NMP EIA & EMPR AMENDMENTS

Revisions based on supplemental studies and scientific advances

Prepared for Namibian Marine Phosphates

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Prepared by Drs Robin Carter & Nina Steffani

1 Introduction

Following conclusion of the High Court proceedings in Namibia in June 2021, Namibian Marine Phosphate (Pty) Ltd (NMP) is required to re-submit an application for an environmental licence certificate. The new application needs to include updates to the Environmental Impact assessment (EIA) and Environmental management Plan (EMP) submitted in 2014 and updated 2016, based on new additional studies conducted by NMP in 2019/2020 and recent relevant research. The authors of this report have provided input to the EIA and EMP previously completed for NMP.

These additional studies in 2019/2020 were completed in compliance with the recommendations put forward in a report commissioned by and submitted to the Environmental Commissioner on 20 December 2018 by Dr R.C. Newell (United Kingdom) and Mr N. Muhapi (Namibia). on an "*Independent Review of Written Comments by Interested and Affected Parties (I&AP) in respect of an Environmental Impact Assessment (EIA) submitted by Namibian Marine Phosphate (Pty) Ltd. (NMP) for an Environmental Clearance Certificate (ECC)*"

Below we provide comment and amendments on elements of the EIA and linked EMPRs that refer in light of current knowledge including these additional studies. These assessments are focused on ore recovery by dredging in the SP-1 area in Mining License ML170.

2 Assessments

The assessments are partitioned into:

- Effects of disturbances to the seabed,
- Sediment plumes generated at the seabed and from dredger overflows at the surface,
- Toxicity risks, and
- Underwater sound.

Additionally, the potential for CO_2 fluxes has been assessed and is submitted separately to this document.

3 Effects of disturbances to the seabed in the Mine area

3.1 Mining activity: Exploration and environmental work with gravity cores, vibrocores, and grabs

Aspect(s) - Further exploration and environmental work will be conducted in the larger ML170.

Predicted Impacts (EIA) - The exploration and environmental work will remove the associated benthic biota. The total area disturbed by these tools, even after extensive exploration campaigns, will be very small (largest individual core/grab size is 1.5m^2. Recovery is predicted to be fast (short term) due to slumping of material from the sides and migration of benthic fauna from undisturbed adjacent areas. Significance rating is low.

Review Comments - No amendments.

Review implications on impact ratings - No amendments.

Review implications on EMPR - No amendments.

3.2 Mining activity: Phosphate ore recovery by dredging

3.2.1 Sediment removal

Aspect(s) - Removal of the upper 1-<2.5 m (possibly up to 3 m) of sediment by dredging with loss of benthos.

Predicted Impacts (EIA) - The removal of 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 can be expected to differ from the original superficial deposits, and sediment refill rates at this depth are likely to be slow. Colonising assemblages may differ from those present prior to the dredging activity. The loss of the benthic community is restricted to the dredged-out areas (maximum of 2.5-km^2 and average of 1.7 km^2 per annum), but the recovery to the original community is likely to take longer than the life of mine (permanent (~20 years life of mine)) or may even not be achieved in a meaningful timescale. Recovery to functionally similar communities that provide similar ecosystem services as the original communities might, however, occur sooner (Long term). Significance rating is medium.

Review Comments - A verification survey was conducted in 2013 to increase the knowledge on the benthos distribution (macrofauna and meiofauna) specific to the mining target area in SP-1 (26 stations) (Lwandle Technologies, 2014). In the macrofauna study, the survey firstly tested whether a 1000-, 500-, or 300- micron sieve should be used for future monitoring surveys and recommended the 500-micron sieve going forward. The following results are based on the combined 500- and 1000-micron fractions. A total of 36 taxa were recorded, strongly dominated by polychaetes (69%). Other phyla were very sparsely present. Overall, the average similarity in macrofaunal community structure among the samples was high at 71%. The most abundant species were an oligochaete Tubificidae spp. (33%) and the polychaete *Paraprionospio pinnata* (31%). Most species were also found in the 2010 baseline, with only six 'new' species recorded in the 2013 survey. A relationship between macrofauna distribution pattern and environmental variables (only copper contributed 15.9% to macrofauna variation) was not obvious. This does not suggest that trace metals and/or sediment properties are not shaping the macrobenthic communities. Instead, it is suggested that due to the minimal variation in macrofauna distribution and associated environmental conditions, a causal relationship between the two cannot be readily seen.

In the meiofauna study 135 nematode and 36 harpacticoid copepod taxa were recorded. Nematode species richness values of up to 42 species per sample were found with densities of up to 30 400 nematodes per litre sediment. These numbers are comparable to other offshore seabed sites at similar water depth. Analyses of community structure identified six robust, coherent clusters of communities comprising structurally related species assemblages. The community clusters were consistent with the presence of an apparent environmental gradient along the west-east axis of the sampling sites. This gradient in sediment granulometry, with finer sediments and silts occurring to the west of the survey area and overall coarser sediments, due to inclusions of shell fragments, to the east, was the principal determinant of meiofaunal community structures.

Dredging outside of port areas has had and does have widespread application globally, including *inter alia* mineral and aggregate recovery, landfill, seawall and beach maintenance. Effects of these dredging activities are well studied. However, recovery rates of benthic communities from dredging disturbance at water depths similar to the proposed SP1 mine area (190-225 m) in Namibia have not been reported. Potential recovery times can thus at best be inferred from other types of mining operations in the region, but in shallower depths, such as the extraction of alluvial diamonds from seabed sediments that is conducted in Namibian and South African waters in water depths extending to 150 m. 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. First colonisation of the newly exposed surface can start soon after cessation of dredging. The main pathway of colonisation is through settlement and recruitment processes from the plankton but can also take place by passive translocation of animals during storms or sediment sliding from nearby unaffected areas, and the active immigration of mobile species. Opportunistic species, characterised by being small, mobile, highly reproductive, and fast growing, are typically dominating the first successional stage(s). Long-lived species, however, need longer to re-establish the natural age and size structure of the population.

A recovery to pre-mining conditions is commonly defined as the recolonization of previously mined areas by marine faunal communities to the point that they can be considered to have an ecological function equivalent to those that exist in comparable undisturbed reference sites. This is deemed to be achieved when the communities have, after a number of years , reached a similarity to the undisturbed sites of at least 80% (MacDonald, L. and Erickson, W., 1994; Newell, R., Seiderer, L. and Hitchcock, D., 1998). This similarity is based on a combination of univariate (abundance, biomass, diversity, species richness) and multivariate (species composition based on abundance, biomass, and functional traits) indices. Such an approach has also been adopted by the Debmarine Namibia (DBMN) Marine Scientific Advisory Committee (MSAC) and is applied in their diamond mining monitoring program (Risk Based Solutions (RBS), 2021). Long-term benthic monitoring carried out by De Beers Marine (DBM) provides a good record of mining-related impacts and subsequent recovery of benthic marine habitats and associated macrofaunal communities. Monitoring in a diamond mining licence area on the South African side of the Orange River has been undertaken since 2003 and mining in this area occurred in 2007. The latest 2018 survey suggests that recovery at the mined sites is very close to complete and shows that recovery period seems to be in excess of 10 years (Risk-Based Solution (RBS), 2021). From the DMBN benthic monitoring studies conducted in the Namibian Mining Licence Area Atlantic 1 MLA, it can be gleaned that rates of recovery can vary. Impact sites have been mined in 2012, 2014, 2016, and 2018. The monitoring studies found that in terms of univariate measures (abundance, biomass, and species richness) some mined sites showed clear lasting impacts of mining, while others showed little evidence of mining related impacts, or had even reached pre-mining conditions although they had been mined within the last four years (Risk-Based Solutions (RBS), 2021). In terms of species composition and biological traits (multivariate analyses differences between mined and unmined sites were recorded for sites that had been recently mined. Recovery rates seem to be closely linked to sediment refill rates as those mined sites closer to the Orange River mouth showed earlier signs of recovery compared to those further away.

Another seabed mining activity proposed to be undertaken in the international waters managed under the auspices of the International Seabed Authority established by the United Nations (not for Namibian or South African continental shelf waters) is the mining of deep-sea polymetallic nodules from abyssal (>3 000 m depth) plains, primarily in the central eastern Pacific. To date there has been no commercial mining of such minerals and there is no clear consensus on best available mining techniques. However, there is a recent upsurge in interest in mining of polymetallic nodules spurred by the need for critical metals to support growing populations, urbanization, high-technology applications, and the development of a green-energy economy (Hein et al. 2020). The first commercial test mining for polymetallic nodules was carried out in 1970, followed by small-scale commercial test mining and scientific disturbance studies. A meta-analysis on these long-term datasets (up to 26 years) by Jones and co-authors (2017) assessed the impacts of these experiments on biological communities and found that impacts could be severe immediately after mining, with negative changes in density and diversity of most groups occurring. Almost all studies showed levels of recovery in faunal density and diversity, especially for meiofauna and mobile megafauna, often within one year. However, some of the investigated sites were to a degree still depauperate in most faunal groups assessed over >10-year timescales suggesting longer-term recovery periods from effects of mining across most taxonomic groups.

Review implications on impact ratings - The benthos specialist study predicted possible recovery periods of longer than life-of-mine (>20 years, permanent) but functional recovery may be achieved earlier. Results from the diamond mining monitoring studies suggest potential faster recovery rates but these could still be in excess of 10 years. It must be acknowledged though that diamond mining occurs at shallower depths of up to ~150 m compared to the proposed phosphate recoveries located in 190-225 m water depths. Further, different mining methods are employed as a patchwork of smaller areas is generally targeted. However, these can include lanes 2-3 m deep excavated by crawler, somewhat similar to the TSHD excavated 3 m wide 2.5-3 m deep dredging lanes in marine phosphate mining. In diamond mining diamonds are extracted on-board the mining vessel, which is stationary at the mining site. Oversized material and fines recovered from the seabed are immediately discarded and after screening, the rest of the sediment is also returned directly to the sea (but not all material settles necessarily into the mined-out areas). During phosphate mining all retained dredged material will be transported to shore for treatment and only the fines in the dredger overflow will settle in the dredged areas. Plume dispersion modelling study predicts that sediment re-deposition can attain 200->500 mm in the mined area over the 20-year-life of mine (HR Wallingford, 2020), which would provide some measure of refilling into the dredged areas and may facilitate recolonization. However, it is predicted that sediments in the dredged area are likely to change to a predominantly silty substrate from silty sand leading potentially to a different benthic community composition than at pre-dredge conditions. A conservative approach to predicted recovery times is therefore appropriate and the significance rating remains medium.

