varying shelf topography (i.e. shelf break zones versus non-shelf break zones) that determines the strength of bed shear-stress and thus the potential for re-suspension of POM. This is substantiated by further research that has shown that the spatially heterogeneous organic matter accumulation and burial on the central Namibian shelf is strongly controlled by lateral transport in subsurface nepheloid layers (Inthorn *et al.* 2006, Mollenhauer *et al.* 2007). The bottom nepheloid layer or nepheloid zone is a layer of water above the ocean floor that contains significant amounts of suspended sediment. Its thickness

(100-300 m) depends on bottom current velocity and is a result of a balance between gravitational settling of particles and turbulence of the current. The particles in the layer may come from the upper ocean layers and from stripping the sediments from the ocean floor by currents. A prominent depo-centre of organic carbon is located between 24.5°S and 26°S on the upper continental slope in water depths of 400-1500 m (Figure 4, the outer shelf belt) and it is suggested that the material in this depo-centre is exported from the shelf in the nepheloid layer (Inthorn *et al.* 2006). The noticeably low organic carbon content on the outer shelf and along the shelf edge further support the hypothesis of locally prevailing erosion (Inthorn *et al.* 2006).

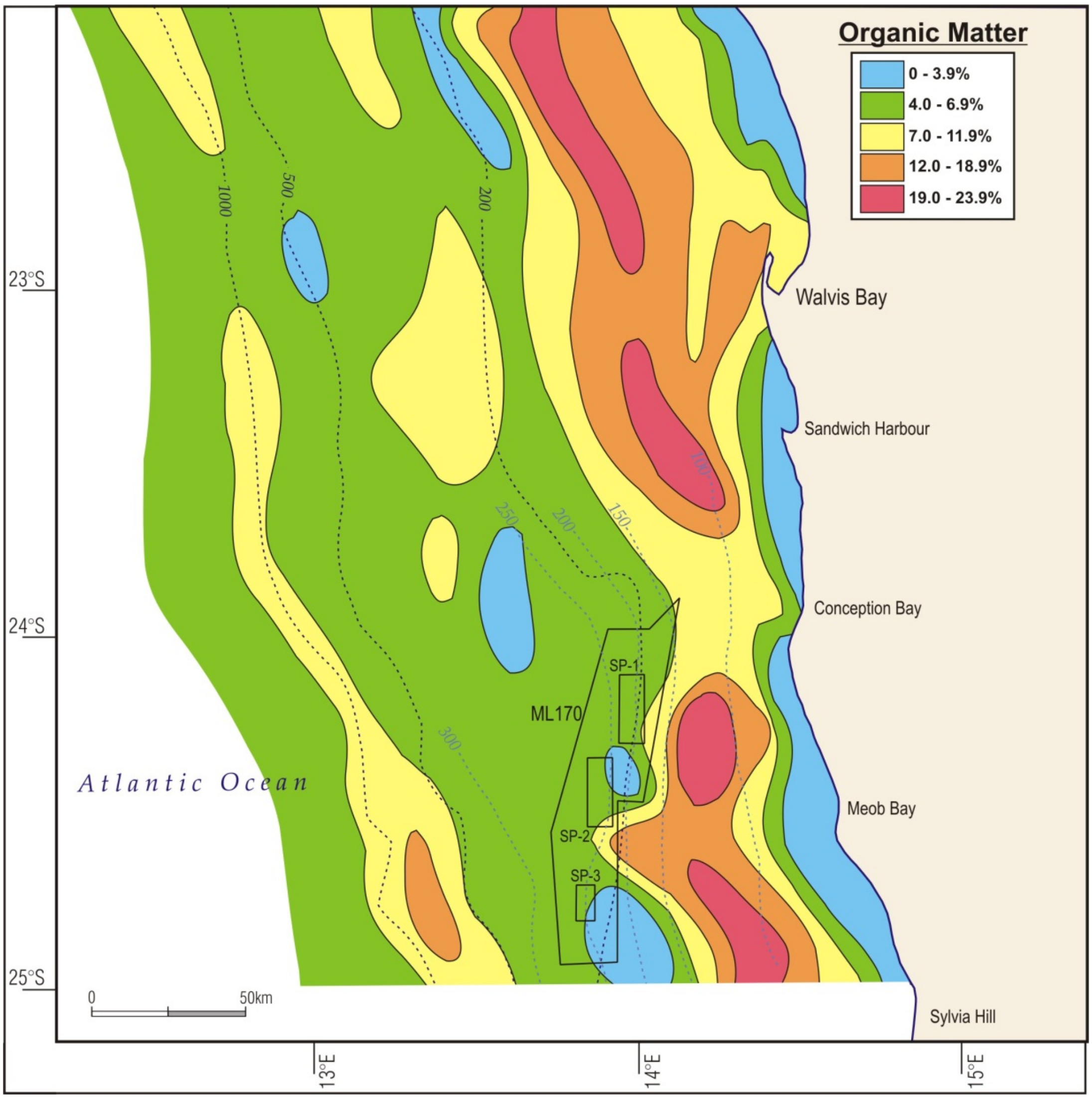


***Figure 4: Spatial distribution of the large-scale high- and low-POM bands along the Namibian shelf (adapted from Bremner 1978). The mining licence area is overlaid, showing the three target areas SP-1, SP-2, and SP-3.***

Further support is provided by acoustic profiling, which defines three main regions with different sediment patterns (Brüchert *et al.* 2006). The continental slope contains soft silts forming a smooth surface with westward prograding layers, whereas the outer shelf area (>150 m) is characterised by hardground and consolidated deposits of mainly silty/sandy sediments that suggest that strong bottom currents prevent the deposition of fine material. These sediments are apparently reworked deposits that contain only small amounts of reactive organic material. At water depths of less than 150 m, high sedimentation of organic matter forms a diatom mud layer, the diatomaceous mud belt. This agrees with the early work by Bremner (1978), who intensively surveyed the region between 17°S and 25°S collecting >550 surficial sediment samples, and found that the outer shelf region from >200 m down to around 300-400 m is characterised by sand deposits, and sandy mud and mud is confined to the inner shelf and beyond the 1000-m isobaths (Figure 5).

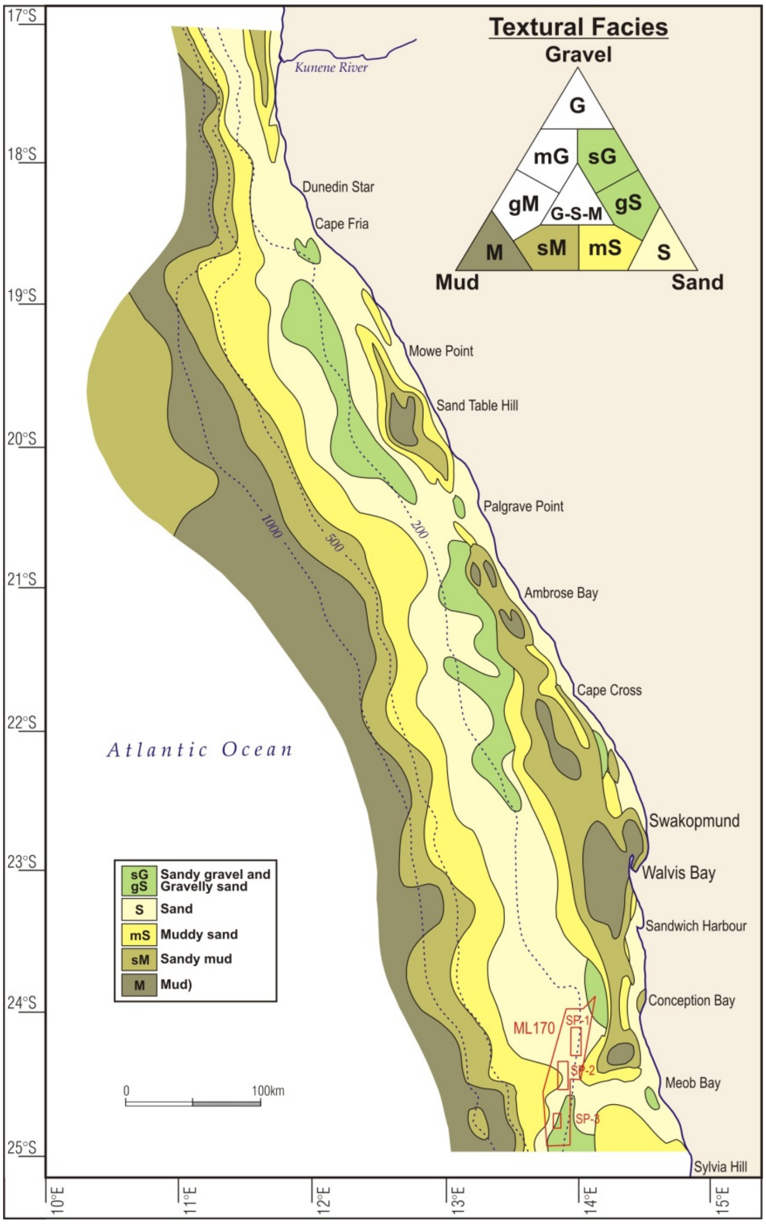


***Figure 5:*** ***Textural facies on the inner and outer shelf and the upper continental slope on the Namibian margin between 17 S and 25 S (adapted from Bremner 1978). The mining licence area is overlaid, showing the three target areas SP-1, SP-2, and SP-3.***

Within this regional context, the mining area and specifically the three target areas ranging in water depth from 190-230 m, 235-275 m and 235-270 m, respectively, are located in a generally sandy environment on the outer shelf beyond the inner shelf break, and thus offshore from the diatomaceous mud belt and south of the mid-shelf high-POM belt. Bremner (1978) recorded that this POM-rich mid-shelf belt disappears south of Sandwich Harbour (~23.5°S) where the concentration of pelletal phosphorite and mollusc shell beds reach their maximum, thus reflecting the slow rate of sedimentation in this area. According to the mapped POM concentrations from the Namibian continental margin (Figure 4), POM concentrations in the target areas are, when put into a regional context, likely to be relatively low with 4-7%, and nutrient concentrations are also likely to be low. This is probably a result of relatively strong bottom currents in this region, preventing the deposition of fine material (Monteiro *et al.* 2005, Inthorn *et al.* 2006, Brüchert *et al.* 2006). Only the southern portion of SP-2 may have slightly higher organic matter concentrations potentially exceeding 7% (Figure 4).

*Phosphate Deposits in the Target Areas*

Bremner and Rogers (1990) describe three types of phosphorite deposits from the Namibian shelf each occurring in geographically distinctly separated areas; phosphorite sand, rock phosphorite and concretionary phosphorite. The phosphorite sand is further subdivided into pelletal phosphorite and glauconotized pelletal phosphorite. Both the pelletal and glauconotized pelletal varieties are found on the outer shelf and have been dated radiometrically as pre-Quaternary, probably Miocene in age (Bremner 1978). The phosphorite sand occurs in three principal deposits centred at 19°S 12°E (off Rocky Point), 24°S 14°E (between Walvis Bay and Sylvia Hill), and 29°S 15°E (off the Orange River). The central deposit is the largest, measuring 550 km in length and 70 km in width, and is situated between 100 and 500 m depth. NMP's Sandpiper phosphate mining licence area falls into this central deposit. In contrast, the concretionary phosphorite forms today by slow authigenic growth in the interstitial waters of the diatomaceous mud belt, facilitated by sulphur bacteria (Goldhammer *et al.* 2010). Rock phosphorite deposits are minor on the Namibian shelf. Rock and concretionary phosphorite deposits are not being targeted by NMP.

The phosphate resources and sedimentological stratigraphy throughout the Sandpiper project area has been ascertained from gravity cores (with a penetration potential of up to 2.8 m) and older (Gencor) vibrocores (with a penetration of up to 6 m) (Annels 2009). Over most of the project area, the phosphorite horizon is generally subdivided into two distinct layers: an upper 0.1 to 1.0 m thick shelly phosphorite layer (Layer 1) that demonstrates a downward fining sequence and a lower 0.05 to >2.0 m thick clayey phosphorite layer (Layer 2), both identified as Miocene in age. The upper Layer 1 is characterised as a coarse broken shell bed that contains delicate off-white to brown bivalves and occasional *Turritella* supported in a very dark brown (blackish) matrix of phosphorite pellets (fine sand sized particles) and dark green organic mud. The photo in Figure 6 contrasts a shelly phosphate rich sediment sample from the target area in 205 m depth with a muddy sediment sample from 139 m water depth from the inner shelf mud belt (Source: Annels 2009). Shell fragments in Layer 1 become smaller and the phosphorite pellet component and clay increase with depth until the horizon becomes mostly a fine phosphorite sand with lesser clay. With sediment depth, this horizon passes gradationally into Layer 2 of 0.05 to >2.0 m thickness that consists of a very dark brown (blackish), soft, sticky, clayey, fine phosphorite sand, which usually becomes more clayey with depth (there are exceptions where the clay content can decrease with depth). The phosphorite content is usually highest in this part of the sequence although in some areas clay predominates. The phosphatic material within the sediment predominantly comprises unconsolidated fine sand sized phosphorite ooliths and pellets, falling in the 100 to 500 micron grain size range (mostly 150 to 250 microns). The phosphorite horizon has a sharp bioturbated contact with an underlying Miocene marine clay. This contact represents a sedimentary hiatus. This zone is a pale grey to dark olive green-grey, firm, sticky clay with coarse burrows in the top 15 cm filled with sediment from the layer above. If the phosphatic Layers 1 and 2 were being removed completely, this clay footwall is likely to be the layer exposed after dredging. However, dredging of the footwall is undesirable from a technical perspective, and since the footwall is undulating, complete removal of the above layers cannot be achieved without occasional penetration of the footwall. The project design thus proposes to avoid the complete removal of the upper layers.


***Figure 6:*** ***A phosphate rich surface sample from 205 m water depth with moderately high shell content (left) compared to a sediment sample from 139 m depth with low shell and phosphate but high organic mud content (right) (Source: Annels 2009).***

During the benthic fauna baseline survey conducted by NMP in 2010, the sediment grain size structure of the upper 20 cm of the sediments was analysed for each benthic of the 80 grab samples taken (Steffani 2010a). This analysis confirms that the surficial sediments over most of the study area (SP-1 and the areas immediately west and east of the block) is littered with molluscan shell fragments, with the sediment being generally described as olive silty sand with shell fragments, or shells and shell fragments with some sand (sediment fractions determined through standard sieving techniques conducted by the geotechnical laboratories of Geoscience, Cape Town). The sediments were poorly sorted and had a strong bimodal distribution, with the two main grain size fractions being gravel (>2 mm which are primarily the shell fragments) and fine (125-250 micron) to medium (250-500 micron) sand, whereas mud and/or silts (<63 micron) were nearly undetectable. A sedimentological study by Rogers (2008) reports on the surficial sediments from the southern part of the licence area EPL 3323 (the eastern part of the Sandpiper licence area) from depth 136-182 m. This study confirms that at depths of 150 m and deeper the sediments are highly dominated by sand whereas the mud content drops significantly to 0-20%, and it also reports the presence of significant amount of shell fragments. Bremner (1978) suggests that the mollusc layer is relict. This sediment characteristic seems to follow the POM banding as described by several studies (Bremner 1978, Monteiro *et al.* 2005 see above). For example, van der Plas *et al.* (2007) have shown for a transect at 22°S that sediments from the inner-shelf mud belt and the POM band on the continental slope at ~1000 m depth consisted predominantly of mud (>96%), whereas the POM-depleted stations between the POM-rich bands were dominated by sand (50-75%) or muddy sediments (25-50%). A station near the inner shelf break at 200 m depth had appreciable amounts of gravel in the sediments.

*Upwelling, Hypoxia, and Hydrogen Sulphide*

The coastal waters off central Namibia are part of the Benguela upwelling system, one of the major upwelling systems of the world (Shannon & Nelson 1996). The strongest upwelling cell is near Lüderitz (27°S), with several secondary upwelling cells, namely the Kunene, northern Namibia and central Namibia cells (at ~18°, 20°, 24°), and the Namaqua, Columbine and Cape Peninsula cells in South Africa. During upwelling, cold nutrient-rich bottom water is transported to the photic zone, resulting in a high primary production, which in turn fuels a rich secondary production in the upper pelagic and nearshore zones.

In common with other eastern boundary upwelling systems (Helly & Levin 2004), the northern and central Benguela regions are characterised by the occurrence of natural shelf hypoxia, which is referred to as oxygen minimum zone (OMZ) (Monteiro *et al.* 2011). OMZs have dissolved oxygen concentrations of ≤0.5ml/ℓ and typically impinge upon the continental margins of upwelling regions at depths of 100-1000 m (Helly & Levin 2004). Off Namibia, this layer extends between at least 18°S and 28°S and up to 60 km from the shore. The hypoxic conditions show seasonal variation, locally shifting to anoxic conditions in late summer-autumn (Monteiro *et al.* 2008). Brüchert *et al.* (2006) provide areal estimation of anoxic and hypoxic bottom waters on the central Namibian shelf, respectively. In accordance with the two shelf breaks they divided the shelf into a total shelf and an inner shelf area defined as the areas inside the 300 m and 100 m isobaths, respectively. Anoxic bottom water conditions occur in an area covering 8,944 km$^2$ while hypoxic conditions can spread over 46,954 km$^2$, 55% of the total shelf area. Hypoxic conditions are a typical feature of highly productive systems as they are rich in decaying organic material, leading to high biological and chemical oxygen demands in the water column and the underlying organic-rich sediments (Brüchert *et al.* 2006). The local oxygen consumption during remineralisation of sinking detritus, however, cannot fully maintain the observed low oxygen concentrations on the northern Benguela shelf. It appears that a combination of local oxygen consumption and the relative dominance of oxygen poor South Atlantic Central Water, which flows southward from the Angolan Dome, over oxygen rich Eastern South Atlantic Central Water contribute to the observed oxygen status (Monteiro *et al.* 2006, 2008, 2011, Mohrholz *et al.* 2008, Ohde & Mohrholz 2011), and that ventilation rates are related to shelf width (Monteiro *et al.* 2011).

Intense anaerobic decomposition of organic matter in the sediment by sulphate reducing bacteria leads further to the formation of toxic hydrogen sulphide ($H_2S$) with the highest rates of bacterial sulphate reduction occurring right below the sediment-water interface (Brüchert *et al.* 2003). Ultimately, even sulphate availability is limited in these sediments, and methanogenesis starts a few centimetres below the sediment surface, which leads to the accumulation of free methane gas (Emeis *et al.* 2004). Nearshore hydrogen sulphide eruptions are a well-known phenomenon off the Namibian coast, particularly near Walvis Bay, that can affect areas of ocean surface of more than 20,000 km$^2$ (Weeks *et al.* 2004, Ohde & Mohrholz 2011) often with devastating effects on the fish and invertebrate fauna. Hydrogen sulphide accumulation in the water column is mainly driven by the diffusive flux of hydrogen sulphide from the sediment and transport with methane bubbles (Brüchert *et al.* 2006, 2009). Large areas of the organic-rich sediments of the inner shelf mud belt also host free gas just decimetres to metres below the sediment-water interface. The gas is a mixture of methane and $H_2S$, whereby a portion of the $H_2S$ is produced by the reduction of sulphate with methane in the sulphate-methane transition zone. Emeis *et al.* (2004) reported that eruptions of these biogenic gas accumulations are a further significant contributor to inshore hydrogen sulphide outbreaks. Sediment craters and severely disrupted sediments at the seafloor give evidence to these eruptive degassing events. Potential triggers for eruptions include changes in the atmospheric and oceanographic pressure fields. A key factor controlling the extent of the gas-charged zones is the thickness of the diatomaceous mud whereby sediments of >12 m thickness may be completely gas-charged. According to Brüchert *et al.* (2006), gas-charged sediments cover an area of approximately 1,350 $km^2$ and are confined to the landward side of the diatomaceous mud belt (Figure 7, from Brüchert *et al.* 2006).

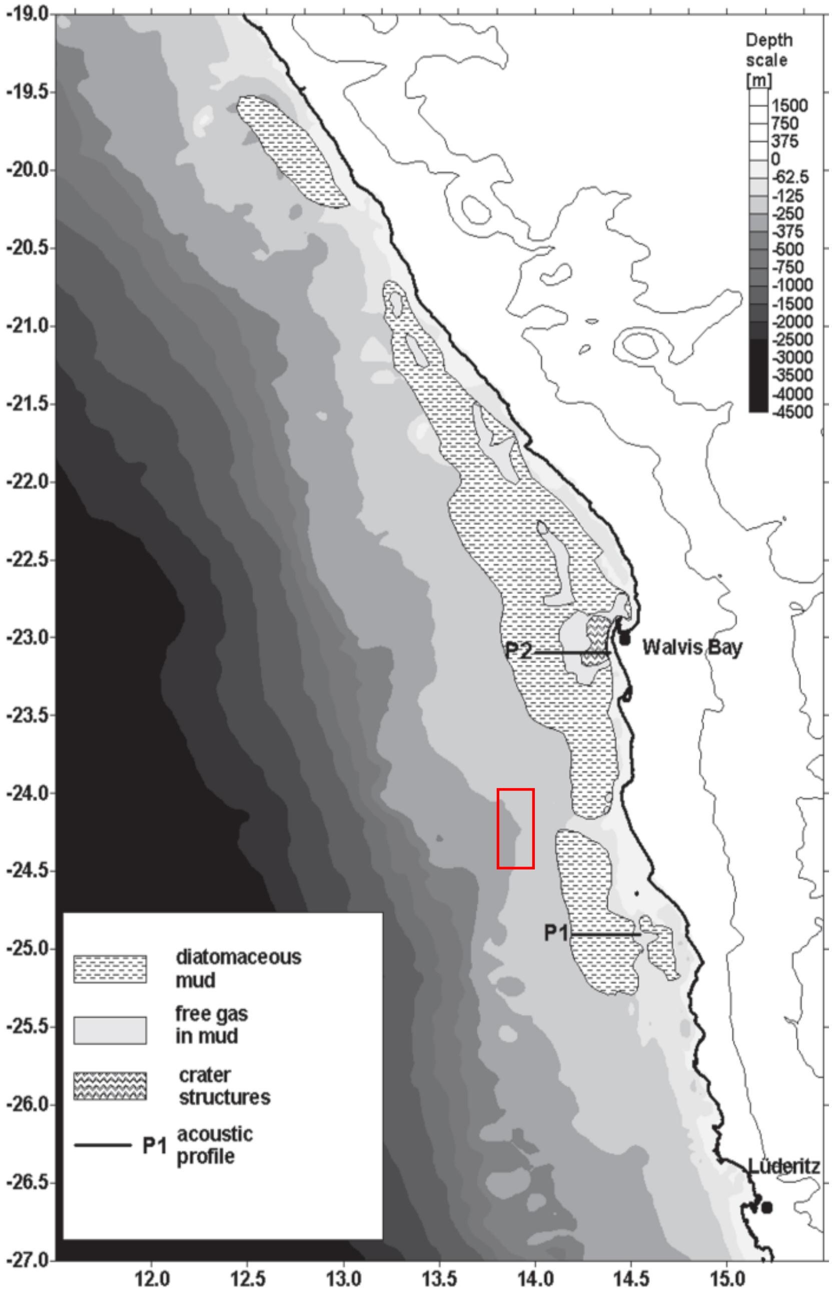


***Figure 7: Distribution map of diatomaceous mud belt, the gas-charged area and areas with abundant crater structure and disrupted seafloor. The lines depict acoustic profiling transects (Map according to Brüchert et al. 2006). The red rectangle indicates the approximate location of the Sandpiper licence area (not to scale).***

Brüchert *et al.* (2003) investigated the regulation of bacterial sulphate reduction and hydrogen sulphide fluxes along two transects ranging from 25-2000 m depth; one off Walvis Bay (23°S) and one offshore of south of Sylvia Hill at 25.30°S. They reported that sulphate reduction rates are high in mud belt sediments but that across the inner shelf break, sulphate reduction rates drop drastically from 150 to 200 m water depth. They postulated that this is linked to re-suspension of fines in bottom currents leaving the remaining particulate organic matter increasingly altered and more refractory. Some of the re-suspended material is subsequently also exported to the continental slope through the bottom nepheloid layer (Inthorn *et al.* 2006). Further research has confirmed that for central Namibian shelf areas with depths >150 m, sulphate reduction rates and pore water concentrations of dissolved sulphide are low, and fluxes of hydrogen sulphide across the sediment-water interface were found to be below detection (Brüchert *et al*. 2006). Similar results were reported from a cross-shelf transect at 22°S, which showed that the $H_2S$ concentration was high in the pore waters of the inshore mud-belt sediments (up to 190 µmol in 4-8 cm sediment depth) but low (<5 µmol) beyond the shelf break (van der Plas *et al.* 2007). In general, significant sulphide fluxes to the water column correspond to the distribution of the diatomaceous mud belt, which has a seaward limit at the inner shelf break (Brüchert *et al*. 2006). With increasing abundance of biogenic carbonate sand in the sediment, the fluxes of sulphide decrease (Brüchert *et al.* 2003). Sediment cores taken near 23°S from 100 to 300 m water depth confirm that sulphide concentrations are high at depth <150 m but decrease drastically at 200-300 m depths, at least in the upper 10-40 cm of sediment (Figure 8, V. Brüchert, RV *Poseidon* cruise 250/2, May 1999, unpublished data). $H_2S$ concentrations in deeper sediment layers were, however, not measured and potentially could be higher.

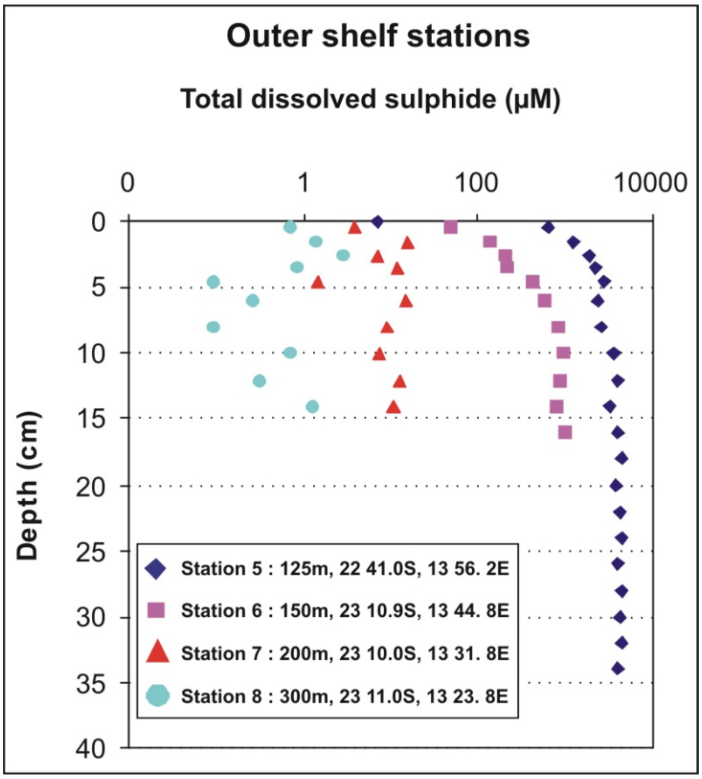


***Figure 8:*** ***Total dissolved sulphide concentrations in the upper 40 cm sediment layer in 125, 150, 200 and 300 m water depth near 23°S (V. Brüchert, RV Poseidon cruise 250/2, May 1999, unpublished data).***

Benthic sulphide-oxidising bacteria such as the large bacteria *Thiomargarita* and *Beggiatoa*, which cover large areas of the shelf, particularly the mud belt, and other chemolithotrophs play an important role in modulating the flux of dissolved sulphide to the water column (Brüchert *et al.* 2003, 2006). A recent study in Namibian waters has observed that large inner shelf areas covered by sulphidic water were detoxified by blooming chemolithotrophs that oxidized the biologically harmful sulphide to environmentally harmless colloidal sulphur and sulphate (Lavik *et al.* 2009). The detoxification proceeded by chemolithotrophic oxidation of sulphide with nitrate and was mainly catalysed by two discrete populations of Gamma- and Epsilon proteobacteria. These chemolithotrophic bacteria, which can account for 20% of the bacterioplankton in sulphidic waters, created a buffer zone between the toxic sulphidic subsurface waters and the oxic surface waters. The study suggests that sulphide can be completely consumed by bacteria in the subsurface waters and may easily be overlooked by monitoring of surface waters. Consequently, sulphidic bottom waters on continental shelves may be more common than previously believed.

Over the last few years, a substantial number of studies have been published concerning the central Namibian OMZ (e.g. Brüchert *et al.* 2000, 2002, 2003, 2006, 2009, Dale *et al.* 2009, Emeis *et al.* 2004, Monteiro & van der Plas 2006, Monteiro *et al.* 2005, 2006, 2008, 2011, Bartholomae & van der Plas 2007, van der Plas *et al.* 2007, Lavik *et al.* 2009, Julies *et al.* 2010, among others). These publications, however, focussed primarily on the centre of the OMZ off Walvis Bay (e.g. MFMR has a monthly monitoring transect off Walvis Bay at 23°S) and/or on the mud belt, which is in shallower waters than the project area. The average bottom water concentrations of sulphide and oxygen as observed during 2004-2005 off the central Namibian coast (Lavik *et al.* 2009) are summarised in Figure 9.

From these maps, and substantiated by other publications (e.g. Brüchert *et al.* 2006), it is apparent that hydrogen sulphide in bottom water is largely confined to inshore of the 200 m isobaths. Low oxygen bottom water (~0.5ml/ℓ) also occurs primarily in waters <200 m but between 24°S and 25°S, bottom hypoxia can extend into deeper waters. Sulphide and oxygen bottom water concentrations are, however, temporally and spatially very variable.

Data from the mining area itself do not exist (or were not available at the time of writing of this report), but from the information in the literature it can be surmised that the licence area lies at the southern offshore fringe of the OMZ, with perennial low dissolved oxygen levels (<0.5 ml/ℓ) at the bottom but not anoxic conditions (Lavik *et al.* 2009, see also Figure 5 in Bartholomae & van der Plas 2007). Hydrogen sulphide pore water concentrations, $H_2S$ fluxes from the sediments and low $H_2S$ bottom water concentrations are, however, likely to be very low (<5 µmol $m^{-2}$ $d^{-1}$, Brüchert *et al.* 2003, 2006, van der Plas *et al.* 2007), because the hydrogen sulphide forms slowly and most of it is oxidised within the sediment. While there can be little doubt that the project area is affected by low dissolved oxygen concentrations in the bottom waters, it seems that conditions of severe hypoxia, anoxia, biogenic gas accumulations, high $H_2S$ concentrations (both in sediment and near-bottom waters) and frequent $H_2S$ eruptions are more prevalent in the inner-shelf mud belt and not in the deeper, further offshore located phosphate target areas. Nevertheless, all long sediment cores that have been retrieved from the inner and outer Namibian shelf contained some hydrogen sulphide, whereby concentrations typically increased with sediment depth (V. Brüchert, pers. comm.). It can thus be expected that hydrogen sulphide concentrations in sediments buried more than 50 cm are higher and may exceed 1 mmol/ℓ.