Review implications on EMPR - As per the recommendations in the benthic study, mitigation measures are to leave behind a residual sediment layer of at least 30 cm of the original deposit thickness to cover the clay footwall and leave undredged corridors adjacent to dredged areas. A small-scale physical disturbance experiment simulating mining of phosphate deposits at the Chatham Rise in New Zealand, indicates that undisturbed areas near mined areas, despite being subjected to some level of sedimentation, may act as a critical source of colonising fauna (Murray, 2021). This highlights the importance of leaving unmined patches of seabed adjacent to or within targeted areas, to aid the recovery of macrofaunal communities through migration of adult mobile organisms from these areas (see also (Powilleit, Kleine and Leuchs, 2006; McLaverty *et al.*, 2020). It is noted though that this favours those organisms that are capable of lateral movement and may thus, at least initially, lead to a different community composition. The main pathway of colonisation is through settlement and recruitment processes from the plankton.

The benthic monitoring programme should be reinforced with adequate reference sites to make before/after comparisons and recovery rate estimates robust. Results of predicted dredge plume fines sedimentation patterns in the hydrodynamic modelling study (HR Wallingford, 2020) are to be used in allocations of reference and impact sample sites. Given the relocation of reference sites and the fact that the initial baseline surveys were conducted in 2013 a pre-dredging baseline survey needs to be conducted. In this benthos sampling should focus on macrofauna but the sampling of the larger mobile epifauna and small commercially non-targeted fish species should be included. Regular macrofauna monitoring surveys of all selected sites should be carried out over the life of the project and if necessary, beyond to determine the time of recovery once dredging has ceased (see separate benthic monitoring programme for details on the survey design).

3.2.2 Changes to seabed topography

Aspect(s) - Seabed Topographic changes affect change local near bottom hydrographical conditions.

Predicted Impacts (EIA) - The dredging of the target area will probably result in an uneven slightly 'hummocked' surface. This may sufficiently alter seafloor topography to slow down/change currents over deeper tracks, thereby possibly acting as traps for finer sediments and particulate organic matter (POM). This could lead to local concentrations of decomposing organic matter and possible development of anoxic conditions and elevated H_2S concentrations in the affected areas. The impact should be limited to the deeper trenches of the dredged area (maximum of 2.5 km^2 per annum) but can be permanent (>20 years) as sediment refill rates are expected to be low. Localised effects are moderate to serious as anoxic conditions are deadly for most benthic communities, but large sulphur-oxidising bacteria can thrive under these conditions. Significance rating is medium

Review Comments - If POM settles into the deeper trenches due to altered current speed, it would be restricted to surficial layers of the sediment and not be buried at least for decades (unless it is being covered by fines settling from the hopper overflow plume). The POM is mainly refractory material imported from the inner continental shelf productive area and would be mostly deposited in hypoxic and/or oligoxic conditions allowing oxic remineralisation. Thus, generation of anoxic conditions post mining should be unusual. In addition, the plume dispersion modelling study predicts that sediment re-deposition rates can attain 200->500 mm in the Mined area over the 20-year-life of Mine (HR Wallingford, 2020) thereby mitigating the potential for altered current speed over time.

Review implications on impact ratings - No amendment.

Review implications on EMPR - As per the recommendations in the benthic study, leave behind a residual sediment layer of at least 30 cm. This in connection with the modelled sedimentation rates from the dredger overspill of 200->500 mm over the 20-year life of Mine will reduce the depth of the Mined-out area but may not prevent settling of POM.

3.2.3 Removal of surficial organisms

Aspect(s) - Dredging removes mats of large sulphur-oxidising bacteria from the sediment surface and from the upper layer.

Predicted Impacts (EIA) - The large sulphur bacteria are important in oxidising the toxic H_2S thereby reducing its diffusion into the water column. The removal of sulphur-oxidising bacteria is limited to the dredge area (34 km^2 for life of Mine (20 years)). The recovery of bacterial mats will likely be medium to long term as it depends on the development of anoxic conditions in the sediment and subsequent sufficient H_2S concentrations. Removal of the sediment also removes H_2S and fluxes from the dredge area are thus not expected unless there is sufficient particulate organic matter deposition and anoxic conditions are generated through anaerobic remineralisation. If this happens, the bacterial mats are likely to return. Significance rating is low as concentrations of large sulphur bacteria are assumed to be low or absent.

Review Comments - A study analysing 8 samples spread across the mining target area in SP-1 and 1 site each to the west and east of SP-1, detected several sulphur-oxidising bacterial species including *Thiobacillus thioooxidans*, *Thiobacillus ferrooxidans*, *Thiobacillus denitrificans* cluster and several *Acidothiobacillus* species (Kirby, 2014). Neither the giant *Thiomargarita namibiensis* nor *Beggiatoa* species were recorded. In general, sulphur reducing bacteria were slightly more common than sulphur oxidising bacteria, the latter ranging from <3800 to 8.9x10^5 per gram of soil.

Review implications on impact ratings - Although this report supports the conjecture made in the benthic study that mats of the large sulphur-oxidising bacteria (i.e., *Thiomargarita namibiensis* and *Beggiatoa*) in the mining target area may be low to absent, the study is based on a relatively small number of samples and conclusion should therefore be drawn with caution. The presence or absence of bacterial mats also do not signify the same for *Beggiatoa* as they can exist deep below the sediment surface (Priesler *et al.*, 2007). On the other hand, it does not provide a reliable reason to amend the significant rating.

Review implications on EMPR – The main focus is on demonstrating the existence and abundance of sulphur oxidising bacterial mats in the proposed dredging areas pre and post ore recovery operations. This is best achieved by seafloor imagery by ROV/AUV or sled-mounted cameras along transects across disturbed (impact) and control areas as the reflectivity of stored sulphur microgranules gives the bacteria and the surface of the sediment a white appearance, which can be seen on video/photos. Surveys at impact and reference areas should be done pre-dredging and then at sequential 3-year intervals until an apparently stable condition is attained according to metrics on patch numbers and seabed area covered. Such surveys will also apply to benthic epifauna monitoring.

3.2.4 Dissolved nutrient releases

Aspect(s) - Mobilisation of dissolved nutrients from the sediments due to dredging.

Predicted Impacts (EIA) - 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. The effects are local as the released nutrients will spread with the overflow plume; are expected to be very short term as the overflow plumes will only be generated during dredging which occurs within a 37-hour dredge cycle for approx. 16 hours; and have minor effects as dissolved nutrient concentrations in the target areas are expected to be relatively low. Significance rating is low.

Review Comments - The verification survey sampled surficial and underlying sediments up to a depth of ~2m at 26 stations across the mining target area in SP-1 (Lwandle, 2014). Average particulate organic matter (POM) concentrations in the survey area sediments were 7.4%, marginally higher than the upper level of 6.9% quoted in the EIA. C:N ratios in the POM were 11.4 in the surficial layers and higher at 19.8 in the underlying sediments. These ratios are indicative of refractory organic matter and are consistent with the predictions made in the EIA. Inorganic nutrient concentrations in the sediment pore waters averaged 156 µg/L for nitrogen (ammonia + nitrite + nitrate-nitrogen) and 209 µg/L for phosphorus. The latter is considerably enriched compared to the overlying water column and it is expected that there is a considerable departure from the classical Redfield ratio of 16 and that measured in the overlying water column (17.7). The phosphorus enrichment is attributed to be derived from the pelletal phosphate ore body. However, pore water volumes are low due to low sediment porosity, and therefore significant modifications to upper water column Redfield ratios through translocation of pore water to the surface are not expected to occur.

Review implications on impact ratings – No amendment to the significant rating.

Review implications on EMPR – Prior to commencement of dredging operations, to refresh the baseline refence data, pore water nutrient concentrations need to be confirmed on fresh sediment obtained by a large GOMEX box core (see Blomqvist et al, 2015). Pore water is to be extracted without exposure to air or overlying water with a pore water extracting device according to Cranford et al (2017) and samples stored chilled or frozen prior to analyses according to Przeslawski et al (2018).

3.2.5 Sulphide releases

Aspect(s) – Release of hydrogen sulphide from the sediments due to dredging.

Predicted Impacts (EIA) – The dredging may release hydrogen sulphide from the sediments, which will affect benthic communities. The impact is local as the released hydrogen sulphide may spread along the sea bottom affecting undredged areas and the associated biotic life; short-term as the spread of hydrogen sulphide across the seafloor will be very short term and the compound will eventually mix with the seawater; and with moderate effects as it is very toxic and can kill many animals in its path. 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, but it cannot 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. Significance rating is low due to assumed low hydrogen sulphide concentrations.

Review Comments – The verification survey sediment-sampling regime covered a total of 26 verification sites, which extended across SP-1 and approximately 1-2 km to the west and east of SP-1 (Lwandle Technologies, 2014). As a proxy for potential HS^- fluxes from the sediment, Acid Volatile Sulphides (AVS) and Oxidation-Reduction Potential (ORP) were measured from superficial sediments (Day grab) and from subsurface sediments down to 2.2m (gravity core). AVS was undetectable in the surficial sediments and there were very low AVS concentrations in the underlying sediments, which predicts low to zero flux of HS^- into the benthic boundary layer. Furthermore, elevated nitrate concentrations at the base of the water column and in the sediment pore-water supports the contention that HS^- flux is low as the two compounds cannot coexist.