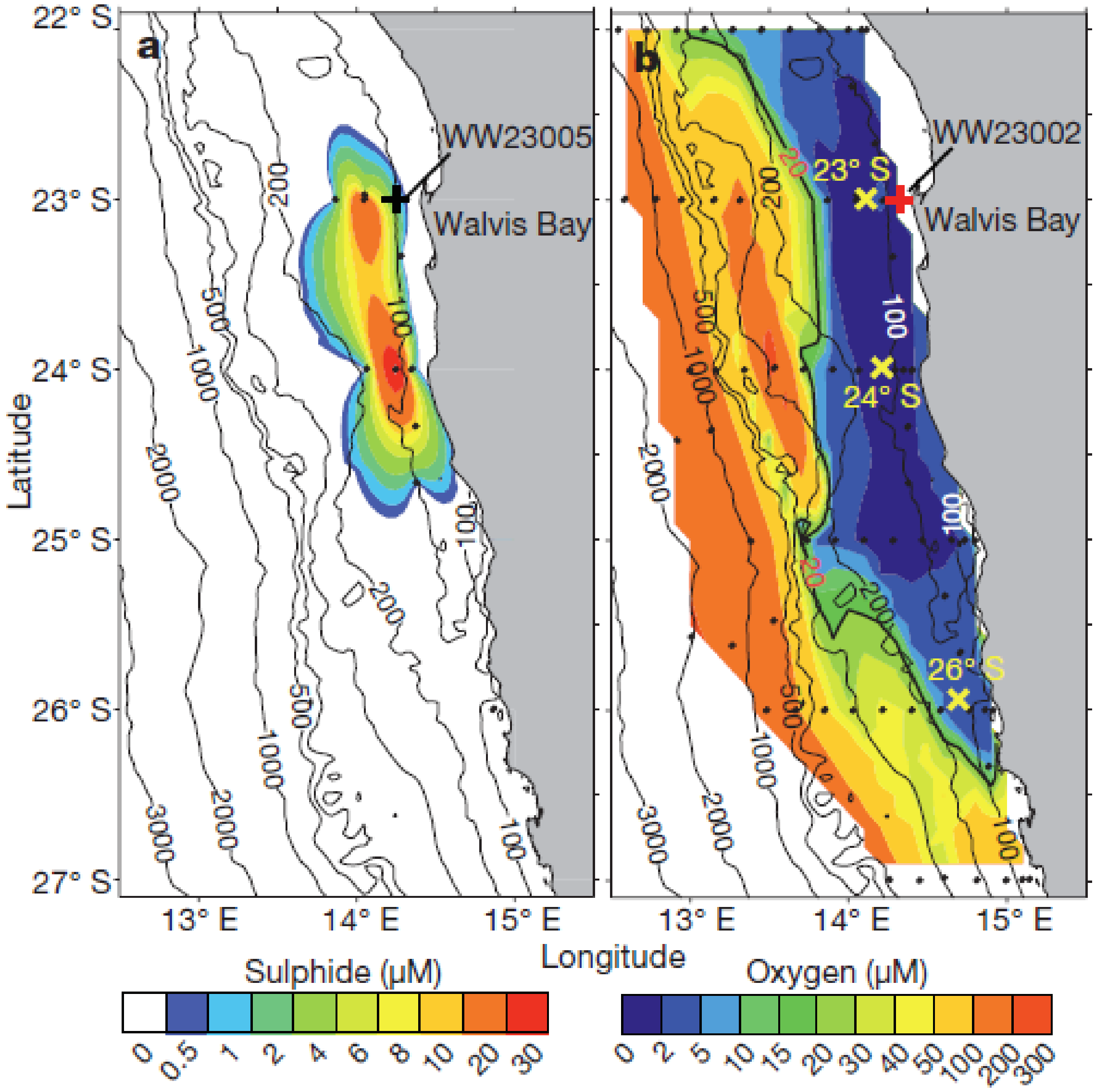


***Figure 9:*** ***Bottom-water concentrations (~2m above the sediment) of sulphide (a) and oxygen (b) during 2004-2005 (Figure extracted from Lavik et al. 2009). The crosses in (b) and WW positions indicate sampling locations investigated by Lavik et al. 2009. The red numbering on the thick contour in (b) indicates the 20 µM oxygen concentration.***

## 3.2 BENTHIC COMMUNITIES

Marine sediments are home to many benthic organisms living on (epifauna) or in the superficial sediments of the sea floor, with the greatest abundance to a depth of ~20 cm (infauna). The fauna is typically divided by size into megafauna (>10 cm), macrofauna (large enough to be retained on 1 mm sieve, while some researchers also use a 500-µm sieve), meiofauna (0.1-1 mm) and microfauna (<0.1 mm). Megafauna include large crustaceans, molluscs, and echinoderms, and are often associated with the surface of the sea floor. The macrofauna usually constitute the dominant biomass or organisms within marine sediments and typical taxa include polychaete annelids, smaller crustaceans (e.g. amphipods, isopods, shrimps, crabs), molluscs (gastropods and bivalves) besides many other phyla. The meiofauna is dominated by the large and diverse groups of nematods and harpacticoid copepods, while microfauna include bacteria and protists. Macrofauna and other benthic fauna are a major food source for fish and other benthic predators, and play important roles in ecosystem processes such as nutrient cycling, pollutant metabolism, and dispersion and burial of organic matter (Sanders 1968, Snelgrove 1998, Gray 2002).

As established in the previous section, the project area lies at the fringe of the central Namibian OMZ and is thus affected by variable dissolved oxygen conditions with bottom water oxygen concentrations probably below 0.5ml/ℓ. Despite oxygen depletion, protozoan and metazoan assemblages can thrive in OMZs (Levin 2003). Many organisms have adapted to low oxygen conditions by developing highly efficient ways to extract oxygen from oxygen-depleted water. Adaptations include small, thin bodies, enhanced respiratory surface area, blood pigments such as haemoglobin, biogenic structure formation for stability in soupy sediments, an increased number of pyruvate oxidoreductase enzymes, and the presence of sulphide-oxidising symbionts (Rosenberg *et al.* 1991, Diaz and Rosenberg 1995, Lamont and Gage 2000, Gray *et al.* 2002, Wu 2002, Levin 2003).

Within OMZs, foraminiferans, meiofauna, and macrofauna typically exhibit high dominance and relatively low species richness (Levin 2003). Macrofauna and megafauna often have depressed densities and low diversity in the OMZ core, where the oxygen concentration is lowest, but they can form dense aggregations at the OMZ edges (Levin 2003, Levin *et al.* 2009). This so-called 'edge effect' has been observed for both macrofaunal and megafaunal invertebrates, typically at oxygen concentrations between 0.15 and 0.40 ml/ℓ. Current understanding predicts the existence of oxygen thresholds, below which most taxa are excluded through physiological intolerance to hypoxia, and above which selected taxa are able to take advantage of an abundant food supply. The availability of both oxygen and organic carbon seem to determine the richness of macrofaunal species in OMZs until the oxygen content rises to about 0.45 ml/ℓ; above that level, oxygen is much less important (Levin and Gage 1998). Notably, meiofauna do not appear to exhibit these responses and often persist at high densities throughout OMZs (Levin 2003). Taxa most tolerant of severe oxygen depletion (~0.2 ml/ℓ hypoxia) include calcareous foraminiferans, nematodes, and polychaetes. Agglutinated protozoans, harpacticoid copepods, and calcified invertebrates are typically less tolerant. It has been hypothesized that small-bodied animals, with greater surface area for $O_2$ adsorption, should be more prevalent than large-bodied taxa under conditions of permanent hypoxia (Levin 2003). However, body sizes were found not to be smaller within the lower OMZs of the Oman (Levin *et al.* 2000) and Pakistan margins (Levin *et al.* 2009), and it was suggested that the abundant food supply in the lower or edge OMZs promotes larger macrofaunal body size.

Very little is known about the Namibian OMZ's benthic fauna (see Arntz *et al.* 2006, Zettler *et al.* 2009). In May 2010, a macrofauna baseline survey was conducted by NMP in SP-1 as this is the initial priority area for proposed mining operations (Steffani 2010a). The survey design consisted of three transects, 5 to 10 km apart, with six sampling sites per transect at approximate depths of 175, 185, 200, 215, 230 and 240 m. The target mining area lies between the 190 and 235 isobaths, so that between three and four of the sampling stations per transect fall into the mining area while the shallower and deeper sites are outside the area. Overall species richness of the benthic macrofauna assemblages was relatively low and strongly dominated by polychaetes (64% of species), followed by crustaceans, molluscs and a number of species belonging to a variety of other phyla (e.g. Cnidaria, Oligochaeta, Echinodermata, Ascidiacea, Nemertea). By far the most abundant species, and present in every sample, was the spionid polychaete *Paraprionospio pinnata* (44% of overall abundance) followed by a number of other polychaetes and two mollusc species. Crustaceans, on the other hand, were both in terms of abundance and biomass very poorly represented. Most species found in the study area have a larger geographical distribution and/or have been recorded elsewhere from the Namibian and/or South African west coast (e.g. Savage *et al.* 2001, Steffani and Pulfrich 2004, 2007, Steffani 2007, 2009, 2010b, c, Zettler *et al.* 2009). There are almost no data available on Namibian's OMZ, but the benthic community composition in terms of species diversity, phyla dominance, as well as the presence of certain low-oxygen indicator species (e.g. *P. pinnata*) is generally in good agreement with studies from other OMZs around the world (e.g. Gutiérrez *et al.* 2000, Levin *et al.* 2000, Gallardo *et al.* 2004, Lim *et al.* 2006, Gooday *et al.* 2009, Levin *et al.* 2009). *Paraprionospio pinnata*, for example, is found all over the world in low-oxygen conditions, usually being the dominant species (Gutiérrez *et al.* 2000, Gallardo *et al.* 2004, Lim *et al.* 2006, Zettler *et al.* 2009), and has been classified as resistant to severe hypoxia (Diaz and Rosenberg 1995). Its tolerance to hypoxic conditions is apparently related to it possessing four pyruvate oxidoreductase enzymes, which provide a greater metabolic capacity to cope with functional and environmental hypoxia (González and Quiñones 2000). Other polychaete genera that are repeatedly recorded from OMZs include *Sigambra* sp., *Lumbrineris* sp., and members of the family Cirratulidae (Gutiérrez *et al.* 2000, Levin *et al.* 2000, Gallardo *et al.* 2004, Lim *et al.* 2006), all of which were common in the study area (Steffani 2010a).

Molluscs are often found to be sparse in OMZs, but contributed a relatively significant proportion to the fauna in SP-1. A similar finding was presented by Zettler *et al.* (2009), who studied the macrofauna community in the OMZ off northern Namibia (offshore the Kunene River mouth, which is at the northern fringe of the OMZ). In agreement with other OMZs, they reported a far lower species diversity in the hypoxic zone compared to oxygenated nearshore areas, but recorded that the community was dominated by two mollusc species (contributing >70% numerically). They suggested that this community is probably more representative of the fringes of the upwelling cells of the northern Benguela (Kunene River to Walvis Bay) than for the centre with severe anoxia and high hydrogen sulphide concentrations. The SP-1 area lies at the southern fringe of the OMZ and a similar scenario is likely to apply. In an early study by Sanders (1969) of the benthos in the Namibian OMZ, a reduction in macrofauna species diversity has been observed in the core of the OMZ, whereas higher abundances and biomasses have been recorded from the edge of the OMZ, which is in agreement with findings from other OMZs. While there is at least some limited information on OMZ macrofauna, no data are available on meiofauna distribution and community composition, whether area specific or regional (at least to the author's knowledge).

For shallow, organic-enriched sediments that experience seasonal or episodic hypoxia, it is hypothesized that hypoxia rather than organic matter availability drives the majority of benthic faunal responses (Gray *et al.* 2002). In contrast, Levin *et al.* (2009) postulate that within OMZs, where animals have evolved to cope with permanent low oxygen, organic matter supply and quality may play a more important role in structuring benthic communities than hypoxia. Organic matter content in the SP-1 sediments was not measured during the baseline survey but regional data suggest that the area lies within a re-suspension zone with relatively low POM concentrations (see Figure 4). The significance of organic matter over low oxygen concentrations is thus likely to be an important factor for structuring the benthic communities in ML170. It is therefore important to include organic matter measurements into future monitoring programmes, as well as measurements of bottom water dissolved oxygen and $H_2S$ concentrations.

Toxic sulphides might also contribute to low diversity in OMZs. As discussed earlier, the central Namibian shelf is locally subject to hydrogen sulphide in the sediments and the bottom waters. While $H_2S$ is highly toxic to most benthic animals, large sulphur-oxidising bacteria live in the sediments or build extensive mats on the surface-water interface in such regions. The most common in Namibian waters is the giant bacterium *Thiomargarita namibiensis*, which is currently the largest known bacterium up to 0.75 mm in diameter (Schulz *et al.* 1999). It occurs in the upper few centimetres of sediments together with other sulphide-oxidising bacteria such as *Beggiatoa.* However, only *Thiomargarita* appears to tolerate environments with high concentrations of hydrogen sulphide (Schulz *et al.* 1999). These bacteria oxidise sulphide to sulphur with either oxygen or nitrate to obtain energy (i.e. chemosynthesis), whereby both sulphur and nitrate are stored internally (Schulz *et al.* 1999, Schulz 2006). *Thiomargarita* has been estimated to account for as much as 55% of the total sulphide oxidation of the central Namibian shelf (Brüchert *et al.* 2003), which emphasizes its important ecological role. The high reflectivity of the stored sulphur microgranules gives the bacteria and the surface of the sediment a white appearance, which makes them easily detectable by sea bottom photography (Brüchert *et al.* 2009). Zones of nitrate and hydrogen sulphide do not overlap in marine sediments, but even though *Thiomargarita* species are non-motile, they can populate sediments in which their electron acceptor (nitrate) and energy source (hydrogen sulphide) do not coexist. They probably rely on external transport events, such as periodic re-suspension of loose sediments possibly caused by fluid and gas eruptions to contact the oxygenated water column, or on temporal variation in the chemical environment. Recent investigations have shown that *T. namibiensis* appears to have a life mode that is unusual for marine bacteria (Schulz and Schulz 2005). Under anoxic conditions, it takes up sulphide and, presumably, acetate, which appears to be stored as glycogen. Because there is an insufficient supply of a suitable external electron acceptor, internally stored nitrate and polyphosphate are sacrificed and sulphide is oxidized to elemental sulphur to generate energy. Under oxic conditions, the bacterium can generate energy by the oxidation of both sulphur and, presumably, glycogen. At the same time, it invests energy in the accumulation of polyphosphate and nitrate, the latter of which is stored in a central vacuole at concentrations of up to 800 millimoles. Thus *T. namibiensis* is able to take up each of these chemical compounds under conditions where the chemicals are readily available and use them under different redox conditions when they are a valuable energy source that would otherwise be impossible to obtain at that time. *T. namibiensis* is also thought to be the driving force for the formation of new phosphorite in marine sediments of the diatomaceous mud belt by episodically releasing phosphate into the anoxic sediment thereby concentrating pore water phosphate sufficiently to drive spontaneous precipitation of phosphate-containing minerals, such as hydroxyapatite (Schulz and Schulz 2005, Goldhammer *et al.* 2010).

Studies suggest that chemosynthesis-based nutrition plays an important role in OMZ systems, either through symbiosis or through heterotrophic consumption of chemosynthetically fixed carbon (reviewed by Levin 2003). Numerous OMZ heterotrophs consume free-living chemosynthetic bacteria or prey on those species that do, which provides a link to higher trophic levels. For example, the giant bacteria as well as benthic infauna are implicated in the diet of the bearded goby *Sufflogobius bibarbatus* (Utne-Palm *et al.* 2010). The goby's distribution ranges from southern Angola to southern South Africa, with peaks in abundance reflecting the distribution of the diatomaceous mud belt off central Namibia (Staby and Krakstad 2006). Recent research has provided insight into the goby's remarkable physiology, behaviour and dietary plasticity, which allows it to thrive under even the most severe conditions (Utne-Palm *et al.* 2010). It is able to tolerate extreme hypoxia and high concentrations of hydrogen sulphide and capitalises on this advantage by staying on the hypoxic seafloor during the day in order to avoid predators such as horse mackerel and hakes. At the seafloor, it consumes diatom-containing mud and benthic infauna. At night, individuals migrate into the water column, where they associate with, and feed on jellyfish, re-oxygenate their blood and digest the food before returning to the seafloor at dawn (Utne-Palm *et al.* 2010). Dietary studies have shown that the goby is very opportunistic in its diet being able to utilise a large variety of food sources, and that changes in the diet are linked to ontogenetic changes in the habitat occupied: large individuals are primarily demersal where they feed on benthic infauna but move up into the water column at night, whereas smaller individuals are more pelagic and feed primarily on zooplankton (Cedras *et al.* 2011, Hundt *et al.* 2011). A combination of analytical methods has further revealed that jellyfish can contribute >70% to the goby's diet while diatom- and bacteria-rich sediments from the mud belt may contribute 15% to the diet, and that small gobies feed more on zooplankton while large gobies consume more sedimented diatoms (van der Bank *et al.* 2011). The migratory behaviour makes the goby available to a wide variety of predators, including pelagic seabirds, seals and a variety of fish. Indeed, since the collapse of the pelagic fishery off Namibia during the 1970s, the bearded goby has replaced the sardine *Sardinops sagax* in the diets of many of the higher trophic levels within the system and it is now playing a key role within the regional food webs (Cury and Shannon 2004). Despite the high level of predation pressure, the regional biomass of the bearded goby is increasing (Staby and Krakstad 2006). Its success within the altered ecosystem off Namibia is likely to be a result of its physiological adaptions to hypoxic conditions as well as its ability to utilise the increasing jellyfish biomass and the bacteria-rich sediments for nourishment (both were up to now considered dead end resources) (van der Bank *et al.* 2011). A recent study addressed cross-shelf differences between an inshore station at 120 m water depth in the mud belt and a deeper station at 180 m, and found clear differences in diet and migratory behaviour. In nearshore environments, characterised by a deep layer of anoxic bottom water overlying thick diatomaceous sediments, *S. bibarbatus* of all size classes were found to feed primarily on the mud and associated sulphur bacteria with digestion at night near the surface (van der Bank *et al.* 2011). By contrast, at the offshore station the low oxygen layer (~0.4 ml/ℓ) was by far thinner and fish in these environments did not contain diatoms (or bacteria) in their stomachs but had fed on other food items. The diet reflected what is available in the habitat occupied: larger, more demersal gobies appeared to feed more on benthic infauna, whereas smaller, more pelagic individuals fed on zooplankton, and digestion appeared to take place throughout the 24-hour day. It was hypothesized that at the more oxygen-rich offshore station, the quality of the detritus is lower (which would imply less or no diatom and/or bacterial mats) and the benthic fauna richer (van der Bank *et al.* 2011).