Review implications on impact ratings – The verification survey in conjunction with biogeochemical inputs by Monteiro (in litt. In Lwandle Technologies, 2014) concluded that the proposed Mine area in SP-1 does not show the characteristics of the mud belt and the SO_4 reduction and therefore HS^- flux from the sediments would be minimal. This supports the conclusion of the EIA that environmental risks of mining by dredging would be physical as opposed to biogeochemical. No amendment to the significance rating.

Review implications on EMPR – Prior to commencement of dredging operations, baseline reference data should be refreshed. Pore water free sulphide concentrations need to be confirmed on fresh sediment obtained by a large GOMEX box core (see Blomqvist et al, 2015). Pore water is to be extracted from replicate sub-cores without exposure to air or overlying water with a pore water extracting device according to Cranford et al (2017) and analysed spectrophotometrically within a short period after collection (Cranford et al 2017, 2020).

3.2.6 Mobilisation of toxins

Aspect(s) – Exposure of potentially toxic sediments on the seabed by ore removal and transfers of potential toxicity to the upper water column by fine sediment overspill during ore recovery

Predicted Impacts (EIA) – In both cases exposed biota may suffer acute (lethal) of chronic (sub-lethal) effects with associated negative effects on ecological functionality.

The dredging may increase bioavailability of trace metals in the target ore body through promoting dissolution from the solid phase during turbulent mixing in dredging. Toxicity effects may be exerted on biota within the fine sediment plume path as it mixes with the host water body. Any deleterious effects are predicted to be limited to the dredge sediment plume, be short term as effects will be exerted within the lifetime of the plume, while the intensity of effects would be minor.

Review Comments - Measurements of Mine area sediment trace metal concentrations showed that two of the 14 metals analysed for exceeded recommended guideline concentrations while three, arsenic, cadmium, and nickel, were present above probable effect concentrations (Lwandle Technologies, 2014). Trace metal elutriation tests on sediment core samples showed that, on average, <1% of each of arsenic, cadmium, chromium, copper, nickel, and lead entered the dissolve phase and would be bioavailable. Factors limiting the dissolution of metals would have been the presence of AVS as, although present at low concentrations, it was in excess of the SEM fraction. Additionally other metal binding complexes such as iron and manganese hydroxides and particulate organic matter, would have prevented the dissolution of metals.

These measurements supported and added confidence to the EIA conclusions on toxicity (Lwandle Technologies, 2014).

Following recommendations by reviewers (Newell & Muhapi, 2018), toxicity tests measuring acute and chronic effects on sea urchin fertilisation success rates and larval development in elutriates from Mine area surficial sediments previously sampled by gravity corer were conducted. The methods applied followed those of the USA EPA (US EPA, 2000) and the results are reported in (Lwandle Technologies Pty Ltd, 2020). The elutriation tests showed similar dissolved trace metal concentrations to those determined on fresher core material collected in the Verification Study (above). In both cases the proportion of metals entering the dissolved phase and therefore being bioavailable was low; average values on both sets of material were <1% of the sediment concentrations and were below water quality guideline thresholds for the region (CSIR, 2006).

Consistent with the low trace metal concentrations liberated by elutriation from surficial sediments the toxicity testing conducted did not show any acute level toxicity in sea urchin fertilisation success or the chronic level toxicity larval development test measured after 72 hours of exposure to elutriates. The respective performance levels as a percentage of the controls were 97.58% fertilisation success and 94.28% non-deformation of larvae. Toxicity effect levels exceeding -10% of that in control tests are considered indicative of toxicity.

Pore water ammonia is also a source of toxicity effects in marine sediments (Batley and Simpson, 2009). The in-situ guideline concentration for total ammonia, comprising ionised and non-ionised forms, based on background concentrations in estuarine and marine sediments is 3.9 mg/L (Batley and Simpson 2009). Pore water total ammonia concentrations on the Namibian continental shelf offshore of the nearshore mud belt are reported as 3.6 mg/L (van der Plas, Monteiro, and Pascal, 2007). In the dredging process sediment pore water would be mixed with bottom water to form the slurry transporting dredged sediment to the surface. Mixing volumes are large and this will dilute the pore water derived total ammonia to < 1.55 mg/L (dilution factor of 2.3 x), the revised toxicity guideline concentration. (Batley and Simpson 2009).

In the suite of trace metals examined in the elutriation testing arsenic showed the highest solubility although this was very modest at <1% of the mass processed (Lwandle Technologies, 2014). Arsenic is an abundant element in the marine environment ranked 22nd in terms of abundance, primarily occurring in the inorganic forms of arsenite (As^{III}) and arsenate (As^{V}), the latter being the dominant form in oxic water and the former being more important in reduced, hypoxic/anoxic environments (Neff, 1997). The average sediment bound total arsenic concentration determined in the verification survey (Lwandle Technologies,2014) was 49 µg/g whilst that in the water column was 3.21 µg/L. These are close to the estimated world average in marine sediments of 40 µg/g and 3 µg/L in coastal seawater reported in Neff (1997).

The pathway of arsenic from the dissolved phase into higher trophic levels in the euphotic zone of the water column is predominantly via uptake and production of organoarsenicals in phytoplankton, and transfers to zooplankton and their predation by fish. Phytoplankton take up arsenate, and within their cells reduce this to arsenite, with biotransformation through methylation to organoarsenicals, primarily dimethylarsinous and monoethylarsinous acids. Arsenite and organoarsenicals are readily excreted back to the water column when phosphorous is non-limiting for phytoplankton growth, as is generally the situation in the Benguela Current system, where the arsenite is rapidly oxidised to arsenate (Neff, 19978; Rahman et al, 2012).

The minimal dissolution of arsenic from marine sediments during dredging as predicted from elutriation testing is unlikely to materially affect apparent ambient concentrations and bioaccumulation from these.

Review implications on impact ratings - The trace metal elutriation measurements conducted in the toxicity study are consistent with those of the 2014 Verification study. The toxicity test results indicate no acute effects, consistent with the apparent low solubility of metals and, for divalent metals, the minor excess of AVS over SEM (ratio <1). The conducted toxicity measurements support the impact assessments of trace-metal toxicity in the sea surface layers and at the seabed due to disturbances during dredging for phosphate ore. Further, possible toxicity effects of pore water total ammonia should be unlikely due to high dilutions in the dredging process.

No amendments to the significance rating are justifiable.

Review implications on EMPR – As part of updated baseline measurements of sediment properties, above, toxicity tests need to be conducted on fresh sediments collected by box core. The sampling method needs to ensure that surficial sediments are retained in the samples submitted for toxicity testing. It is recommended that a large GOMEX box core is employed for this due to its ability to achieve this when properly deployed (see Blomqvist et al, 2015).

3.2.7 Proliferation of human health hazardous bacterium

Aspect(s)-Proliferation of the anaerobic bacterium Clostridium botulinum type E in the dredged area.

Predicted Impacts (EIA) - 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. The impact is restricted to the Mine site (maximum of 2.5 km^2 per annum, 1.7 km^2 on average) and 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, and the effects of botulism are serious (botulism caused by the bacteria can be lethal to human and wildlife). However, in-situ contamination of fish populations by the bacterium has not been reported for southern African fish populations, and literature data suggest that the distribution of the bacteria is generally limited in deeper saline waters. Significance rating is low as proliferation of bacteria is assumed to be a rare probability.

Review Comments - An extensive literature search could not find any reported evidence of in-situ outbreaks of *C. botulinum* contamination in southern African water and/or fish populations.

Review implications on impact ratings - No amendments

Review implications on EMPR - No amendments

3.3 Seabed plumes generated at the dredge head

Aspect(s) - Bottom sediment plumes generated by the dredge head.

Predicted Impacts (EIA) - High suspended sediment concentrations near the sea bed generated by the drag head and subsequent re-deposition of the material causes smothering effects on the benthos. This impact is very localized and short term, and effects will only be relevant along a narrow strip around the outer edge of the dredge site since any re-deposition inside the dredged area will have no impact as the animals are removed. Significance rating is low.

Review Comments - Predictability of the dispersion of bottom sediment plumes induced by dredging activities in the deep sea is still very limited due to limitations on in-situ observations required for a thorough validation and calibration of numerical models. Suspended sediment concentrations generated at the point of dredging tend to decline as the sediments become coarser and usually decrease rapidly with distance from the dredger. The size fractions of greatest consequence are the silts, muds and clays (<63 µm) as these create the highest level of turbidity. The sediments in the target area are dominated by medium to fine sand and the mud fraction (<63 micron) contributes only 30% to surficial sediment composition (Lwandle Technologies, 2014). HR Wallingford (2020) reviewed draghead generated turbidity plume measurements for a similarly sized draghead to that proposed to be used in the phosphate ore recovery. In silty sands with ~20% fines release rates of 1 kg/second were recorded. Modelling based on this indicated suspended sediment concentrations of 1 mg/L and confinement of these to the lower 10 m of the water column and within a few hundred metres of the dredging, i.e., within the dredge area. This is supported by a recent in-situ plume dispersion experiment for the exploration of polymetallic nodules in 4 200 m water depth that recorded sediment re-deposition rates of up to 9 mm directly next to the dredging tracks, reducing to 0.07 mm about 320 m away from the dredging centre (Purkiani, K., Gillard, B., Paul, A., 2021). Previous similar but larger-scaled deep sea (>4 000 m) resuspension experiments using numerical modelling recorded re-depositions ranging from 1 to 12 mm in an area of about 2 km^2 (Jones et al., 2017). This reinforces the conclusion of the EIA that any effects of the draghead will be limited and mostly confined to the dredging area.

Lwandle (2014) reported on seabed current velocities and turbidity as measured over two periods during the verification survey. Turbidity was typically low (median values of 1.01 NTU and 3.51 NTU) but episodic short-term elevations (hours to <2 days) in turbidity occurred with values more than 350 NTU and a maximum value of 1 300 NTU. There was no clear association between the recorded turbidity events and current speed and direction (accelerations or switching events), which supports the notion that the observed turbidity events are derived from the farfield, e.g., the nearer shore mud belt, and are advected past the survey area towards a depocenter of organic matter on the upper continental slope (Inthorn, M., Morholz, V., Zabel, 2006).

Review implications on impact ratings - Sediment re-deposition from the draghead is expected to be very localized. The mining area is subjected to natural turbidity events, which seem to pass over the mining area without settling due to high bottom shear stress, and the addition of fine particles suspended by the draghead to these is negligible. Accordingly, no amendment to the significance rating is warranted

Review implications on EMPR – The environmental baseline needs updating due the now > 7-year time lapse since its establishment. This should be conducted after licence award in the 24–36-month period prior to commencement of dredging to allow development of the onshore component of the project. The specified sample station distributions in the EMPR need to be adjusted such that the HR Wallingford predicted ZOI sediment deposition patterns are accounted for, especially for the reference sites.

3.4 Water column sediment plumes generated by hopper overflows

Aspect(s) - Sediment plumes generated by Dredger hopper overspill

Predicted Impacts (EIA) - Dredging generates plumes of suspended sediments that can adversely affect organisms in the water column. The set effect threshold was 20 mg/L total suspended solids (TSS) extending >3 days. The predicted extent was the dredge area (<6.6 Ha) for each dredge cycle, existing over the short term (<3 days) with a low intensity and a low significance.

Review Comments - Suspended sediment thresholds for effects on a wide range of marine organisms including phytoplankton, zooplankton, crustacea, molluscs and fish have been reviewed by Smit et al (2008). Effects included survival, feeding behaviour, growth, mobility, reproduction, oxygen consumption and effects on the gastrointestinal tract. Median effect concentrations (EC_{50}) for mortality – this end point is less subject to error than chronic effects to a large extent- were compiled in species sensitivity distributions (SSD) for barite and bentonite. The respective 50% and 5% hazardous concentrations (HC) for barite were 3 010 mg/L and 17.9 mg/L. The corresponding figures for bentonite were 1 830 mg/L and 7.6 mg/L. Following a precautionary approach, the HC_5 for bentonite should be applied in assessing possible impacts of sediment plumes. The data used for the compilation of the SSD were taken from standard toxicity tests with 72- or 96-hour exposures. Thus, similar exposure durations in the field can be applied in determining risks posed by overspill sediment plumes.

Sediment plume behaviour put forward in the EIA was based on available data on regional currents and measured and modelled marine diamond mining discharge plumes in ~100 m water depth. Time series current measurements conducted as part of the Verification Study(Lwandle Technologies, 2014) provided details of the current regime in the Mine license area. The regional current regime data are consistent with these adding confidence to the EIA plume behaviour predictions.

Following recommendations by reviewers (Newell & Muhapi, 2018), HR Wallingford conducted simulation modelling(HR Wallingford, 2020) of dredge plume behaviour in the Mine area based on the existing metocean knowledge plus measurements on sediment properties from the mine site and technical details on proposed dredging programme, rates and equipment provided by the dredging contractor. Process design scenarios modelled included operations with and without an environmental (green) valve. This reduces air bubbles from the overflow discharge and therefore plume buoyancy, dramatically reducing the extent and sediment concentration in the surface turbidity plume. Consequently, more of the discharge plume behaves in dynamic mode to about 50 m depth and then enters the passive mixing mode. This minimises plume dimensions and effects in the upper water column such as light attenuation affecting phytoplankton photosynthesis. In the passive mixing phase, the sediment plumes are advected by ambient currents and dilute through mixing and sedimentation. The bulk of the particles in the sediment plumes will be fine silts and clays with particle diameters of 20 µm or less. Sedimentation rates of these will be enhanced by particle flocculation due to electrostatic forces to 1-3 mm/s. HR Wallingford applied 1 mm/s in the modelling which is conservative.

Based on the acute effect threshold (HC_5), above HR Wallingford defined a 20-year suspended sediment plume zone of influence (ZOI) that encompassed the SP-1 Mine area and extended 25 km north, 9 km east, 17 km south and 3 km west of its borders. The ZOI is the area within which a suspended sediment concentration above 7.6 mg/L may occur at any location within the water column at any time over the modelled period. For actual dredging events the modelling indicates that plume dimensions are considerably smaller falling into the range of 1-5km^2 compared to the ZOI 546 km^2.In terms of mean concentration increases across the model domain these are limited to <5 mg/l except for within 100 m of the dredger where the mean increase can be up to 18 mg/L. HR Wallingford modelled durations of exposures to elevated suspended sediment concentrations above the HC_5 threshold at locations 2 km distant from the border of the 20-year mining area.

These showed no exceedances for the surface and maximum exceedance durations of 15 hours in the mid- and bottom depth layers.

Review implications on impact ratings - Employing a stricter suspended sediment concentration limit the modelling indicates that the spatial extent of exceedances may extend into the specific Mine area, in this case SP-1. Based on modelled plume behaviour durations of exceedances that may possibly exert deleterious effects on water column biota would be limited to mostly within the borders of 20-year Mine area. However, here maximum exposure durations would be 30 hours, well within the usual toxicity test durations indicating that such effects should be unlikely. The impact significance rating remains low.

Review implications on EMPR – The requirement for predictive sediment plume modelling in the EMPR has been met (HR Wallingford, 2021). Given the predicted scales and intensities of the modelled plumes the requirement for operational modelling is moot.

The environmental baseline needs updating due the now > 7-year time lapse since its establishment. This should be conducted after licence award in the 24–36-month period prior to commencement of dredging to allow development of the onshore component of the project. The specified sample station distributions in the EMPR need to be adjusted such that the HR Wallingford predicted sediment plume distributions ZOI are accounted for, especially for the reference sites. To diminish turbidity effects in the upper water column, ensure that the environmental valve is employed where and when possible in the dredging programme.

3.5 Sedimentation from the dredger overspill plumes.

Aspect(s) - Re-deposition of particles from the TSHD overflow plume.

Predicted Impacts (EIA) - The re-deposition of fine material from the overflow plume causes smothering of benthic organisms, with a local extent. A minor fraction of the deposited material may be resuspended by bottom mixed layer turbulence and incorporated into the nepheloid layer with a possibly wider deposition footprint. As the nepheloid layer is reported to be extensive on the Namibia continental shelf (Inthorn *et al* 2006) the contribution of resuspended material from overspill plume deposition will be an infinitesimally small proportion of the mass of the fine particulates transported off the continental shelf. Consequently, any far field deposition effects on benthos are predicted to be similarly small. For an individual dredge cycle (58.5 hours being 16 hrs active dredging and remainder of time dredger is offsite), the impact is short term. Benthic biota in the immediate vicinity of the dredge area may be affected by smothering, elsewhere sedimentation rates are expected to be very low. Significance rating is low.

Review Comments - At the time of the 2012 EIA, plume dispersion models specific to this project were not available. A sediment plume dispersion modelling study was therefore recently commissioned (HR Wallingford, 2020) and the results thereof are briefly summarized here.

According to the model, the Zone of Influence (ZOI) from the overflow plume for the 20-year mining period extends over an area of 513 km^2 outside of the mining target area in SP-1. The total area outside the mining area affected by a re-deposition of >5 mm is 412 km^2, whereby 151 km^2 are predicted to experience an overall deposition thickness of >10 cm. The modelling indicated that this may result in a change to a siltier substrate in this area. These ZOIs for plume dispersion and re-deposition are calculated taking the entire life of Mine (20 years) into consideration (cumulative effect). The plume dispersion from ~16 hours of hopper overflow during a single dredging cycle (58.5 hours1), on the other hand, extends 1-5 km^2 outside the mining plan area, while the corresponding re-deposition of sediment is predicted to be ≤0.3 mm from a discharge volume of ~15 000 m^3 of fine material.

The effect of burial by sediments on benthic animals depends on several factors including thickness of deposits, rate of deposition, nature (grain size) of the deposits, and species-specific traits (e.g., mobile or sessile biota). Species with burying behaviour may experience little or no effect, while sedentary animals are

[1] The average 58.5 hour cycle time for a 2.5 km dredge path used in the modelling is an adjusted figure provided by the dredging company Jan de Nul (JDN) taking into account maintenance and downtime.

more sensitive. The development of marine species sensitivity distributions (SSDs) for the burial by sediment based on effect levels published for 32 species (molluscs, crustaceans and polychaetes) determined a 50% hazardous level (HL_{50}, i.e., 50% of animals are negatively affected by the burial) of 54 mm and a 5% hazardous level (HL_5) of 6.3 mm (Smit, et al., 2008). This is a conservative estimate as these levels were based on instantaneous and complete burial, i.e., a 54-mm deposition of sediment in one single event. If deposition is slow or occurs in pulses (as is the case for the proposed project, modelled deposition rates being ≤0.3 mm per dredging cycle), (non-sessile) species have time to escape burial and can move upwards with a rate equal to the deposition rate (Bolam, 2011). Murray (2021) studied the indirect effects of sedimentation on macroinfauna from experimental phosphate mining conducted in 2019 and 2020 on the crest of the Chatham Rise, New Zealand, at depths of ~450 m. The experiment allowed a comparison between high-disturbed (physically disturbed and subjected to a high level of sedimentation) and low-disturbed (not physically disturbed but subjected to relatively lower levels of sedimentation) areas of seabed to test how sedimentation may affect deep-water macrofaunal communities in-situ. Sampling was carried out before, immediately after and one year after disturbance. Community structure changed in both high and low disturbed areas after the experimental disturbance event with reduced densities particularly in the high disturbed area, but the macrofaunal communities showed an ability to recover from the sedimentation impacts across both disturbance treatments as most community parameters returned to baseline levels after one year.

Review implications on impact ratings - According to the plume dispersion model, an area of 151 km^2 outside of the dredge area will have a total re-deposition rate of >10 cm, which would well trigger the SSD HL_{50} of 54 mm. However, this is an accumulative prediction for dredging activity for the entire 20 years of dredging, while a single operational cycle is predicted to result in a 0.3 mm deposition, well below the HL_5 (6.3 mm). Accordingly, no amendment to the significance rating is warranted.