During NMP's benthic baseline survey, bacteria (microfauna) were not sampled, and it is thus unknown whether these large sulphur-oxidising bacteria occur in the project area. Lavik *et al.* (2009, supplementary information) state that *Thiomargarita* and *Beggiatoa* are only present in considerable abundances north of 22°S and do not appear to thrive in the area south of 23°S. Brüchert *et al.* (2006), however, provide regional distribution maps of *Thiomargarita* and

*Beggiatoa*, showing that the two bacteria occur south of 22°S but are concentrated in different areas (Figure 10). *Beggiatoa* is abundant in the area north of Palgrave Point and is found north to Cape Frio, whereas *Thiomargarita* is most common in the area south of Palgrave Point and around Walvis Bay. *Thiomargarita* and *Beggiatoa* are also abundant between 24°S and 25.30°S near Sylvia Hill and between Conception Bay and Walvis Bay, but are largely confined to the mud belt in depths <200 m. In depths <200 m, regional differences in the distribution of these bacteria affect the development of sulphidic bottom waters. While hydrogen sulphide is quantitatively oxidised in sediments covered by *Beggiatoa* mats, only a fraction of the sulphide is removed by *Thiomargarita* (Brüchert *et al.* 2006). In an earlier study, Brüchert *et al.* (2003) also report that the sediments between the inner shelf break (>150 m depth) and the upper slope contained no large sulphur bacteria although bacterial sulphate reduction was detected. It was suggested that the sulphide fluxes at these depths are too low to support the populations of the large sulphur bacteria, and that chemical sulphide oxidation or oxidation by sulphide-oxidising bacteria other than the large sulphur bacteria may dominate on the outer shelf and continental slope. Furthermore, the distribution of *Thiomargarita* on the central Namibian shelf was found to be linked to the presence of free gas in the sediments (Vogt 2002, cited in Emeis *et al.* 2004), and the accumulation of free gas in turn is restricted to the diatomaceous mud belt, specifically to areas of substantial sediment thickness (>8m) in the centre of the mud belt (Emeis *et al.* 2004, Brüchert *et al.* 2006, see also Figure 7). On the other hand, recent ROV (Remotely Operated Vehicle) footage off Rocky Point at 19°S (>550 km to the north of the licence area) clearly shows small patches of white bacterial mats in ~200 m water depth (Figure 11, footage made available by Rolf Koppelmann, Institute for Hydrobiology and Fishery at Hamburg University, GENUS project).

However, this particular area and depth falls within the high POM mid-shelf belt that extends from ~18.5°S to just south of 23°S (Figure 4), which due to its high organic matter content may have sulphide concentrations high enough to support sulphur bacteria. The mining area, however, is located far south of this belt in a region of lower organic matter concentrations.

Although at present it cannot be substantiated by empirical evidence, it is thus highly likely that sulphur-bacteria abundance is very limited in the target mining areas at depths of 190-275 m (V. Brüchert, Department of Geological Sciences, Stockholm University, Sweden, pers. comm.). The probable reason for this is that the sediments in this area are primarily reworked sediments with sulphate reduction rates and resulting hydrogen sulphide concentrations that are too low to support large abundances of sulphur bacteria (V. Brüchert, pers. comm.). During the exploration for the Sandpiper project, over 1500 geological grab or core samples (to a depth of around 2 m) were collected from within ML170 as well as the surrounding area over a five-year period by the same group of geologists. The core samples in particular provide an undisturbed picture of the sediments from the seawater-sediment interface to deeper sections. The geologists reported that they had not observed any bacterial-mat components, and that $H_2S$ smells had only been noticed occasionally, and this from the shallower (<170 m) waters of EPL 3323 (B. Ludick, Operations Geologist, NMP, signed letter from 1 November 2011). This is obviously only an anecdotal account and not verified through analytical measurements, but while it is very possible that bacterial mats can be overlooked (in a 6.5 cm diameter core or from grab samples, especially if they are not the subject of investigation), the rotten egg smell typical of $H_2S$ is very hard to miss. The fact that a distinct $H_2S$ smell was noted from some benthic samples (Steffani 2010a) and sediment cores is also not conclusive as humans are very sensitive to hydrogen sulphide gas and can smell it at concentrations as little as <1 µmol; concentrations apparently too low to sustain sulphur-oxidising bacteria (Brüchert *et al.* 2003). In addition, while sulphur bacteria are associated with present-day phosphate precipitation (Goldhammer *et al.* 2010), the presence of phosphorites in turn does not imply the present-day occurrence of sulphur bacteria, as the pelletal phosphorite sands in the licence area represent older reworked phosphorite deposits, meaning their formation is not recent but occurred probably during the Miocene age since, as a consequence of lower sea levels, the region was an intertidal mudflat in a subtropical estuarine environment (Bremner and Rogers 1990). The concretionary phosphorite deposits, on the other hand, which are forming today and whose formation is being facilitated by the sulphur-oxidising bacteria, is confined to the mud belt sediments (Goldhammer *et al.* 2010), being richest along the landward edge of the diatomaceous mud belt (Bremner 1980, Bremner and Rogers 1990). These concretionary deposits are not being targeted by the Sandpiper Phosphate Project.

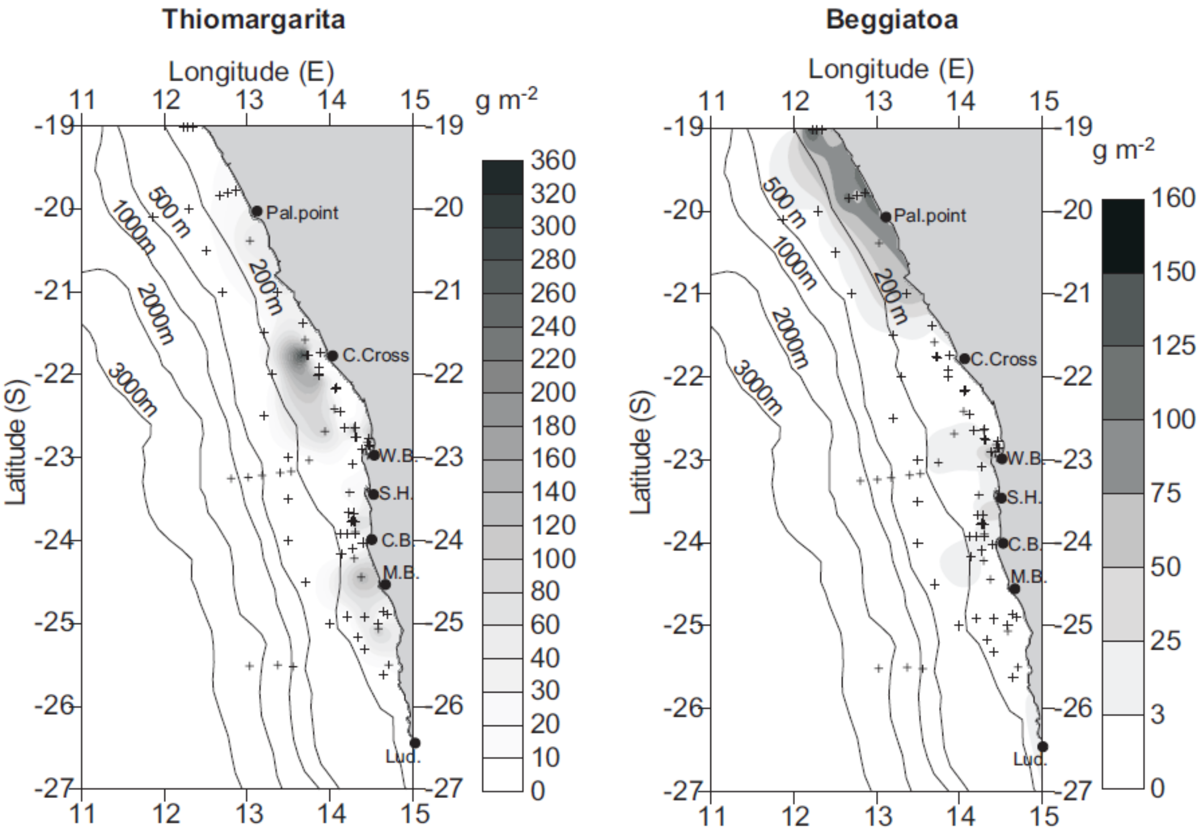


***Figure 10:** Spatial distribution map of (a) Thiomargarita and (b) Beggiatoa (Source: Brüchert et al. 2006, crosses indicate sampling stations in their study).*


***Figure 11:*** ***ROV photo of white bacterial mats in 207 m water depth at 19°S (Rolf Koppelmann, Hamburg University, Institute for Hydrobiology and Fishery, (GENUS), unpubl. results).***

## 3.3 EXISTING IMPACTS IN THE STUDY AREA

Hydrocarbon exploration wells in the region have been drilled in roughly the same water depth but to the immediate north and south of the mining licence area. Information on other marine mining (e.g. diamond mining) or petroleum exploration drilling operations planned in the ML170 licence area is not available.

The demersal trawling industry operates offshore of the 200 m isobath. The trawling grounds thus overlap with the mining licence area and ML170 will therefore be potentially subjected to trawling disturbance (as well as having been impacted historically by trawling). A detailed account of the fishing industry, fishing areas and effort is given in the Fishery Specialist Study (Appendix 1a). In summary, trawling for hake is done with bottom-trawl (otter) gear. Hake-directed fishing effort is centred in 300-500 m water depth with proportionately lower hake trawling effort in shallower (<300 m) waters. The mining licence area overlaps with this fishery only on the fringes of their effort distribution profile with moderate intensity. The monkfish trawl-fishery operates largely at the same depths as the hake sector but with more effort concentrated in the shallower depth range (<300 m). It is estimated that about 50% of the ML170 area has been and is likely to be subjected to monk-directed trawl operations, and trawling intensity is expected to be relatively high (compared with other trawled areas). This primarily affects SP-2 and SP-3 while almost no trawling occurs in the shallower SP-1 area. The monk fishery deploys similar bottom-trawl gear to the hake fishery with some modification such as the use of tickler chains, which increases the potential impact of the contact with the trawling gear on the substrate. Monkfish vessels generally trawl for 24 hours with shorter trawls during the day and longer trawls at night. Other fisheries such as the midwater trawl-gear for horse mackerel and purse seine gear for small pelagic species use gear that will have little or no contact with the seafloor and are thus not of direct consequence for the benthos in the ML170. Only one other fishery may impact on the benthos in the MLA - the demersal bottom set longline fishery for hake is expected in the MLA area. The impact of this fishery, however, is expected to be low as the gear set is static and comprises lines and hooks set on or close to the substrate. There will thus be contact with the substrate by this fishing gear, but the nature of this contact is expected to be such that impact on the substrate is very low.

Fishing activities result not only in the removal of target and non-target by-catch species, but also cause varying levels of disturbance to the environment. Over the past decade, it has become widely acknowledged that bottom trawling activities have a profound disturbance effect on the ecosystem (e.g. Dayton *et al.* 1995, Watling and Norse 1998, Rosenberg *et al.* 2003, Thrush *et al.* 2006, Tillin *et al.* 2006, Kaiser *et al.* 2006, Shephard *et al.* 2010, among many others) whereby towed bottom-fishing gears are described as one of the largest global anthropogenic sources of disturbance to the seabed and its biota (Kaiser *et al.* 2006). The effects of trawling on benthic assemblages has been the focus of many international studies too numerous to summarise here, also since this is not the topic of this report. From the wealth of studies, however, it is clear that trawling can alter habitat complexity by homogenising soft-bottom sediments, that it can remove, damage or kill biota thereby reducing overall benthic production, and that it can lead to substantial changes in benthic community composition and habitat (see for example Dayton *et al.* 1995, Collie *et al.* 2000, Kaiser *et al.* 2006, Thrush *et al.* 2006 for reviews of studies). The type of physical impact the fishing gear has on the seafloor and the biota, depends largely on the mass and design of the gear, degree of contact with the seafloor and the towing speed (Thrush and Dayton 2002), while the severity of the impact varies with the type of habitat affected and may differ for different types of benthic assemblages (e.g. in- or epifauna) (Kaiser *et al.* 2006). The greatest measurable direct impacts are usually detected in epifaunal communities, i.e. the biota that live on or protrude from the sediments, and are thus most vulnerable to the passage of trawl gear (Jennings and Kaiser 1998). A significant different between bottom trawling and dredge mining is the repeated impact of bottom trawling, which permits limited time for recovery between trawls, and the abandonment of mined-out areas after 2 - 3 passes of the dredge head, which allows recovery to begin without further disturbance.

Although the demersal trawl fishing industry is the largest, most valuable fishery in southern Africa, up to now there has been only one, very recent study addressing trawling impacts on the benthic communities in the Benguela upwelling region (the study included one sampling area in Namibia) (Atkinson 2009, Atkinson *et al.* 2011). The study focussed on differences between lightly and heavily trawled areas, as control (untrawled) areas in similar habitats are not available. The results suggest that differences in trawling intensity are at least partially responsible for significant variation in benthic assemblage composition between heavily and lightly trawled areas. The study found that epifaunal abundances, number of species and species diversity decreased with increasing trawling intensity, and that there were also considerable changes in epifaunal assemblages in more heavily trawled sites. Impacts on infaunal species richness and abundances were, however, less evident.

# 4 SOURCES OF RISKS TO THE BENTHIC MARINE ENVIRONMENT

The identification of potential environmental issues associated with the phosphate mining operations set within the framework of the NMP project is based on a review of the proposed activities and the nature of the affected environment. It incorporates all issues raised during the Public Participation Process as summarised in the Scoping Report.

The most immediate effect of dredging is the loss of benthic organisms by the removal of the substratum, but a typical by-product of dredging activities is the re-suspension of sediments into the water column and the eventual re-deposition of this material. More specifically, aspects of the dredging activities that need to be considered include:

- The loss of benthic communities through removal of sediment during the dredging process;
- The effects of sediment removal on (re-)colonisation and recovery rates of impacted communities;
- Change in sediment characteristics as a result of dredging;
- The potential indirect effects of the loss of benthic communities on demersal fish in the area (will be specifically addressed in the Fishery Specialist Study Appendix 1a);
- The effects of re-deposition of suspended material; and
- Release of nutrients by dredging and its direct/indirect effect on benthic communities, and release of hydrogen sulfide from sediments during dredging (will be more specifically addressed in the Water Column Specialist Study Appendix 1b).

This specialist report focuses on the benthic soft-bottom environment and therefore only discusses impacts relating to this habitat. Sources of risks to the pelagic environment, the water quality (e.g. turbid plumes, re-suspension of contaminants), and fish communities and associated fisheries are described in detail in other specialist studies (this EIA).

## 4.1 REMOVAL OF SEDIMENT AND POTENTIAL CHANGE IN SEDIMENT CHARACTERISTICS DUE TO DREDGING

*Sediment Removal and Colonisation*

Typical dredging activities in the ocean include dredging for navigational purposes (capital dredging to deepen certain areas or maintenance dredging to maintain a required depth for navigation), extraction of sediments as construction material (aggregate extractions) and dredging works needed for construction projects. These dredging activities are the best studied (e.g. Newell *et al.* 1998), but differ in operational water depth (typically in shallower waters <50 m) and often dredging depths (in general an aggregate-extraction trench is 20 - 25 cm deep) from those proposed for the Sandpiper Phosphate Project. Other marine mining operations include experimental deep-water mining for manganese nodules, exploration for sulphide systems (hydrothermal vents), and diamond mining in shelf waters. As yet, dredging for marine phosphate rock has not been conducted but on 22 October 2010, a ten-year offshore mining licence for an area 180 km north of Lüderitz (ML159) was granted to Samicor's sister company LL Namibia Phosphates (www.chamberofmines.org.na). LL Namibia Phosphate will continue to develop the marine phosphate project, but since this is a large and capital intensive project, it will take a number of years to bring into production. A number of other Exclusive Prospecting Licences (EPL) for marine phosphate mining have been granted and exploration work has been conducted in some EPLs, but no further applications have been submitted yet (www.chamberofmines.org.na). Although the proposed Sandpiper Phosphate Project is thus the first of its kind, marine phosphate deposits, due to the recent substantial increases in the market value have become attractive in other parts of the world. For example, the New Zealand company Chatham Rock Phosphate Limited has been granted an offshore prospecting permit for Chatham Rise, located 450 km east of Christchurch, that covers an area of 4,726 km$^2$ at a depth of approximately 400 m with an estimated reserve area of 380 km$^2$ of significant seabed deposits of rock phosphate (www.rockphosphate.co.nz). The advanced technology needed for dredging in such depths is being provided by the Dutch company Royal Boskalis Westminster. Environmental studies for this project are underway but not available yet (webpage accessed 30 November 2011).

Marine mining operations are not new to Namibia. Marine diamond exploration in shallow waters off the west coast of southern Africa began as early as the 1960s. Significant quantities of diamonds in deeper waters, however, have only been mined since the 1990s, particularly in an area off the south west coast of Namibia in the mining licence area known as Atlantic 1, an area of approximately 6,000 km$^2$ (Penney *et al.* 2007). The mining vessels used for recovery of diamond-bearing gravel are fully self-contained mining units, with a processing facility on board, and are potentially able to operate 24-hours a day for at least 11 months a year. Mining at depths beyond 75 m is conducted using either vertically mounted, large-diameter rotating drill-heads (Wirth drills), or seabed crawler systems, which allow a horizontal approach along a 'lane', and are capable of mining sediment thicknesses of up to 5 m, in water depths up to 150 m. The mined sediment is pumped on board and discharged onto a series of sorting screens. The oversize boulders and undersize fine tailings are immediately discarded overboard, and the intermediate fraction is fed into a dense medium separation plant for extraction of the diamonds. The recovery tailings after diamond extraction are also discarded overboard and thus ~99% of the pumped material is finally returned directly to the sea (Penney *et al.* 2007). Recently, the use of trailing suction dredge systems have been evaluated by diamond mining companies as this would allow the removal of thicker overburden. In 2006, trial dredging was undertaken near the terrigenous mud belt in Atlantic 1 MLA at 105-130 m depth whereby all of the dredged sediment was transported to shore (Penney *et al.* 2007), a similar operation thus to the one proposed for the Sandpiper Phosphate Project.

Since the majority of dredging operations are for navigational purposes or for aggregate extraction, it is not surprising that most of the studies concerning the impact of dredging on benthic fauna have focused on this type of dredging operations. This research has shown that typical dredging operations can be expected to result in a 25-70% reduction of species diversity, 45-95% reduction in abundance, and a similar reduction in biomass (see Newell *et al.* 1998 and Herrmann *et al.* 1999 for references). Dredging for aggregate, however, leaves behind trenches of around 20-25 cm depth only. In the present case, the upper sediment layer of an average of 1.7 m, and up to a maximum of 2.5 m (possibly up to 3 m) will be removed. Soft-bottom benthic species live on or in the superficial layers of the sediment, with >90% of the animals in the first 20-30 cm. Evidently, the removal of the upper 1-2.5 m (possibly up to 3 m) of substratum can be expected to result in the total destruction of this benthic biota. Similar total-destruction results are assumed for marine diamond mining operations (Penney *et al.* 2007).

Colonisation (recovery) of the newly exposed surface usually starts rapidly after cessation of dredging (Hall 1994, Newell *et al.* 1998, Herrmann *et al.* 1999, Ellis 2003). Colonisation can take place by passive translocation of animals during storms or sediment sliding from nearby unaffected areas, and the active immigration of mobile species, whereby undisturbed deposits left behind between dredged furrows can provide an important source of colonising species (van Moorsel 1993, 1994, Cheshire and Miller 1999). The main pathway of colonisation though is through settlement and recruitment processes from the plankton (Hall 1994), which are highly variable functions both in space and time, and cannot be predicted with any certainty (Woodin 1986, Butman 1987).

The rate of colonisation depends largely on the extent to which the original community is naturally adapted to sediment disturbances, on physical factors such as depth, exposure (waves, currents), and sediment character, and on the biochemical environment (Newell *et al.* 1998, Herrmann *et al.* 1999). Opportunistic species, characterised by being small, mobile, highly-reproductive and fast growing, are the early colonisers and may already attain increased densities within months after sediment removal. Long-lived species, however, such as molluscs and echinoderms need longer to re-establish the natural age and size structure of the population, and biomass therefore often remains reduced for several years (e.g. Kenny and Rees 1994, 1996, Kenny et al. 2008, van Dalfsen *et al.* 2000, Desprez 2000, Newell *et al.* 2004, Boyd *et al.* 2004, 2005, Hussin *et al.* 2012).

A key factor governing the speed and nature of colonisation is the quantity of the original sediments remaining in the dredged area and the rate of refill with sediments (Newell *et al.* 1998). Aggregate extraction is conducted in shallow locations within the wave base where sediment movement is usually greater than in deeper areas and therefore recovery can be expected to be faster. However, even in 20-30 m water depth, dredged depressions have been observed to persist for 3 to >7 years depending on tidal currents and wave exposure (Boyd *et al.* 2004, Birchenough *et al.* 2010). As a result, recovery rates for benthic communities disturbed by shallow aggregate dredging vary greatly from <1 year to >11 years and are highly site-specific (e.g. Kenny and Rees 1994, 1996, van Dalfsen *et al.* 2000, Newell *et al.* 1998, 2004, Boyd *et al.* 2004, 2005, Robinson *et al.* 2005, Birchenough *et al.* 2010, Borja *et al.* 2010, Hussin *et al.* 2012, among many others). Sediment refill and recovery rates in deeper waters are likely to be far slower. Results from environmental monitoring surveys conducted by De Beers Marine Namibia as part of their Environmental Programme to assess the impacts of mining on soft-bottom communities, suggest that recovery after mining with drill ships or crawlers may take as long as 10 years for areas below the wave base (Parkins and Field 1998, Pulfrich and Penney 1999, Savage *et al.* 2001). Diamond mining operations differ from dredging operations in that most of the recovered sediment is discharged back to sea. Settling rates of the discarded material depend on particle size, and larger particles often settle back into the mined-out area. Monitoring surveys in a South African deep water mining licence area have demonstrated refill rates in mined-out areas of >80 cm over an 8-months period, which was linked to intensive nearby mining (Steffani 2009). While this facilitated an early start of colonisation, benthic communities were still different 2.5 years after cessation of mining, and sediment composition was vastly different from its original state (Steffani 2010b). In contrast, the TSHD trial dredging that was undertaken near the mud belt in Atlantic 1 MLA in 105 m depth is in terms of operational procedure comparable to the Sandpiper project as all of the dredged sediment was transported to shore. The dredged area was 200 x 500 m in size with an estimated average dredged depth of 5.28 m (Steffani 2010c). Monitoring surveys revealed that three years after mining the re-colonisation by macrofauna was relatively advanced, although abundance, biomass, species number, and species diversity were still lower than before the dredging and also when compared with a nearby unmined control site (Steffani 2010c). Differences in community structure were related primarily to a shift in the relative abundance of common species, and not due to a change in species composition. Interpretation of the data in terms of natural temporal variability was, however, compromised due to limited sampling of the control site. Although recovery was not fully accomplished three years after dredging, the recovery process was much faster than has been anticipated for such depths, and in view of the depth of the dredged area (>5 m). It is suggested that this is linked to the natural high sedimentation rate in the region (the area is near the terrigenous mud belt of the Orange River mouth) that facilitated rapid refill of the mined-out area. Local species in turn, are likely to be adapted to continuous and/or periodical sediment deposition. Macrofaunal communities typical of such environmentally stressed habitats are more resilient than those of more environmentally stable habitats (Newell *et al.* 1998), and species typical of naturally stressed assemblages possess life-history traits that allow rapid re-colonisation after disturbances leading to shorter recovery times (Bolam and Ress 2003, Whomersley *et al.* 2010). It is evident that recovery rates after physical disturbance events are extremely site-specific and depend on local hydrographical, physical and biochemical conditions, and the inherent ecological plasticity exhibited by many benthic species.

*Changes in Hydrographical Conditions and Sediment Characteristics*
An important factor for the recovery of the benthos is whether the new substratum available for colonisation differs from that prior to the disturbance. This can be due to a number of reasons, for example when dredging exposes different seabed sediments from those prior to the activity, when dredging/mining alters the remaining surficial sediments through selective screening or discharging of certain particle classes, or when the mined-out area changes local hydrographical conditions leading to changed particle settlement/re-suspension conditions. For example, diamond mining operations off the Orange Delta in 130 m depth were shown to significantly increase the gravel and coarse sand components in the surficial sediments, as well as to increase the spatial heterogeneity of the sediment composition (Rogers and Li 2002). Furthermore, the natural stratification of the deeper layers of sediment, formed during long periods of time as a result of sea level changes, is irrevocably destroyed by mining, and will not be recreated until the next cycle of sea level change. Long-term or permanent changes in grain size characteristics of sediments will affect other factors such as organic content, pore-water chemistry, and microbial abundance and composition (Snelgrove and Butman 1994). Together these features manifest as changes in the physico-chemistry of the substratum. The total removal of the phosphate deposits in the Sandpiper target areas would expose the footwall, which consists of firm clay, compared to the sandy/shelly deposits of the original surface layer. Hard consolidated clay is less than ideal for small burrowing fauna, and the clay may function more like a hard bottom substrate. This would lead to the area being colonised by a very different suite of animals. Dredging of the clay footwall, however, is undesirable from a technical perspective and it is proposed to leave behind a residual layer of phosphate sand to avoid direct contact of the draghead with the footwall. Mitigation recommendations will also include leaving behind a minimum layer of approximately 30 cm of sediments. Nonetheless, exposed residual sediments will be different from the original surficial layers as evidenced by the change in sediment composition with depth in the geological core samples (Annels 2009). New sediment in the dredged out area can accumulate through bedload transport of mobile sand, by natural deposition of fines from the water column, through slumping from the undredged pit walls (and potentially undredged corridors between trenches), and/or by deposition of outwash fines and

sand from the dredger. Literature data suggest that the licence area is in an area of sufficient near-bottom current movement that fine material may be re-suspended and exported in the bottom nepheloid layer to the continental slope (Monteiro *et al.* 2005, Inthorn *et al.* 2006, Mollenhauer *et al.* 2007). Nonetheless, at this depth the strength of the bottom currents can be expected to be too low for bedload transport of sand, as a significant shear-stress is needed to mobilise sand particles, and only fine material and detritus may be moved by the current. The local current regime is frequently altered after dredging, with consequences for the sedimentation and deposition process. Dredging may sufficiently alter seafloor topography to slow down currents over the tracks or pits, thereby acting as traps for finer sediments and detritus (Newell *et al.* 1998). This can lead to an increased organic matter loading and increased oxygen demand, which in an already low-oxygen stressed environment can result in anoxia and hydrogen sulphide production. However, it is questionable whether a depth difference of 1-2.5 m (possibly up to 3 m) over a relatively large area will lead to a drastic drop in near-bottom current strength over the entire dredged area. In SP-1, the maximum resource depth is also only 1.85 m, so that the dredge-depth in this target area will be mostly 1 m, and only localised areas will exceed 1.5 m. It is more likely that after dredging the area will be structurally very heterogeneous, criss-crossed by small pits and mounds, perhaps similar in effect to megaripples that can sometimes be found in deep-water environments (Viana *et al.* 1998). Areas of detritus accumulation will then also be patchily distributed occurring in trenches and pits within the larger dredged area.

Where long-term changes in sediment characteristics and/or biochemical conditions occur, a shift in community structure is therefore likely as the original community may be unable to adapt to the new conditions (Kaplan *et al.* 1974, Herrmann *et al.* 1999, Desprez 2000). Traditionally, the ecological recovery of the disturbed seafloor has been defined as the establishment of a successional community of species, which progresses towards a community that is similar in species composition, population density and biomass to that previously present (Ellis 2003). Measures used to assess recovery typically include biodiversity analysis such as the numbers of species and/or individuals in an assemblage. However, this approach presents a number of challenges, especially when the physical characteristics of the sediment have been altered to such an extent that it can no longer accommodate its original assemblage. Recovery in the sense of the above definition may thus not be achieved (only when the sediment properties revert to their original state). For this reason, and because it is important to know how ecosystems work, it may be more sensible to consider the functional capacity (or health) of the ecosystem rather than simply the range and proportion of species present. Benthic organisms perform a number of ecosystem-level processes, also termed ‘ecosystem functions’ (Hooper *et al.* 2005). These functions include all metabolism, catabolism and dynamic processes such as sediment bioturbation, active re-suspension, and biogenic structure formation, as well as organic matter degradation, carbon burial, nutrient flux, microbial grazing and the sequestration of harmful substances (Snelgrove 1998). Some ecosystem functions can be undertaken by a variety of different organisms, leading to the notion of possible functional redundancy, whereby the loss of a particular species may not affect the basic functioning of an ecosystem as long as the function performed by that species is taken up by another species from the same functional group (Solan *et al.* 2004, Hooper *et al.* 2005). The extent to which species can be lost or changed before basic ecosystem processes are compromised depends on the functional richness (i.e. the number of functional groups) in an ecosystem (Hooper *et al.* 2005). To address this issue, many studies have recently focussed on functional diversity to assess faunal recovery following anthropogenic perturbations by incorporating biological differences among species showing that function- or trait-based diversity metrics may represent appropriate additional methods for assessing changes in ecosystem function (e.g. Borja *et al.* 2003, 2010, Bremner *et al.* 2006, Josefson *et al.* 2009, Cooper *et al.* 2008, Hussin *et al.* 2012). In terms of dredging impact on functional diversity, communities of organisms inhabiting an area of dredged seabed may possibly differ in composition or diversity from the pre-dredged state, but may develop similar functional capacity through the recovery process (functional recovery). Therefore, system recovery may not require similar biomass, biodiversity or community composition. This concept allows for an ecosystem to be altered without immediately reaching the conclusion that it has been permanently damaged (Cooper *et al.* 2008, Hussin *et al.* 2012). However, a drawback of the use of functional analysis is that many of the functional-trait indices developed over the last years (e.g. Biological Traits Analysis (BTA), Infaunal Trophic Index (ITI), Somatic Production, AZTI Marine Biotic Index (AMBI), among others) require detailed knowledge of the functional traits or guilds on the species level; information that is often not available for many species in a given habitat.