Review implications on EMPR - The benthic baseline and monitoring survey programme needs to take into consideration the ZOI in positioning the reference sites outside the ZOI (see above).

3.6 Underwater sound generation

Aspect(s) - Underwater noise generated by dredger operations

Predicted Impacts (EIA) - Dredging operations result in displacement and/or redistribution of marine mammals and/or seabirds within the local extent at a minor intensity and, after mitigation, low significance.

Review Comments - Jan de Nul N.V., the NMP dredging service provider, provide details on sounds generated by operating TSHDs (of equivalent specification to the dredger intended for use in ML170) and measured and modelled attenuation(Jan De Nul N.V., 2020) . Sound (SPL) source levels for an operating TSHD in the Jan De Nul fleet are 180-190 dB re 1 µPa at 1m, dominant sources are main engine (500 Hz) and propeller (300 Hz). Higher frequency noises can be produced by hopper loading but these attenuate rapidly in the sea compared to noises <1 kHz. Measurements around an operating TSHD show that produced sound attenuates to 150 dB within 100 m distance from the dredger. Modelled sound attenuation predicts that attenuation to 100 dB re 1 µPa at 1 m will be attained at an average range of 15 km.

Sound receptors in the operations area will be mainly cetaceans, seals, and fish. Temporary (hearing) threshold shift (TTS) in cetaceans and seals are reported as being 175 dB re 1µPa at 1 m SPL received level and above (Southall *et al.*, 2019). Mortality can be caused in fishes at SPL >207 dB re 1µPa at 1m for fish with swim bladders and >213 for fish without bladders, TTS thresholds are ≥186 dB; mortality in fish eggs and larvae can occur after exposure to 207 dB (Popper and Hawkins, 2019). Mortality or potentially mortal injury to sea turtles can follow exposures to similarly high SPLs (Popper and Hawkins, 2019). Given the dredger sound source level (above) such effects are unlikely.

Received sound level thresholds causing moderate behavioural shifts for baleen and odontocete cetaceans and seals range from 130 to 180 dB re 1µPa at 1 m (Southall *et al.*, 2007). Modelled sound attenuation for the TSHD *Gerardus Mercator* provided by Jan de Nul (Jan De Nul N.V., 2020) indicate that received sound levels >130 dB will be restricted to within a radius of 2-3 km from the operating dredger.

Predictions of fish behaviour sound level thresholds are confounded to an extent by varying hearing modes in fish (pressure or particle motion, presence/absence of air bladders) and varying experimental protocols (Popper and Hawkins, 2019). No estimates are provided here.

Review implications on impact ratings - The extent of the impacts of underwater noise generated in phosphate ore recovery by dredging extends to the specific Mine site as opposed to the local extend in the EIA.

Review implications on EMPR – Section 2.2.3 of the EMP describes monitoring procedures for sound. The *ad hoc* approach needs to be formalised such that noise attenuation at stand-off distances is measured by hydrophone deployments. Distances can be 500 m, 1 000 m, 2 500 m, 5 000 m and 10 000 m. During these measurements 'source' levels at the operating dredger need to be measured simultaneously. The purpose is to confirm measured and modelled attenuation rates supplied by the dredging contractor. This can only be carried out once the dredger is actively operating onsite in ML170.

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Environmental Management Plan

DREDGE AREA SEDIMENT PROPERTIES: BASELINE UPDATE AND MONITORING DURING DREDGING

Prepared for Namibian Marine Phosphates

7 March 2022

Prepared by Dr Robin Carter

1 Need for Sediment Properties Monitoring

Namibian Marine Phosphate plans to extract pelletal phosphate ore from the SP-1 dredging target area within the Mining Licence Area ML 170. Environmental baseline data on this site have been reported on in the 2012 EIA for the planned dredging operations and the associated Verification Programme, the latter being completed in 2014. The phosphate ore is to be recovered by dredging with a trailer suction hopper dredger (TSHD). This will disrupt and physically modify sea floor sediments and possibly affect the sediment biogeochemistry and associated biological processes and communities. This is recognised in the Environmental Management Plan (EMP 2012, EMP 2014 and EMP 2016) for the proposed dredging which requires the assessment of the scale and extent of these effects during dredging operations.

2 Monitoring Design

2.1 Monitoring Objective

The monitoring will comprise a baseline survey into sediment properties and associated biological communities to update that established in the 2014 Verification Programme to be conducted prior to the commencement of the dredging operations. The survey needs to be conducted in the 12-24-month period following granting of the EC during the development of the on-land facilities for ore processing. This will be followed by surveys in the 1st, 4th, 7th, 10th and 13th years of dredging. After the 1st three cycles of monitoring the interval between surveys can be revised according to the results obtained. Benthic macrofauna monitoring (see below) will be carried out in conjunction with these surveys. The monitoring objective will be to assess the degree of change and its progression against the updated environmental baseline. The monitoring target will be the surficial sediments as the major effects are expected to be exerted on these through, e.g., sedimentation and smothering effects from sediments suspended at the dredge head and the rapidly sinking coarser fractions of the sediments in the lean water discharges from the TSHD hoppers during dredging.

2.2 Monitoring Survey Approach and Variables

The monitoring survey is designed to build on the data and information acquired for the baseline characterization in the verification surveys, conducted in June 2013, but focused on the more localized target dredging area of the initial three-year period of dredging. Surficial sediment samples will be obtained by a modified GMX-50 GOMEX box corer XL (added weights and enhanced leverage on closing mechanism). This corer samples 0.25 m^2 seabed area to a depth of ~30 cm. The modifications are intended to enable penetration and sampling of dense shell layers that, as shown in the Verification Survey, occur in the region. Sediment properties will be analyses on sub-cores taken from the box core samples.

The monitoring variables analysed for will include:

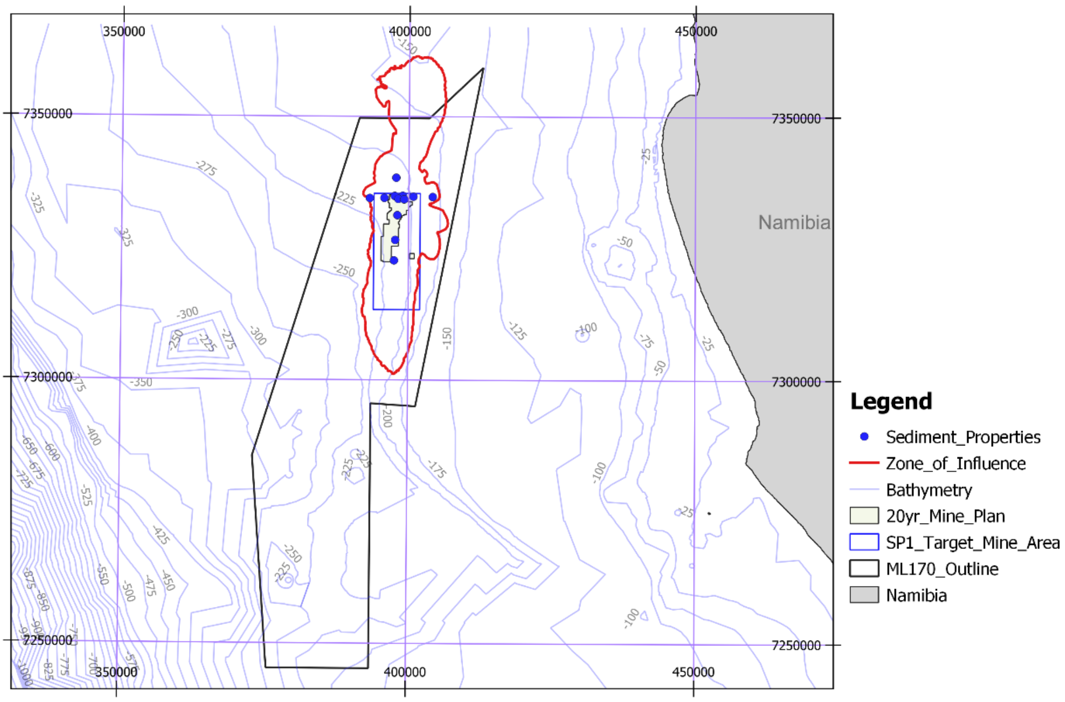
- Sediment texture and particle size;
- Sediment porosity;
- Sediment particulate organic matter (POM), particulate organic carbon (POC), total nitrogen and total phosphorus;
- Trace metal concentrations, acid volatile sulphide AVS and simultaneously extracted metals (SEM), Pore water free sulphide concentrations
- Sediment pore water toxicity[1], and
- Macrofauna[2].

The presence and abundance of thiobacteria mats in the mine area and, if present, their recovery post dredging disturbance will be photographically monitored in conjunction with the above surveys by ROV or AUV. Reference areas will be included in this.

The data obtained will be analysed to determine changes from the baseline condition and to confirm spatial variations in the individual variables. This will include assessments of changes in risk factors for biological communities inhabiting the sediments and biogeochemical processes linked to sulphide dynamics.

3 Survey Area (schematic)

The surveys will be conducted in and around the initial dredging area in the SP-1 target area (Figure 1). Sample sites will comprise impact and reference locations distributed in a cross- and long-shore oriented grid based on the simulation modelling (HR Wallingford 2020) predicted sediment deposition zone of influence (ZOI) and those established in the Verification Surveys. The reference sites are those for benthos monitoring located north of the predicted ZOI as there are currently no seabed data south of the SP-1 20-year Mine area. The benthos impact sites are included in the sample set in the north of the 20-year mine plan area.



[1] Baseline survey only

[2] Methods for this described by Dr N. Steffani in a separate document.

Figure 1: Distribution of sediment sampling sites (filled blue circles) for baseline and monitoring surveys in the SP-1 dredging target area in the ML 170 mining licence area. The red lines show boundaries of the ZOI from simulation modelling (HR Wallingford 2020).