In sum, the removal of the upper 1-2.5 m (possibly up to 3 m) sediment layer will result in near total loss of the benthos. At a maximum annual dredging rate of 3 km$^2$, the size of the dredged area will be approximately 60 km$^2$ after 20 years of dredging. Note that this value of 60 km$^2$ represents the sum of all the dredge paths rather than a discrete mined block or blocks. The dredged paths, however, will be distributed primarily over two target areas (SP-1 and SP-2) that are >10 km apart. It can be expected that the exposed sediments after dredging will differ from the original surface sediments and that at this water depth, sediment refill rates into the dredged areas are very slow. Furthermore, it is possible that dredge pits may act as trap for finer material, which through increased decomposition rates could lead to severe hypoxic or anoxic conditions. The study area is naturally subject to hypoxic conditions and the local community will be adapted to low-oxygen conditions, but a further reduction in dissolved oxygen or a switch to anoxic conditions will result in colonisation by very different benthic assemblages (e.g. potentially bacteria-dominated) to those prior to dredging. Such conditions can be expected to persist for a long time whereby recovery to the original community composition may not be achieved for a very long time (several decades or even longer) but functional recovery may be reached sooner.

During the life of the mine, NMP will conduct the collection of exploration and environmental samples (core samples and grab samples) in the larger ML170. Exploration sampling will be conducted primarily by gravity cores of 6.5 cm in diameter that can penetrate to a depth of 2 m, vibracores penetrating to 6.0 m, and occasionally larger grabs with a 2.5 tonne (3 m$^2$) bite. Environmental sampling will typically be conducted with grab samplers such as Van Veen grabs that sample a maximum area of 0.2 m$^2$ down to a depth of 20 cm. The area disturbed by these tools is very small, and slumping from the sites of the hole will fill in the area rapidly. No measurable effects are expected from this exploration and environmental work. However, information gained during ongoing exploration can contribute to further understanding of the environment.

*Loss of Benthic Organisms as Food Source*

Benthic organisms serve as an important food source for many fish and larger benthic animals. The loss (temporarily or permanently) of this basic food source reduces the amount of food available both directly to demersal fish, as well as to their prey utilising these resources, and may thus have a cascading effect through the food chain. Fish are mobile and will move out of the dredged areas to feed elsewhere but this will increase pressure on the food source in the surrounding areas. However, the spatial extent over which food supply in form of benthic biota is reduced, is in a regional context relatively small. After 20 years of dredging an area of 60 $km^2$ will have been dredged, but this does not imply that an entire area of 60 $km^2$ is being lost at the same time. The dredging schedule prescribes that an area is dredged to depletion (which might require several dredge cycles) before the dredger moves to a new area. This means that while there is for 20 years a continuous disturbance of new areas, completed dredge sites will in the meantime begin the long process of (functional) recovery. Colonisation usually starts soon after cessation of dredging and although the community might be different from the original one, and is likely to remain less diverse and abundant for a long time, it still might serve as food source. The 60 $km^2$ of dredged area will also be divided primarily over the two target areas, which are separated by >10 km. There are thus two smaller populations of displaced demersal fish, rather than one bigger one. This spreads and therefore reduces the pressure on food resources in the undredged areas surrounding the target areas. However, if large areas or all of the dredged areas turn anoxic, macrofauna is unlikely to colonise the areas in significant numbers (if at all). In this case, bacterial mats may develop and be fed upon by the bearded goby, which seems to be an opportunistic and versatile feeder. The loss of benthic biota as food source is further discussed and evaluated in the Fishery Specialist Study (Appendix 1a).

*Sulphur-oxidising Bacterial Mats*

A major concern voiced during the public participation meetings, is the potential removal of bacterial mats by dredging, specifically mats of the large sulphur-oxidising bacteria *Thiomargarita* and *Beggiatoa*. There is no direct proof of the absence or presence of sulphur-bacteria in the mining licence area, but evidence from published data strongly suggests that offshore the mud belt at 24°S and at depth beyond 200 m, concentrations of large sulphur-oxidising bacteria are likely to be very low, if present at all. Consequently, there would be no disruption of the chemosynthesis due to removal of extensive large sulphur-oxidising bacterial mats. However, as this is based on circumstantial evidence, it is absolutely necessary to verify this assessment by surveying the target areas for bacterial concentrations.

In this context, a point for consideration is that if the giant bacteria would normally occur in this area, it is likely that demersal trawling will have disturbed such mats as these overlie the surficial sediments (see ROV photo Figure 11), which typically are being heavily disturbed by bottom-trawl gear (Thrush and Dayton 2002). Depending on the frequency of trawling, this disturbance may generally prevent the development of bacterial mats in trawling areas where $H_2S$ concentrations would otherwise be sufficient for sulphur-oxidising bacteria (if such areas exist beyond the 200-m isobaths).

Clostridium botulinum *Proliferation*

There is concern that the dredging works might result in a possible proliferation of bacteria, particularly the anaerobic bacterium *Clostridium botulinum*. *C. botulinum* is a Gram-positive, rod-shaped bacterium that produces several neurotoxins, subdivided in types A-G, that cause the flaccid muscular paralysis seen in botulism. Only types A, B, E, and F cause disease in humans while types C and D cause disease in cows, birds, and other animals. The bacteria are usually found in terrestrial soil but occur also in freshwater and marine sediments. The spores of the non-proteolytic *C. botulinum* type E are widely distributed in the aquatic (marine and fresh water) environment in the temperate and arctic zones and this type seems to be a true aquatic bacterium (Huss 1980). Up to 100% of sediment samples from coastal areas, particularly in closed, shallow fjords and from aquaculture ponds may contain the organisms (Huss 1980, Huss *et al.* 2004). A much lower prevalence is found in live fish although up to 100% of fish from aquaculture and certain coastal waters may carry this organism. For example, fish caught in Scandinavian waters are often contaminated with *C. botulinum* type E (Hyytiä *et al.* 1998) but it also occurs in other marine animals such as crabs, shrimps, and shellfish (Craig *et al.* 1968). In contrast, fish caught in the high seas are generally free from *C. botulinum* (Huss *et al.* 2004).

*C. botulinum* is an obligate anaerobe (i.e. it cannot tolerate oxygen), and is only able to produce the neurotoxin during sporulation, which can only happen in an anaerobic environment. In unfavourable growth environments, it produces endospores to preserve the organism's viability and permit survival in a dormant state until the spores are exposed to favourable conditions. *C. botulinum* type E is able to grow and produce toxins at temperatures as low as 3.3 C, but their spores are thermosensitive. Multiplication of *C. botulinum* type E in the marine environment seems to occur *in situ*, but might not occur in the bottom sediments themselves (Huss 1980, Huss *et al.* 2004). Rather fish, or the presence of a rich aquatic fauna, appear to contribute significantly to a high incidence of type E. Live fish harboring the organism would be active in its dissemination, and on death would be foci for its multiplication. It is suggested that *C. botulinum* type E outbreaks are linked to episodic anoxia events in shallow waters that lead to a mass-kill of fish and other aquatic life. The rotting biomass at the seafloor can then enable propagation of *C. botulinum* type E (Hielm *et al.* 1998). Contamination of fish with *C. botulinum* is through ingestion of contaminated sediments or invertebrate fauna (Hielm *et al.* 1998, Perez-Fuentetaja *et al.* 2006). In the freshwater systems of the Great Lakes, large mortalities of fish and waterfowl that have fed on infected fish, have been linked to botulism caused by *C. botulinum* type E (Perez-Fuentetaja *et al.* 2006).

Little is known about the factors influencing spore survival of *C. botulinum* in natural marine environments. It is possible that oxygen, combined with high salinity in oceans, exerts a substantial stress on the viability of *C. botulinum* type E spores (Hielm *et al.* 1998). It is also hypothesized that shallower rather than deeper locations are the growth foci for the bacterium. This might explain why the Baltic Sea, with its low salinity level and low average depth, has such a high prevalence of type E spores (Hielm *et al.* 1998). The closely adjacent North Sea is as rich in aquatic biomass in places, but is not contaminated with *C. botulinum* type E (Cann *et al.* 1968), despite a steady influx of spores from the Baltic Sea (Cann *et al.* 1965).

Large areas of the Namibian central shelf are affected by seasonal or perennial anoxic conditions, and it is thus possible that *C. botulinum* type E occurs naturally in the underlying sediments. Its presence, however, seems to be primarily restricted to shallow and enclosed waters and it seems unlikely to occur in appreciable numbers in sediments of open waters at depth of 200 m. An intensive literature search has produced no evidence of any reported *in situ* outbreaks of *C. botulinum* contamination in southern African fish populations. Any incidences of botulism in the region were related to contaminated canned food where the infection occurred in the can (Frean *et al.* 2004).

In the worst case scenario, changes in the small-scale near bottom hydrographical conditions as a result of dredging might create an anaerobic environment over an area of 60 $km^2$ after 20 years of dredging. Put into a regional context, the addition of 60 $km^2$ anaerobic environment where the bacterium could occur to the naturally occurring vast area of the azoic zone, is minor. If proliferation of *C. botulinum* in anaerobic zones were a problem in Namibian waters, it is unlikely that it would be aggravated by this spatial addition, and the impact is thus regarded as negligible. This, however, should not indemnify the fishing industry from complying with any regulations with regard to potential *C. botulinum* contamination (see FAO Technical Paper No. 444 by Huss *et al.* 2004).

## 4.2 RE-DEPOSITION OF SUSPENDED MATERIAL

During dredging with a TSHD, suspended sediments can be generated near the sea bottom as the draghead moves over the seabed, and at the surface arising from the overflow of material from the hoppers during loading. The impact of suspended sediment plumes on the water column and its inhabitants are discussed in detail in another specialist report (Appendix 1b). In the present study, only the effects of the near-bottom plume and the impacts relating to the re-deposition of suspended material will be discussed.

Sediment re-suspension from the draghead increases with hopper filling speed and travel speed of the dredger, but is mainly dependent on the properties of the sediments (Johnson and Parchure 1999). Suspended sediment concentrations generated at the point of dredging tend to decline as the sediments become coarser. The size fractions of greatest consequence are the silts, muds and clays (<63 µm) as these create the highest level of turbidity (Johnson and Parchure 1999). Near-bottom dredge plumes usually decrease rapidly with distance from the dredger. Kirby and Land (1991), for example, recorded suspended sediment concentrations decreasing from a maximum of 1,100 mg/ℓ at the cutter head of a cutter suction dredger to 20-90 mg/ℓ only 50 m away. The overburden in the target areas is described as silty sand with significant gravel component and will thus have relatively rapid settlement rates, and the re-suspended sediment should disperse only over short distances.

Benthic species which may be impacted by re-suspension of sediments at the bottom include bivalves and crustaceans. Suspended sediment effects on juvenile and adult bivalves occur mainly at the sublethal level with the predominant response being reduced filter-feeding efficiencies occurring generally at concentrations of about 100 mg/ℓ. Lethal effects are seen at much higher concentrations (>7,000 mg/ℓ) and at long-term (3 weeks) exposures (Clarke and Wilber 2000). Crustaceans appear similarly resistant to lethal effects with 25% mortality rate reported at 10,000 mg/ℓ for >240 hour exposures (Clarke and Wilber 2000). One has to note though that any noticeable effect will only occur where the near-bottom plume disperses over the edge of the target panels. Organisms inside the dredge site will all be removed and further impacts by suspended sediments are irrelevant to them. Effects related to draghead generated plumes are thus expected to be negligible.

The second more important plume is a result of the overflow from the dredger's hopper compartments. Generally, the extent and area over which plumes disperse are dependent on the strength and direction of the prevailing currents and winds, and the particle size of the material in question. The rate of sedimentation from the water column is also a function of the particle size of the suspended material. The larger the particles, the greater the rate of settlement and therefore larger particles will be deposited closer to the site of dredging than smaller ones. It is estimated that sediments <200 µm will be lost in the overspill during loading. The portion >63 µm will settle more rapidly to the seabed and thus not disperse very far, whereas muds, silts and clays will stay longer in suspension. For a more detailed account of the possible size and extent of the plumes refer to the specialist study on water column issues (Appendix 1b). Relevant for bottom-dwelling communities is the eventual re-deposition of these suspended sediments, potentially smothering benthos adjacent and downstream to the dredged area. For example, a prominent depo-center of fine material is located between 24.5°S and 26°S on the upper continental slope in water depths of 400-1500 m and it is suggested that the material in this depo-center is exported from the shelf in the nepheloid layer (Inthorn *et al.* 2006). Sediments released with the overflow may thus enter the nephelopid layer and ultimately settle in this depo-center increasing the sediment settling rate in this area, as well as in areas closer to the dredging site.

Deposition of sediments can lead to smothering which involves a reduction in nutrients and oxygen, clogging of feeding apparatus as well as affecting choice of settlement site (Hiscock 1983, Rodrýguez *et al.* 1993, Lam *et al.* 2003), and post-settlement survival (Hunt and Shebling 1997), and may affect animals directly or indirectly, either lethally or sublethally. In general terms, the rapid deposition of material from the water column is likely to have more of an impact on the soft-bottom benthic community due to smothering effects, than gradual sedimentation to which benthic organisms are adapted and able to respond. However, this response depends to a large extent on the nature of the receiving community. Studies have shown that some mobile benthic animals are able to migrate vertically through more than 30 cm of deposited sediment (Maurer *et al.* 1979, 1986, Newell *et al.* 1998, Ellis 2000), and in a study on macrofauna on a bathyal sea floor 50% of the organisms burrowed back to the surface through 4 - 10 cm of rapidly deposited sediment (Kukert 1991). Meiofauna was also found to be able to migrate through deposited sediment (Schratzberger *et al.* 2000). In contrast, sedentary communities potentially could be adversely affected by both rapid and gradual deposition of sediment. Filter-feeders are generally more sensitive to suspended solids than deposit-feeders, since heavy sedimentation may clog the gills. However, research on filter-feeders living in coastal waters showed that bivalves in particular are highly adaptable, and can maintain their feeding activity over a wide range of inorganic particulate loads (Iglesias *et al.* 1996, Navarro *et al.* 1996, see also above). Impacts on highly mobile invertebrates and fish are likely to be negligible since they can move away from areas subject to re-deposition.

A dredge-plume study for the TSHD trial dredging in 105-130 m depth in the Atlantic 1 MLA off southern Namibia measured low deposition rates of 2 mm about 1.5 km from the dredge panels (CSIR 2006). Model predictions for dredging operations outside the mud belt estimated potential accretion of 100 mm in the immediate vicinity of the dredge panels, reducing to negligible thickness over several kilometres in direction of the prevailing current/wind direction (CSIR 2006). While the Atlantic 1 MLA dredging area and the Sandpiper Licence Area have different hydrographical conditions, a similar scenario might still be expected in the licence area. This suggests that re-settling of sediment may occur over a relatively large area but the sedimentation thickness is likely to be very low. Smothering effects are restricted to localised areas near the dredged area. Again, only deposition on undisturbed seafloor areas is of importance while deposition of sediments in the dredged-out area will have no effect. In areas of naturally high deposition such as the depo-center of the continental slope, the local biota will also be adapted to higher settling rates, and are likely to cope with the intermittently higher sedimentation. However, the cumulative effect of dredging in a relatively restricted area can result in repeated sedimentation and thus higher deposition rates. Depending on the depth of the resource, the same area may be dredged up to 5-6 times (maximum resource depth 2.25 m; depth of a single dredge swath is 0.4 m x 6 cycles = 2.4 m) which would at a dredge cycle of ~37 hours result in <10 days of intermittent dredging in a particular area. Re-deposition of fines over exactly the same area, however, would require that the weather conditions are very stable over this period (same wind direction and strength). In sum, re-deposition impacts are expected to be low as deposition-thickness rates are assumed to be relatively low and biotic communities living in areas of naturally high sedimentation rates (e.g. the depo-center on the continental slope) can be expected to be adapted to sedimentation disturbances.

Some studies have documented a positive effect of dredging in that the disturbance of sediments released sufficient organic material to enhance the species diversity and population density of organisms outside the immediate zone of deposition of suspended material (Jones and Candy 1981, Poiner and Kennedy 1984). Crushed organisms returned to the sea with the overspill may also cause organic enrichment (Newell *et al.* 1998). On the other hand, a further increase in organic matter with associated bacterial decomposition in bottom waters, which are already oxygen-depleted, may be considered deleterious (Newell *et al.* 1998). The organic matter content in the sediments of the target area is suspected to be relatively low consisting primarily of reworked deposits originating from the inshore mud belt. The small amount of organic matter that will be released with the overflow water including any crushed benthic fauna is also likely to spread sufficiently enough that once it settles it will not result in a measurable effect.