4 References

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NMP EIA & EMPR AMENDMENTS

Revisions based on supplemental studies and scientific advances

Prepared for Namibian Marine Phosphates

7 March 2022

Prepared by Drs Robin Carter & Nina Steffani

Environmental Management Plan

Benthic Macrofauna Monitoring Programme

Compiled by Dr Nina Steffani

1 Need for Benthic Macrofauna Monitoring

A benthic monitoring programme needs to be established, whose principal objective is to study the rate of recovery of disturbed benthic communities once the mining activity has ceased in a particular mining block. Recovery has been shown to be both spatially and temporally variable, and to confidently measure the ecological recovery rate of mined areas, it is therefore necessary to develop a benthic monitoring plan that is not only appropriate in the medium-term (~5 years) but has the flexibility and potential to be extended into the long-term.

2 Approach

The key questions for the monitoring programme are the following:

- What is the rate of recovery of the physical environment, e.g., in-filling of mined-out areas with unconsolidated sediments?
- What is the process of the recovery of the benthic communities?
- How long after the disturbance does it take for the benthos to recover to at least an ecologically functional community?
- How does the physical environment (e.g., sediment particle size, organic matter, dissolved oxygen) influence the recovery rate?

In identifying and assessing the impacts of dredging of phosphatic sediments on the benthic communities, it is important to recognize that the marine environment can be very variable both in space and time. An impact should therefore be distinguished as being the relative difference between changes at a disturbed/impact site compared with changes that have occurred in a similar undisturbed reference site. In other words, there must be some change from before to after a disturbance and such change must be importantly different from what occurred in undisturbed reference areas. Community parameters, however, vary both spatially as well as with time, fluctuating in response to natural variations in the environment (these may be monthly, seasonal, or annual variations). Without adequate indices of natural variability, it will be inherently difficult to place dredging-related impacts in context. It is therefore important to have a number of impact sites in association with a number of reference sites that are in a similar environment (e.g., depth and sediment texture) but will remain undisturbed over the period of the monitoring programme.

The baseline and environmental impact assessment (EIA) verification survey in 2013 sampled a grid pattern of 15 sites within the target area in SP-1 (Figure 1). The results have shown that the macrofaunal assemblages and sedimentological characteristics are fairly homogenous across the target area. For operations in SP-1, a first pre-dredging baseline was conducted in 2013. Four impact stations and four associated reference sites to the north and four to the south of the 20-year mining areas were selected (see Figure 1). However, a plume dispersal and sediment deposition model has shown that over the lifetime of the mine (20-years of dredging) the selected reference sites are likely to be affected by sediment re-deposition (Figure 2, HR Wallingford 2020). In any case, a new baseline survey needs to be done as with increased elapsed time between baseline sampling and the onset of dredging (>1 year), the baseline data are likely to become less reliable for the purpose of subsequent monitoring and assessment (Ware & Kenny, 2011). Moreover, the risk of falsely attributing any natural changes in the benthos to the impacts of dredging increases with increasing time between the baseline survey and the onset of dredging.

Three target areas have been identified in the licence area, i.e., Sandpiper-1 (SP-1), 2 (SP-2), and 3 (SP-3). The northern centre portion of SP-1, with water depth ranging from ~200-225 m, has been identified as the initial target area with a 20-year mine plan for the issued mining licence. The benthic monitoring programme focusses therefore on this target area in SP-1. For the purpose of baseline and subsequent monitoring sampling, a robust stratified survey design will be applied, which aims to achieve an adequate and balanced density of sampling within the predicted impact zone along with an adequate density of sampling within adjacent reference (un-impacted) locations with comparable environmental conditions (Ware & Kenny 2011). The impact sites have been selected according to the currently proposed mine schedule to fall into the mine blocks that will be mined in Year 1 and Year 2 of the schedule (Figure 3). This ensures that any information on recovery process can be collected as early as possible to inform the Environmental Management Programme. The reference sites were chosen in relation to the predicted re-deposition rates (see Figure 2) to ensure that they remain unaffected by dredging activities over the 20-year mining. The positions of the impact and reference sites depicted in Figure 3 are schematic only and the final positioning should be decided on in collaboration with NMP's mine planners before the baseline survey commences. To increase the power of detecting disturbance effects, it is recommended to have a greater number of control than impact sites (Underwood, 1994). While the primary focus of the benthic monitoring programme will be on the benthic macrofauna, the baseline survey should include sampling of the mobile epifauna and small fish to broaden the knowledge of the benthic communities in the mining area.

Sampling in SP-1 should be undertaken both before the start of operations (i.e., the baseline), as well as at regular intervals after completion of dredging of Target Blocks 1 and 2 to determine the (functional) recovery rates of the benthic communities. One of the basic assumptions of developing a benthic monitoring program is that recovery of disturbed macrofaunal communities does in fact occur. The process and time of recovery is, however, strongly dependent on the rate of the infilling of sediment in the mined-out areas, and the type of sediment. It is recommended that high-resolution geophysical surveys (e.g., side scan sonar and/or multibeam bathymetry) are conducted immediately after dredging is completed in the individual target blocks, and at 3-year intervals from the start of the dredging programme to determine the depth of the dredged trenches and the sediment infilling-rates. Depending on results the intervals can be extended to, e.g., 5 years if found to be appropriate. Benthic macrofauna community sampling should be conducted at similar intervals. The post-dredging sampling should best be carried out at the same time of year as the baseline survey was to avoid seasonal variation, and for logistical reasons should be aligned with other survey operations that are part of the overall monitoring programme. Periodically reviewing the monitoring plan as new data is collected and analysed will ensure that the monitoring plan and associated sampling schedule remain a dynamic process, and that it can be adjusted as circumstances dictate.

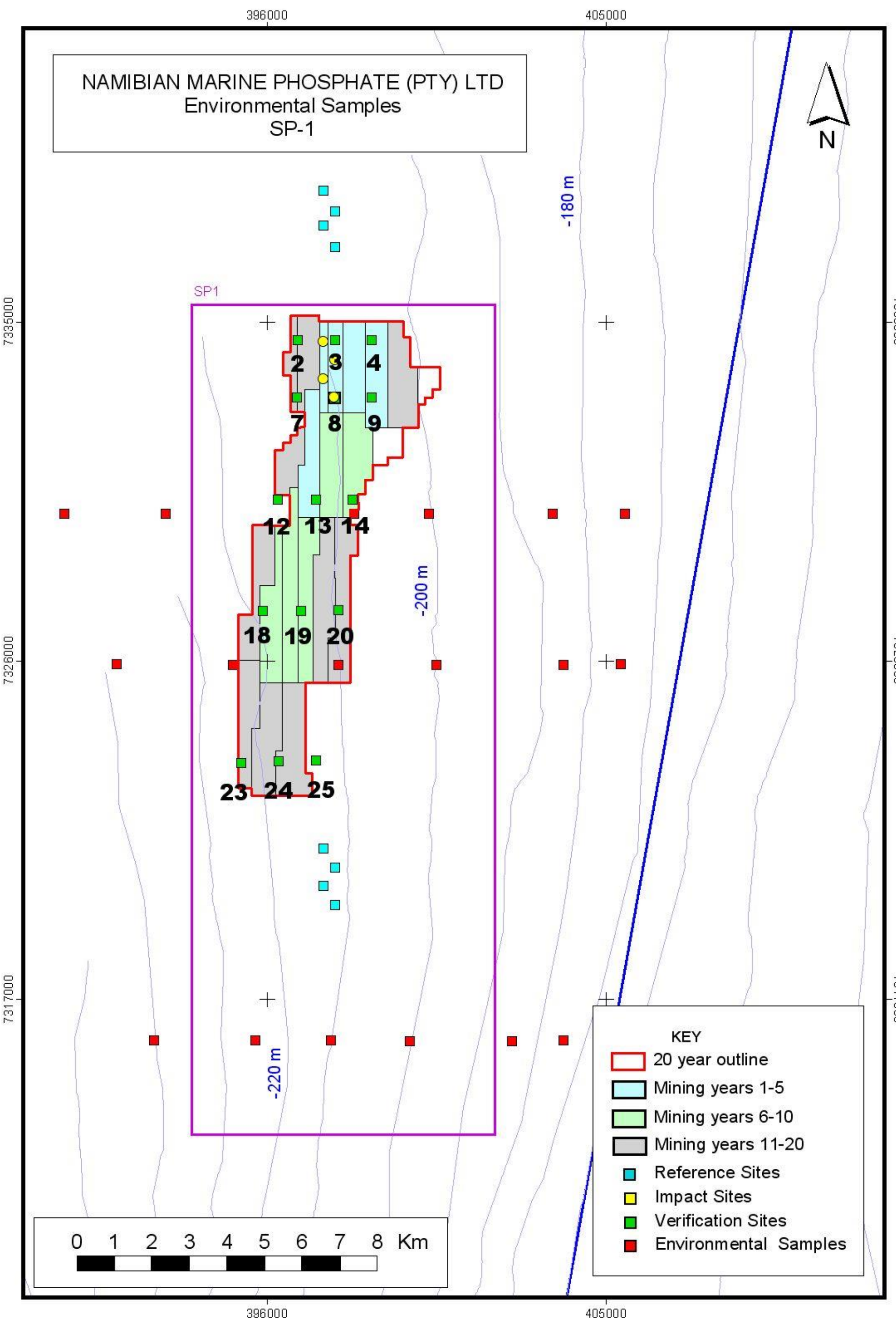

Figure 1: Map of SP-1 showing the locations of the four impact sites (yellow) and the four northern and southern reference sites (blue) sampled during the 2013 baseline survey. The red squares indicate the position of the sites sampled during the 2010 baseline survey, and the green squares those sampled during the verification survey in 2013.

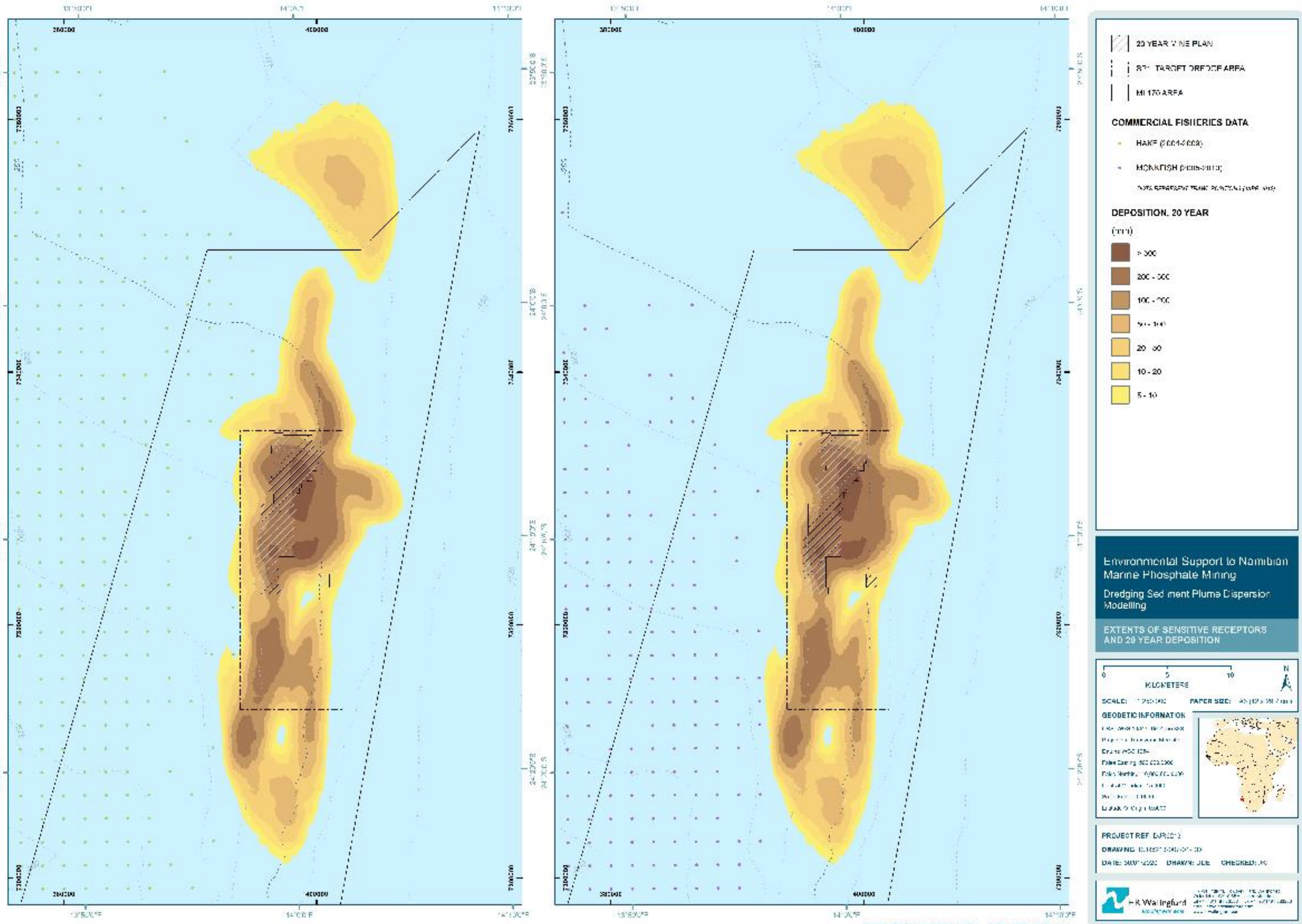

Figure 2: Dredging Sediment Plume Dispersion Modelling results showing the re-deposition rates over the 20-year mining of the target mining area in SP-1.

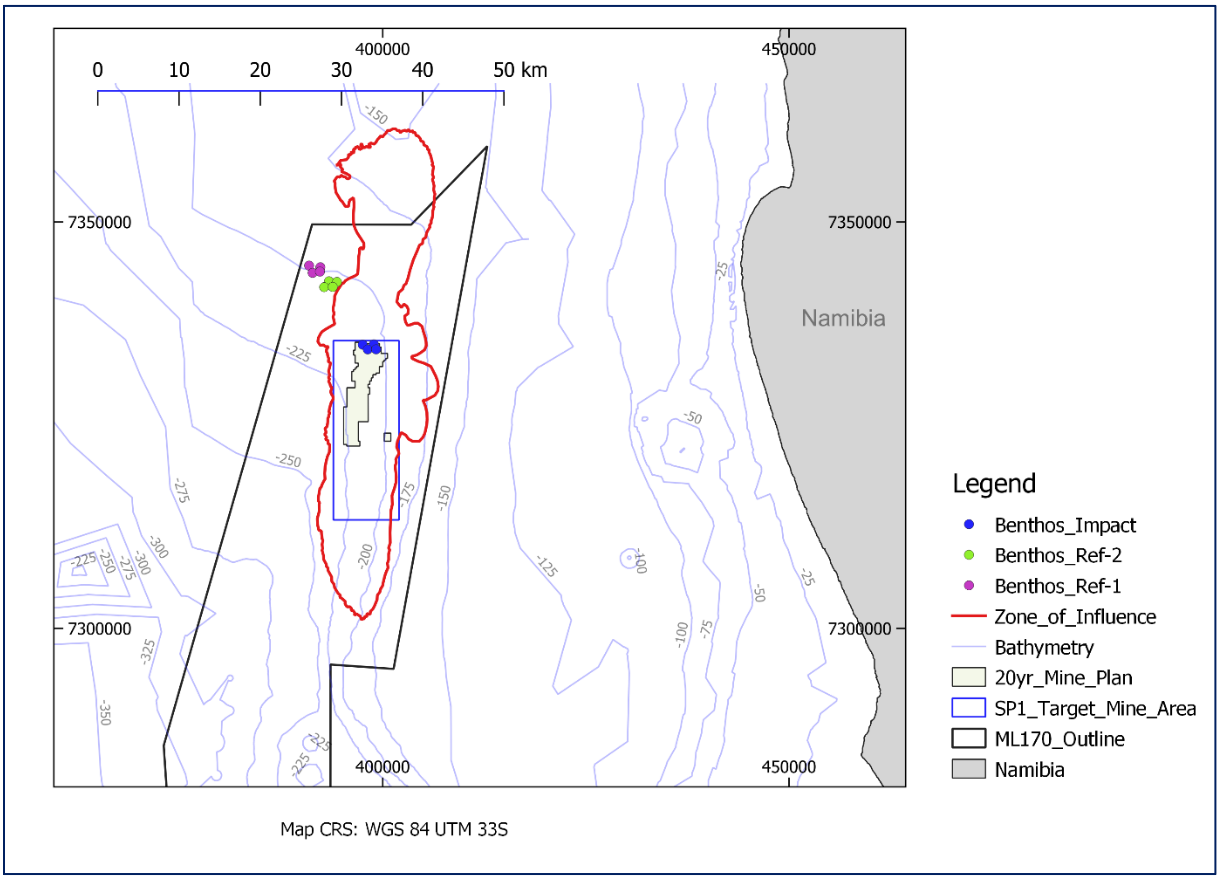

Figure 3: Map of SP-1 showing the provisional macrobenthos impact and reference sampling locations

3 Sampling Procedure

3.1 Macrobenthos

Sampling for macrofauna will involve the use of a box core (GOMEX GMX50 or similar design) with a Day grab for backup. Both can sample 0.25 m^2 surface area. Additional weighting and strengthened leverage on closing mechanisms will be used to counter difficulties in sampling dense shelly layers recorded in the verification survey. The review of the EIA and benthic sampling programme by Newell & Muhapi (2020) suggested the use of a Hamon grab as this is typically used for aggregate dredging operations of coarser sediments (i.e., with a significant gravel component) (Ware & Kenny 2011). However, many studies have proven that a box core and Day grab are appropriate for sampling of silty sand sediments (the sediment properties in the target area) and can reliably obtain quantitative samples to describe the macrofaunal assemblages (e.g., Blomqvist et al 2015, Bolam et al., 2008, 2011; Ware et al., 2010; Ware & Kenny, 2011: Bolam et al., 2021). These are the recommended sampling tools.

The sampling device, fitted with a USBL, is attached to a winch and is lowered down through the water column until ~10 m above the seabed (as determined by USBL) whereafter descent speed is reduced to avoid generation of bow waves as the device strikes the seabed. At this juncture the suspension cable slackens, and the weight stack drives the sample box into the seabed. Re-tensioning of the suspension cable closes the spades under the box and upward motion closes the covers at the top of the sample box avoiding loss of the surficial sediment layer. The sample device is then brought back onto the vessel. Samples below 40% volume should be regarded as failed. If this occurs, sampling is repeated at the same site until an adequately sized sample is collected. Following retrieval, the following procedures should be followed:

- The fullness of the box corer is estimated and the space remaining between the upper level of sediment and the top of the corer noted. This number can be used to convert the fullness of the corer into actual volume of sediment, using the dimensions of the box core.
- A photograph is then taken of the surface of the sediment within the box core. This photograph should include a detailed label (site number, date and where needed a replicate number) as well as a scale object (the label size should remain the same throughout the survey). The label is preferably written on white paper, which serves as a colour comparison between the different sediment samples.
- The colour of the sediment within the corer is noted and all other necessary information recorded on datasheets.
- Using a 50-mm plastic tube a sub-core of the sample is taken, which needs to be stored in a pre-labelled 375 ml jar and placed in the freezer. This sample will be used for analysis of organic substances. The same method is used to collect a 750 g subsample to be used in the particle size analysis. The sediment properties should be analysed according to the Wentworth scale. Organic matter and particle size samples should be taken in such a manner that a suitable surface area of sample is left undisturbed to retrieve the macrofauna sample.
- Sub-sampling of the macrofauna is done using plastic tubes with a 75-mm diameter. This method ensures that a controlled representative sample of the surface and the deeper layers of the sediment can be collected. Eight cores should be retrieved per box core and pooled, resulting in approximately 5 litre of sediment and sampling a surface area of 0.035 m^2.
- The sample is then sieved through a 500-micron sieve. To aid in the sieving process it is recommended to use a 3 mm or 5 mm screen above the 500-micron sieve to separate gravel and shell fragments from the actual macrofauna sample. A handheld hosepipe should be used to carefully wash the sediment from above.
- The sediment including the macrofauna retained on the 500-micron sieve is then transferred into a suitable sized jar and the sample fixed with 4-5% buffered formalin. Shell fragments and larger grains that are retained on the 3-mm sieve are carefully washed, and any fauna retained on the sieve handpicked from the mesh and added to the sample.