## 4.3 DIRECT/INDIRECT EFFECTS OF RE-SUSPENDED DISSOLVED NUTRIENTS AFTER DREDGING

There is concern that dredging will mobilise bioavailable dissolved nutrients from the sediments and/or the pore waters, and release these into the water column either directly in the bottom waters or with the overflow of the fines from the TSHD. The phosphorite pellets, in contrast, are insoluble in seawater and will not contribute to elevated nutrient levels in the water column. Increased nutrient levels near the water surface in the photic zone may lead to greater phytoplankton production, with resultant increased levels of organic matter deposition and bacterial decomposition. This in turn might lead to anaerobic conditions.

An important consideration is thus whether nutrient concentration in the sediments of the target areas can be suspected to be very high. Van der Plas *et al.* (2007) have studied biogeochemical characteristics of sediments along a cross-shelf transect at 22°S including particulate organic carbon (POC) and particulate organic nitrogen (PON) loading in the sediments, and nutrient concentrations in the sediment pore waters. They reported that in mud belt sediments at 115 m depth, firstly the mud content was very high (>96%), and secondly POC and PON concentrations were very high while the C:N ratio was very low indicating that the organic matter was freshly derived material. In contrast, sediments further offshore at 200 m, and at stations between the POM-rich bands as described by Bremner (1978), were mostly sandy and POC and PON concentrations decreased sharply. Sediment pore water nutrient concentrations (as well as $H_2S$ concentrations), except for nitrate and nitrite, followed the same trend as the sediment characteristics and were highest in the mud belt, but drastically lower in the deeper shelf sediments. The low nitrate and nitrite concentrations in the mud belt were linked partly to sulphur bacteria that use nitrate to oxidise $H_2S$. Ammonium and phosphate concentrations were also high in mud belt sediments, but showed a generally decreasing trend with depth offshore from the mud belt. Although this study was conducted further north and thus outside of the licence area, these data in combination with the information collated for the environmental baseline section, suggest that the nutrient levels in the sediments as well as POM and PON concentrations are potentially relatively low when viewed in the regional context. Nonetheless, a certain increase in nutrient levels in the photic zone can be expected but this is unlikely to result in massive dying phytoplankton blooms reaching the sea bottom in such locally dense concentrations that this will cause anoxic seafloor conditions. This assessment is based on assumptions though and the levels of organic matter and nutrients in the target areas should be measured to verify it.

## 4.4 HYDROGEN SULPHIDE RELEASE

The possibility of hydrogen sulphide release from the dredging activity and the effects of $H_2S$ in the water column and on the pelagic communities are discussed in a separate specialist study on water column issues (Appendix 1b). Here are potential impacts on benthic communities described.

Hydrogen sulphide concentrations in pore waters and/or the bottom waters are likely to be relatively low or even absent in the three target areas. If significant concentrations of $H_2S$ were present in the upper layers of the sediments, the fauna would already be affected and only those capable of tolerating $H_2S$ will live in the area. The benthic baseline survey (Steffani 2010a) describes a low macrofauna species richness dominated by polychaetes as is typical for OMZs, but these animals are unlikely to occur in an anoxic hydrogen-sulphide environment. For example, sediments directly from the mud belt at 120 m depth with high concentrations of $H_2S$ and dense bacterial mats were found to not contain any macrofauna (Hundt *et al.* 2011, see also Dale *et al.* 2009). Gas pockets of methane and $H_2S$ may be trapped in deeper layers of the sediments, and could be released by dredging. This released gas could spread over the seafloor before it mixes with the overlying water column, and affect benthic organisms in adjacent undredged areas. However, this is unlikely to occur since most of any $H_2S$ released from the sediment is likely to be entrained by the dredging process and released into surface water in the lean water overflow. Again, such gas pockets have been found in thick (>8 m), muddy diatomaceous sediments from the centre of the mud belt, but have so far not been recorded from areas outside the mud belt (Emeis *et al.* 2004, see also Figure 7). However, it cannot be excluded that deeper sediments of >50 cm may contain $H_2S$ as so far all long sediment cores analysed from the central Namibian shelf contained some traces of the gas (V. Brüchert, pers. comm.). The possibility of some $H_2S$ release from the sediments can therefore not be completely excluded without direct measurements of $H_2S$ presence in the sediments from the target areas. If $H_2S$ is released from sediments, it will, however, presumably be sucked up with the dredged sediments and residual $H_2S$ at the seafloor will be in any case minimal. $H_2S$ release with the overflow from the hopper may, on the other hand, be possible.

# 5 EVALUATION OF IMPACTS

The following methods have been used to determine the significance rating of impacts identified in this benthic specialist study:

1. Description of impact - reviews the type of effect that a proposed activity will have on the environment;
2. What will be affected; and
3. How will it be affected.

Points 1 to 3 above are to be considered / evaluated in the context of the following impact criteria:

- *Extent*;
- *Duration*;
- *Probability*; and
- *Intensity / magnitude* .

These impact criteria are to be applied as prescribed in the table below:

| **Impact Criteria:** | | | | | | |
|---|---|---|---|---|---|---|
| **Extent** | **Dredge Area**<br>Per vessel cycle i.e. ~66,000m$^2$ or 6.6 ha | **Annual Mining Area**<br>Up to 3 km$^2$ | **Specific Mine Site** (SP1 or SP2) each is 22x8 km or 176km$^2$ | **Local**<br>25-50 km or 2,000km$^2$- 8,000km$^2$ | **Regional**<br>50-100 km or 8,000km$^2$ – 30,000km$^2$ | **National**<br>100 km to EEZ (200 nautical miles)[1]<br>100 to 370 km, or >30,000km$^2$ |

| | | | | | |
|---|---|---|---|---|---|
| **Duration** | **Very Short Term**<br>3 days | **Short term**<br>3 days – 1 year | **Medium term**<br>1 - 5 years | **Long term**<br>5 – 20 years | **Permanent**<br>>20 years (life of mine) |

| | | | | |
|---|---|---|---|---|
| **Intensity/ Magnitude** | **No lasting effect**<br>No environmental functions and processes are affected | **Minor effects**<br>The environment functions, but in a modified manner | **Moderate effects**<br>Environmental functions and processes are altered to such extent that they <u>temporarily</u> cease | **Serious effects**<br>Environmental functions and processes are altered to such extent that they <u>permanently</u> cease |

| | | | | |
|---|---|---|---|---|
| **Probability** | **Improbable** | **Possible** | **Probable** | **Highly Probable/ Definite** |

The <u>status of the impacts and degree of confidence</u> with respect to the assessment of the significance are stated as follows:

[1] 1 nautical mile = 1,85 kilometres

**Status** of the impact: A description as to whether the impact is positive (a benefit), negative (a cost), or neutral.

**Degree of confidence in predictions**: The degree of confidence in the predictions, based on the availability of information and specialist knowledge. This had been assessed as high, medium or low.

Based on the above considerations, the specialist provides an overall evaluation of the significance of the potential impact, which is described as follows:

| | None | Low | Medium | High |
|---|---|---|---|---|
| **Impact Significance** | A concern or potential impact that, upon evaluation, is found to have no significant impact at all. | Any magnitude, impacts will be localised and temporary<br><br>Accordingly the impact is not expected to require amendment to the project design | Impacts of moderate magnitude locally to regionally in the short term<br><br>Accordingly the impact is expected to require modification of the project design or alternative mitigation | Impacts of high magnitude locally and in the long term and/or regionally and beyond<br><br>Accordingly the impact could have a 'no go' implication for the project unless mitigation or re-design is practically achievable |

Furthermore, the following are being considered:

- Impacts are described both **before** and **after** the proposed **mitigation** and management measures have been implemented;
- Where possible the impact evaluation takes into consideration the **cumulative effects** associated with this project. Cumulative impacts can occur from the collective impacts of individual minor actions over a period of time and can include both direct and indirect impacts;
- **Mitigation / management actions:** Where negative impacts were identified, the specialists specified practical mitigation measures (i.e. ways of avoiding or reducing negative impacts); and
- **Monitoring (forms part of mitigation):** Specialists recommend monitoring requirements to assess the effectiveness of mitigation actions, indicating what actions are required, the timing and frequency thereof.

## 5.1 ASSESSMENT OF IMPACTS

Dredging is destructive in nature and therefore no positive impacts on the biophysical environment are expected. Impacts on the benthic communities from the proposed Sandpiper Phosphate Project are only expected during the operational phase of the project (but may extend beyond the closure of the project). Impacts are thus assessed for the operational phase only, and not for the initiation and decommissioning phases.

| ***Nature of the impact*** | **The removal of the upper 1-2.5 m (possibly up to 3 m) of sediment by dredging will result in the loss of the benthic biota associated with the sediment. The exposed sediments are likely to be different from the original superficial deposits, and sediment refill rates at this depth are likely to be very slow. Colonising assemblages are likely to differ to those present prior to the dredging activity.** |
|---|---|
| ***Extent*** | Specific mine site - the loss of the benthic community is restricted to the dredged-out areas. Target areas are 22 x 8 km in size but only a maximum of 3 $km^2$ per annum will mined, which amounts to a total of 60 $km^2$ after 20 years of mining (the period for which the mining licence is issued). |
| ***Duration*** | Permanent (>20 years life of mine) - the recovery to the original community is likely to take longer than the life of mine or even may not be achieved in a meaningful time-scale. Recovery to functionally similar communities that provide similar ecosystem services as the original communities might, however, occur sooner (Long term). |
| ***Intensity*** | Moderate to serious effects - recovery to the original community is likely to take very long (several decades, whereby beyond life of mine is classified as permanent), but recovery to a community providing similar ecosystem functioning is likely to occur sooner, e.g. environmental functions and processes are altered to such an extent that they temporarily cease. |
| ***Probability*** | Definite |
| ***Status (+ or -)*** | Negative |
| ***Significance (no mitigation)*** | Medium - the duration of the impact is permanent in view of recovery to original community but recovery to a different community but providing similar ecosystem services may occur sooner, and the intensity is moderate to serious but the extent is confined to the mine site, maximum of 60 $km^2$ after 20 years of dredging. |
| ***Mitigation*** | Leave behind a residual sediment layer of at least 30 cm of the original deposit thickness to cover the clay footwall. Leave behind undredged strips to unable migration of mobile organisms from these areas. |
| ***Significance (with mitigation)*** | Medium - the residual sediment layer will provide a substrate to be colonised by benthic organisms. Nonetheless, the recovering communities will be very different to those prior to dredging. |
| ***Confidence level*** | Medium - the assessment is based on assumptions that are arrived from publicly available data, while data directly from the target areas are limited. A monitoring programme is needed to confirm the assumptions. |

| ***Nature of the impact*** | **Further exploration and environmental work will be conducted in the larger ML170 that will remove benthic biota.** |
|---|---|
| ***Extent*** | Dredge Area – Gravity and vibro-cores are 6.5 in diameter, van Veen grab samples with an area of max. 0.2 $m^2$ and larger grabs sample 3 $m^2$ bite. The total area disturbed by these tools even after extensive exploration campaigns will be very small. |
| ***Duration*** | Short term – it is expected that slumping from the side of the holes will quickly fill in the disturbed area and migration from the adjacent area is fast. |
| ***Intensity*** | No lasting effects – recovery will be very fast as many animals will be transported into the disturbed area with the material slumping from the sides. |

| | |
|---|---|
| ***Probability*** | Probable |
| ***Status (+ or -)*** | Negative |
| ***Significance (no mitigation)*** | None – recovery will be very rapid and effects on the system will not be measurable. |
| ***Mitigation*** | No mitigation necessary |
| ***Significance (with mitigation)*** | None – recovery will be very rapid and effects on the system will not be measurable. |
| ***Confidence level*** | High |

| ***Nature of the impact*** | **The depth of the dredged area might change local near bottom hydrographical conditions and thus act as trap for very fine material. This could lead to high decomposition rates and consequently anoxic conditions and $H_2S$ concentrations in the sediments.** |
|---|---|
| ***Extent*** | Specific mine site - Target areas are 22 x 8 km in size but only a maximum of 3 $km^2$ per annum will mined, which amounts to a total of 60 $km^2$ after 20 years of mining (the period for which the mining licence is issued). |
| ***Duration*** | Permanent - sediment refill rates are expected to be very low at the water depth of the target areas. |
| ***Intensity*** | Moderate to Serious effects - anoxic conditions are deadly for most benthic communities but large sulphur-oxidising bacteria can thrive under these conditions. |
| ***Probability*** | Probable – localised anoxic conditions may occur in the deeper trenches and pits. |
| ***Status (+ or -)*** | Negative |
| ***Significance (no mitigation)*** | Medium - duration is permanent and intensity moderate to serious, but extent is restricted to the mine area and large areas of the inner shelf are naturally subjected to anoxic conditions. |
| ***Mitigation*** | Leave behind a residual sediment layer of at least 30 cm, which will reduce the depth of the dredged-out area. |
| ***Significance (with mitigation)*** | Low to medium - a dredged depth of an average of 1.7 m (possibly up to 3 m) over a relatively large area is unlikely to reduce bottom current speeds to such an extent that very fine material will significantly accumulate in the dredge area. |
| ***Confidence level*** | Medium - the assessment is based on assumptions that are arrived from publicly available data, while data directly from the target areas are limited. A recovery survey is needed to confirm the assumptions. |

| ***Nature of the impact*** | **Dredging removes mats of large sulphur-oxidising bacteria from the sediment surface and from the upper layer.** |
|---|---|
| ***Extent*** | Specific mine site - Target areas are 22 x 8 km in size but only a maximum of 3 $km^2$ per annum will mined, which amounts to a total of 60 $km^2$ after 20 years of mining (the period for which the mining licence is issued). |
| ***Duration*** | Medium to long term – the recovery of bacterial mats depends on the development of sufficient $H_2S$ concentrations. This requires anoxic conditions that can only develop when high concentrations of organic matter accumulate in the dredge area. Although higher organic loading might be a possibility as the dredge area may act as trap, it will take a long time to build up enough material for anoxic conditions and high $H_2S$ concentrations. |
| ***Intensity*** | Minor to moderate effects – the large sulphur bacteria are important in oxidising the toxic $H_2S$ thereby reducing its diffusion into the water column. Their removal will disrupt this, on the other hand, the removal of the sediments will also remove any $H_2S$ contained in the sediments, and $H_2S$ fluxes from the dredge area are thus not expected unless the system turns anoxic. If this happens, the bacterial mats are likely to return. |
| ***Probability*** | Improbable – evidence from published data strongly suggests that offshore the mud belt at |

| | |
|---|---|
| | 24°S and beyond the 200 m isobaths concentrations of large sulphur bacteria are low or absent. |
| ***Status (+ or -)*** | Negative |
| ***Significance (no mitigation)*** | Low – concentrations of large sulphur bacteria is assumed to be low or absent. |
| ***Mitigation*** | No mitigation necessary |
| ***Significance (with mitigation)*** | Low – concentrations of large sulphur bacteria is assumed to be low or absent. |
| ***Confidence level*** | Medium - the assessment is based on assumptions that are arrived from publicly available data, while data directly from the target areas are limited. An initial survey is needed to confirm the assumptions. |

| ***Nature of the impact*** | **The anaerobic bacterium *Clostridium botulinum* type E might proliferate in the dredged area if the system turns anoxic, and may pose a health risk to humans and wildlife when entering the food chain.** |
|---|---|
| ***Extent*** | Specific mine site - Target areas are 22 x 8 km in size but only a maximum of 3 km$^2$ per annum will mined, which amounts to a total of 60 km$^2$ after 20 years of mining (the period for which the mining licence is issued). |
| ***Duration*** | Short term – if the system turns anoxic this will be of long term or permanent duration, but *C. botulinum* proliferation is linked to periodic massive die-offs of fish and other aquatic life that might occur during extreme events such as $H_2S$ eruptions. Once bacteria proliferate they may enter the food chain by ingestion of contaminated sediments from the dredge area. |
| ***Intensity*** | Serious effects – botulism caused by the bacteria can be lethal to human and wildlife. |
| ***Probability*** | Improbable – no *in situ* contamination of fish populations by the bacterium has been reported for southern African fish populations. Literature data suggest that the distribution of the bacteria is limited in deeper saline waters. If the bacteria are a problem in Namibian waters, it is unlikely that the addition of 60 km$^2$ of anoxic seafloor will add any measurable risk of bacteria proliferation to the already large areas of anoxic zone. |
| ***Status (+ or -)*** | Negative |
| ***Significance (no mitigation)*** | Low – proliferation of bacteria is assumed to be a rare probability |
| ***Mitigation*** | No mitigation necessary but this should not indemnify the fishing (canning) industry from complying with any regulations regarding *C. botulinum* contamination |
| ***Significance (with mitigation)*** | Low – proliferation of bacteria is assumed to be a rare probability |
| ***Confidence level*** | Medium – very little is known about the natural life-cycle of the bacteria and this assessment is based on data from the northern hemisphere. |

| ***Nature of the impact*** | **High suspended sediment concentrations near the sea bottom generated by the drag head and subsequent re-deposition of the material causes smothering effects.** |
|---|---|
| ***Extent*** | Dredge area – sedimentation effects will only be relevant along a narrow strip around the dredge site as any re-depositions inside the dredge area will have no impact since the animals are removed. |
| ***Duration*** | Very short term – smothering of a particular area occurs only during the dredging activity, maximum dredging activity per area is assumed to be <10 days for intermittent (16 hour-cycle) dredging. |

| | |
|---|---|
| ***Intensity*** | Minor effects – some organisms in the immediate vicinity of the dredge site may be impacted on a lethal level but the majority of impacts can be expected on a sub lethal level as many animals can cope with relatively high short-term suspended material concentrations. |
| ***Probability*** | Highly probable |
| ***Status (+ or -)*** | Negative |
| ***Significance (no mitigation)*** | Low – very small extent, very short duration and low intensity |
| ***Mitigation*** | No mitigation necessary |
| ***Significance (with mitigation)*** | Low – very small extent, very short duration and low intensity |
| ***Confidence level*** | High – studies on draghead plumes have shown that the affected area is very small |

| ***Nature of the impact*** | **Re-deposition of particles in the overflow plume causes smothering of benthic organisms, particularly in the depo-center on the continental slope** |
|---|---|
| ***Extent*** | Local to regional – the fines (<63 micron) in the plumes may be transported for several kilometres but upon entering the nepheloid layer, material may be transported to the depo-center ~100 km south-west of the licence area. Significant deposition-thicknesses are, however, expected to occur only in the immediate vicinity of the dredge area. |
| ***Duration*** | Very short term – the overflow plumes will only be generated during dredging which occurs within a 37-hour dredge cycle for approx. 16 hours |
| ***Intensity*** | Minor effects – animals in the immediate vicinity of the dredge area may be affected by smothering, elsewhere sedimentation rates are expected to be very low. |
| ***Probability*** | Probable |
| ***Status (+ or -)*** | Negative |
| ***Significance (no mitigation)*** | Low – although widespread, re-deposition rates are expected to be low, and higher rates are limited to the immediate vicinity of the dredge area. Communities in the depo-center where higher settling rates may occur, are also likely to be adapted to sedimentation as this is a naturally high sedimentation area. |
| ***Mitigation*** | No mitigation necessary |
| ***Significance (with mitigation)*** | Low – although widespread, re-deposition rates are expected to be low, and higher rates are limited to the immediate vicinity of the dredge area. Communities in the depo-center where higher settling rates may occur, are adapted to sedimentation as this is a naturally high sedimentation area |
| ***Confidence level*** | Medium – assumed low sedimentation rates are based on a study conducted in slightly shallower waters of southern Namibia with different hydrographical conditions. |

| ***Nature of the impact*** | **Dredging may mobilise dissolved nutrients from the sediments which could be released into the water column with the overflow. The increased nutrient level may result in extensive phytoplankton blooms, which upon death cause aggravated decomposition rates leading to anoxic conditions at the seafloor.** |
|---|---|
| ***Extent*** | Local – the released nutrients will spread with the overflow plume |
| ***Duration*** | Very short term – the overflow plumes will only be generated during dredging which occurs within a 37-hour dredge cycle for approx. 16 hours |
| ***Intensity*** | Minor effects – literature data suggest that dissolved nutrient concentrations in the target areas are relatively low, which means that only low amounts of nutrients will be mobilised. |
| ***Probability*** | Possible – it is likely that some nutrients will be mobilised but it is unlikely that this will result in massive dying phytoplankton-blooms reaching the sea bottom in such locally dense concentrations that this will cause anoxic seafloor conditions. |

| | |
|---|---|
| ***Status (+ or -)*** | Negative |
| ***Significance (no mitigation)*** | Low – due to potentially low dissolved nutrient concentrations in the target areas |
| ***Mitigation*** | No mitigation necessary |
| ***Significance (with mitigation)*** | Low – due to potentially low dissolved nutrient concentrations in the target areas |
| ***Confidence level*** | Medium - the assessment is based on assumptions that are arrived from publicly available data, while data directly from the target areas are limited. An initial survey is needed to confirm the assumptions. |

| ***Nature of the impact*** | **Release of hydrogen sulphide from the sediments affects benthic communities** |
|---|---|
| ***Extent*** | Local – released hydrogen sulphide may spread along the sea bottom affecting undredged areas and the associated biotic life. |
| ***Duration*** | Short term – the spread of hydrogen sulphide across the seafloor will be very short term and the gas will eventually mix with the seawater. The gas is, however, very toxic and will kill many animals in its path. Recovery of the benthic communities will be relatively rapid if hydrogen sulphide conditions are only temporary. |
| ***Intensity*** | Moderate effects – hydrogen sulphide is very toxic and will kill many animals but its presence is temporary. |
| ***Probability*** | Probable – literature data suggest that hydrogen sulphide concentrations in the near-bottom waters, pore waters and in the upper sediment layers in the target areas are very low. It can, however, not be excluded that deeper sediment layers may contain hydrogen sulphide. If hydrogen sulphide is present, it is presumably sucked up with the sediments and residual hydrogen sulphide at the seafloor will be minimal. |
| ***Status (+ or -)*** | Negative |
| ***Significance (no mitigation)*** | Low – hydrogen sulphide concentrations are assumed to be low, and the dredging process will also remove any gas contained in the sediments |
| ***Mitigation*** | No mitigation necessary |
| ***Significance (with mitigation)*** | Low – hydrogen sulphide concentrations are assumed to be low, and the dredging process will also remove any gas contained in the sediments |
| ***Confidence level*** | Medium - the assessment is based on assumptions that are arrived from publicly available data, while data directly from the target areas are limited. An initial survey is needed to confirm the assumptions. |

# 6 MITIGATION MEASURES AND MONITORING ACTIONS

As a result of the dredging operations to recover marine phosphate resources in ML170, trenches will be excavated in the seabed and the benthic biota associated with the sediments will be removed. The mine permit is issued for a period of 20 years, and at a maximum dredging rate of 3 km$^2$ per annum, this will lead to a dredged-out area of 60 km$^2$, primarily over SP-1, SP-2 and to a lesser extent SP-3. The dredge depth will be on average 1.69 m, 1.70 m, and 1.30 m for SP-1, SP-2 and SP-3, respectively. The maximum resource depths are 2.5 m, 2.25 m, and 1.85 m for SP-1, SP-2 and SP-3, respectively, as determined during exploration (possibly up to 3 m). These values may change as more information is obtained during the dredging operations. The phosphate layer contacts a clay footwall, whereby the total stripping of the phosphate resource would expose this footwall. The stiff clay footwall is less than ideal for small burrowing fauna, and it is strongly recommended that a residual sediment layer of 0.3 m of the original sediment thickness be left behind. This will provide unconsolidated soft-bottom substrate for animals to colonise. Nonetheless, it is expected that the residual sediment layer will have different sediment properties than the original surficial layers. Furthermore, if areas of undisturbed sediments are left between dredged furrows, colonisation of the dredged area by benthic organisms can be accelerated. Such undisturbed areas can provide an important source of adult benthic organisms, which subsequently migrate into the disturbed areas, enabling a faster recovery than might occur solely by larval settlement and growth.

As the dredging target sites are located at depths beyond the influence of surface waves, infilling rates will be slow as near-bottom sediment transport is expected to be low. It is recommended that high resolution geophysical surveys (e.g. side scan sonar) are conducted immediately after dredging, and 2-3 years post-dredging (and potentially at later years depending on the results) to determine the depth of the dredged trenches and the sediment infilling-rates.

The deep trenches may potentially result in changes in the near-bottom current regime reducing the speed of the current so that deeper trenches and pits may act as traps for fine material. The residual layer left behind will reduce the overall depth of the dredged area, however, organic matter accumulation may still occur.

Most of the assessments on potential impacts on the benthos are based on assumptions that are arrived from publicly available data from areas outside the ML170 area while data directly from the target areas are very limited. A verification survey is needed to confirm the assumptions. This should include sampling of the macrofauna and/or meiofauna in the target areas, as well as the surveying of the areas for the presence of bacterial mats. Further aspects of the survey should include measurements of organic matter concentrations in the sediments, dissolved nutrients, and $H_2S$ concentrations particularly in the deeper sediments. Surveying for bacterial mats could be done with a ROV. While macrofauna can be sampled with normal van Veen grabs, meiofauna would need to be sampled with more sophisticated grab tools such as a multi-corer. Measurements of dissolved nutrients and $H_2S$ also require analytical equipment that is not available to NMP. It is thus recommended that specialist consultants or scientists be engaged to discuss the programme to collect such verification data.

Continuing from the initial assessment survey, the severity of the removal and destruction of benthic communities by the dredging process and the subsequent recovery (functional recovery) process need to be ascertained. A post-dredging benthic monitoring programme thus needs to be established. There is continuous debate whether such monitoring programmes should focus on macrofauna or on meiofauna, or both (e.g. Somerfield *et al.* 1995, Coull and Chandler 1998, Kennedy and Jacoby 1999, Schratzberger *et al.* 2001). Typically macrofauna is the preferred option as sample collection and species identification is comparatively easier (Kennedy and Jacoby 1999). In low-oxygen environments such as OMZs, however, body size seems to be very important as small organisms are best able to cover their metabolic demands in the OMZ, and besides adaptation to low oxygen often have a capability to conduct anaerobic metabolism. Meiofauna may thus increase in dominance in relation to macro- and megafauna (Levin 2003). Nonetheless, although small organisms prevail, the species inventory of OMZs comprises the whole range between micro- and megafauna and many macrofauna species have developed adaptations to cope with life in hypoxic habitats (Gonzalez and Quinones 2000, Levin 2003, Arntz *et al.* 2006).

The difficulty in conducting meiofauna monitoring surveys in comparison to macrofauna studies favours the use of macrofauna for long-term studies. An inventory of the meiobenthos component during the initial survey will shed light on its relative importance in the benthos. The question is whether macrofauna alone may not sufficiently answer any questions with regard to the severity of the impact and potential recovery time. By no means is this report attempting to give an undisputed answer to this, but the extensive use of macrofauna surveys for a wide variety of anthropogenic disturbances suggests that data on macrofauna composition and abundance should be able to shed light on it. Macrofauna is also routinely collected in studies on OMZ benthos (e.g. Levin and Gage 1998, Levin *et al.* 2000, 2009, Ueda *et al.* 2000, Gallardo *et al.* 2004, Arntz *et al.* 2006, Gooday *et al.* 2009, Zettler *et al.* 2009). The original baseline survey (Steffani 2010a) used a 1-mm sieve to separate the macrofauna from the sediment as this is the standard mesh size used in macrofauna surveys (Rumohr 2009). Studies on macrofaunal abundance in OMZs, however, often use smaller sieve sizes in anticipation that many macrofauna species will be generally smaller (e.g. Gallardo *et al.* 2004, Gooday *et al.* 2009, Levin *et al.* 2009). During the initial survey, a second set of samples could be collected for macrofauna using a 500 or 300-micron sieve. In the laboratory analysis procedure, the size fractions <1 mm and >1 mm should then be analysed separately to permit comparison to the baseline study and indicate the right mesh size for the long-term monitoring study. Sampling should be undertaken both before the start of operations, as well as at regular intervals after completion of dredging to determine the (functional) recovery rates of the benthic communities. The sampling interval can best be determined after the first post-dredging sampling campaign (approx. 2-3 years after cessation of dredging). Sampling stations should include dredged and undredged (reference) stations in comparable environmental habitats (e.g. similar depth and sediment characteristics prior to dredging). Included in the sampling procedure should be at least the sampling for sediment properties (i.e. grain size analysis) as well as near-bottom dissolved oxygen concentrations and organic matter content. Continuous engagement with the authorities could facilitate the measurement of other important parameters throughout the monitoring programme.

# 7 CONCLUSIONS

An assessment of the risks associated with dredging for phosphate rich-sediments in the three target areas in the ML170 area identified nine potential negative impacts on the benthic biota in the target areas or beyond. Of these, two impacts are considered to be of medium significance, six of low significance, and one is assessed as having no significance. The impacts of medium significance are:

*Impact* - The removal of the upper 1-2.5 m (possibly up to 3 m) of sediment by dredging will result in the loss of the benthic biota associated with the sediment. The exposed sediments are likely to be different to the original superficial deposits, and sediment refill rates at this depth are likely to be very slow. Colonising assemblages are likely to differ from those present prior to the dredging activity.
*Significance* - Medium as the duration of the impact is permanent (exceeds life of mine) in view of recovery to original community but recovery to a different community but providing similar ecosystem services may occur sooner. The intensity is moderate to serious but the extent is confined to the mine sites, with a maximum of 60 km$^2$ after 20 years of dredging.
*Mitigation* - Leave behind a residual sediment layer of at least 30 cm of the original deposit thickness to cover the clay footwall. Leave behind undredged areas to enable migration of mobile organisms from these areas.

*Impact* - The depth of the dredged area might change local near bottom hydrographical conditions and thus act as a trap for very fine material. This could lead to high decomposition rates and consequently anoxic conditions and $H_2S$ concentrations in the sediments.
*Significance* - Medium as the duration is permanent and intensity moderate to serious, but extent is restricted to the mine sites and large areas of the inner shelf are naturally subjected to anoxic conditions.
*Mitigation* - Leave behind a residual sediment layer of at least 30 cm of the original deposit thickness, which will reduce the depth of the dredged-out area.

Although the mitigation measures will facilitate the colonising of the newly exposed sediments, and may reduce the risk of large areas of the dredged sites becoming anoxic, the significance will remain medium after mitigation. This is due to the very long time scales anticipated for the disturbed biota to recover to its original status and the expected low infilling rates at this water depth. Functional recovery, defined as recovery to a community that provides similar ecosystem functions to those of the original community despite being different in composition, is, however, likely to occur sooner.

In general, the confidence level in the assessments is medium, as most of the impact evaluations are based on assumptions that are derived from publicly available literature data, and data directly from ML170 are very limited. A survey is therefore critical to confirm these assumptions. In the case that the initial survey data reveal a substantially different habitat to that discussed in the environmental description, the impacts will need to be re-assessed.

Only the risks associated with dredging in three distinct target areas within ML170 (SP-1 and SP-2 each being 22 x 8 km and SP-3 being 11 x 6 km in size) are evaluated in this study. Cognisance though has to be taken of other current or future projects in the region that could result in cumulative effects. At present, the only other anthropogenic disturbance (besides normal vessel traffic) to the benthic environment is that from demersal trawling, whereby the deeper areas (>200 m) of ML170 including SP-2 and SP-3 may have been impacted by bottom trawl-gear. Marine mining for diamonds is unlikely to occur as significant diamond resources have not been reported from as far north as ML170. At least at present, drilling operations for oil and/or gas in or near ML170 have not been announced.

There are several other phosphate EPLs on the Namibian shelf, of which NMP holds six while others are held by other companies such as LL Namibia Phosphates and Gecko Mining (www.chamberofmines.org.na). Exploration work in some EPLs has recently been conducted, and a marine mining licence for phosphate was granted to LL Namibia Phosphates in October 2010, but mining will not commence for a number of years. Nonetheless, should further extensive dredging for phosphate rock occur in the region, the size of the disturbed area could increase significantly. Depending on the scale, this could ultimately lead to unacceptably high cumulative impacts. This has to be taken into account for the decision-making with regard to this and any future mining application, but cannot be covered by this specialist study (or the EIA).

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