In the shoreside laboratories:

- To separate the macrofauna from the sediment, the samples are washed with freshwater through a 500-micron sieve, sorted under a stereo binocular microscope at 10-25 x magnification and all macrofauna extracted from the sample and transferred to a storage solution such as 1% Phenoxatol or alcohol. If needed, the sample may be stained with Rose Bengal to aid in the sorting.
- Specimens are identified to the lowest taxon possible and counted per taxa per sample.
- Wet biomass is estimated by blot-drying the specimens on absorbent tissue for a standard period of time and weights recorded per species per sample using an analytical balance.
- Taxa retained on the 500-micron screen that traditionally are considered to be meiofauna (e.g. nematodes, copepods, ostracods and foraminifera) may be counted if needed but should not be included in subsequent analyses.

3.2 Epifauna and demersal nekton

Sampling of the mobile epifauna and small fish should be done during the baseline surveys and is to be conducted by Capfish. To avoid disturbance to the macrofauna sampling stations, sampling should be done at a suitable distance (~100 m) from these stations. For the collection of mobile epifauna, the use of a 2-meter Cefas Beam trawl rigged with a fine mesh net is recommended (Ware & Kenny, 2011). The Beam Trawl is designed to sample at and just above the surface of the seabed. Its small size makes it easy to deploy and usually results in the collection of a manageable sample size. On each deployment, the trawl is towed over a set distance which will collect a sufficiently large sample to adequately characterise the resident epifaunal assemblage, but not so large that the sample is unmanageable. A first trial dredge during the baseline survey will inform on the desired distance. For further information on the design and methodology of the survey, refer to the epifauna and demersal nekton component of the EMPR compiled by Capfish.

4 Data Analyses

When assessing recovery following a disturbance event, the classic scientific approach of testing the null hypothesis that two populations are identical is not completely valid (Dixon & Garrett, 1992; McDonald & Erickson, 1994; Underwood 1996). The objective instead is to establish an endpoint at which recovery can be declared complete by testing for bioequivalence. The approach is to define two areas to be bioequivalent if, for example, the mean abundance/biomass of the benthic community at the impacted site exceeds a predefined percentage (say 80%) of the mean abundance/biomass at the undisturbed reference site for a defined time interval. Conversely, a site is said to be impacted or disturbed until the selected variable(s) exceed(s) the predefined level over a defined time interval. The number of successive intervals over which this value should be achieved, is necessarily site- and situation-specific. The predefined percentage is also site- or situation-specific, but the value of 80% has attained fairly wide acceptance (McDonald & Erickson, 1994; Underwood 1996). Bioequivalence tests should be supplemented with multivariate graphical and statistical tests (e.g., hierarchical cluster analysis, multidimensional scaling, PERMANOVA) for which no bioequivalent alternatives exist. This approach and definition of recovery aligns with that adopted by the Debmarine Namibia (DBMN) Marine Scientific Advisory Committee (MSAC), which also includes the incorporation of the functional traits analysis (see below) (Risk-Based Solution (RBS), 2021).

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. Measures used to assess recovery typically include biodiversity analysis such as the numbers of species and/or individuals in an assemblage (see above). 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, 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. 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. 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. 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. It is thus proposed to utilise a variety of analyses including, but not limited to, biodiversity measures, multivariate approaches as well as functional traits analyses (see for example Bremner, Rogers, & Frid, 2006 and Bolam, McIlwaine & Garcia, 2021 for methodology of functional traits analysis) to describe the macrofaunal colonization process after the dredging impact.

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Preliminary desktop assessment of potential for ocean CO_2 emissions from the planned NMP phosphate mining operations off the Namibian coast.

Pedro M. S. Monteiro

1. Background

The recently released Intergovernmental Panel on Climate Change (IPCC) 6th Assessment Round (AR6) Working Group 1 report set out in clear terms the science basis for the urgent need for all countries to reduce their CO_2 emissions (Canadell et al., 2021). This is required not only in respect of helping countries remain compliant with the Nationally Determined Contributions (NDCs) under the Paris Agreement but also in reducing effects on oceans. Climate change affects marine ecosystem health and natural resources primarily through increased temperatures, ocean de-oxygenation and ocean acidification linked to anthropogenic CO_2 emissions. Such effects increasingly combine into what is termed – multi stressor extremes – that cause long lasting damage to ecosystems (Gruber et al., 2021). In view of such effects CO_2 emission estimations are required for industries to allow for controls, mitigations leading to reductions and appropriately scaled offsets. This assessment has the objective of providing preliminary quantitative constraints for the magnitude of potential CO_2 emissions arising from the mobilization of sea bottom CO_2 rich waters to the sea surface during the dredging phase of the proposed Sandpiper project.

2. Approach and assumptions

The adopted approach employs a simple observation-based model with the primary assumption that the dominant control on the change of the partial pressure of CO_2 (pCO_2) from bottom / interstitial waters to the surface is temperature. Temperature and hence CO_2 solubility changes from 11 °C at the seabed (NMP data) to two SST scenarios of 12 °C and 18 °C. These two likely extremes of surface warming increase pCO_2 which is instantaneously re-equilibrated with an atmospheric pCO_2 of 410 µatm. This loss of CO_2 during the equilibration constitutes the emissions. It should also be noted that any contributions by benthic organic matter remineralization and carbonate dissolution to the CO_2 concentrations in bottom waters is already implicit in the concentrations of Total Dissolved CO_2 (C^T) and Total Alkalinity (A^T).

3. Data Limitations of this assessment

There are very limited observations of the ocean carbonate system on the Namibian shelf. It is a major gap in the otherwise adequate long-term observations in that coastal ecosystem. The known data sets are those collected in 1992 (Monteiro et al., 1996), a set of observations made by a German team, which to the best of my knowledge has not been published and finally and most recently a data set collected by DFFE: Oceans & Coasts, South Africa along the 23 °S NatMIRC monitoring line in 2014-2015. This latter data set is now being analysed as part of Mr M Tsanwani's[1] PhD but is not yet published. For this reason, this preliminary assessment used data from the observations in 1992 which are publicly accessible through a PhD thesis (Monteiro 1996). The data are derived from the box model means used to calculate the CO_2 fluxes through the Namibian shelf system (Monteiro, 1996). The 1992 observations were collected north of the NMP mining area (±24 °S) on the 22 °S line (Henties Bay line) but they are representative of comparable parts of the system. It is considered that the use of this historical data set is appropriate and will have little or no bearing on the calculated fluxes.

4. Model outputs

The modelled temperature and uncertainty dependent range of total annual CO_2 emissions from a single dredger are shown in Table 1 in units of tonnes of CO_2 per year (see appendix for model set up and outputs).

Table 1: Estimated annual CO_2 emissions from the dredging process (tonnes CO_2 per year)

SST Scenario	Low estimate	High estimate
Emissions at 12 °C	126.3	162.4
Emissions at 18 °C	169.6	204.8

As these numbers show, the uncertainty in the inputs of C^T and A^T had a comparable contribution to the output range as the SST scenario. Medium confidence (±66%) is attributed to these ranges.

5. Implications

These are relatively small magnitudes of CO_2 emissions from the translocation of cool DIC rich bottom water to the warmer surface and comparable to emissions from 60 000 – 100 000 litres of petrol (30 – 50% of the fuel load of a large passenger plane).

The indicative annual estimate for the dredging process itself within ML170 based on the targeted 5.5 million tonnes of dredged sediment, assuming sediment density of 1.8 tonnes/m^3 and CO_2 emissions of 3.2 kg/m^3 of dredged sediment (van der Bilt, 2019), is 9 983 tonnes. The dredger will also travel to and from the dredge site to the discharge location at Walvis Bay, a distance of ~240 km, approximately 221 times a year with an overall estimated distance of 53 040 km. The emission rate of 3.2 kg/m^3 (above) has a sailing distance of 18.5 km per round trip factored in so the net distance in terms of additional emissions is 48 952 km. CO_2 emissions for a large TSHD are ~0.42 tonnes/km (derived from van der Bilt, 2019) and emissions for this phase of the dredging operations are estimated at 20 560 tonnes. The indicative overall emissions for dredging within ML170 and the delivery of product to the discharge site in Walvis Bay is thus 30 542 tonnes CO_2/year. Note that the latter component will be addressed within the EIA for the on-land ore processing elements of the Sandpiper project.

The relatively low annual CO_2 emissions from the proposed dredging and sailing operations for the marine phase of the Sandpiper Project estimated here should be included with the emission calculations for the land-based elements of the Sandpiper Project which includes the materials handling and processing operations. This will provide the total of NMP emissions that may need to be offset for the entire value chain of the mining and beneficiation steps as these are developed and instituted.

Note that under the assumption that the pCO_2 equilibrates rapidly at the surface, the impact on ocean acidification from the translocation of bottom waters to the sea surface by the proposed dredging operations is predicted to be limited.

[1] Senior Scientist at Department of Fisheries, Forestry and Environment: Oceans & Coasts, South Africa, and PhD student at University of Cape Town.

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7. Appendix

Spreadsheet Model to Caculate the likely range of CO2 emissions per year from NMP mining / dredging activities

Dr Pedro Monteiro
CSIR
South Africa

NMP Dredge Volume Calculations (from Dr R Carter)

From	HR Wallingford Dredge Plume Modelling 2020 & JdN 20		
Dredge Slurry Pump Rate	6	m^3/s	
%Water	60%		
%Sediment	40%		
Water Rate	3.6	m^3/s	
Dredge Cycle Period	58.5	hrs	Discharge pt/sail/dredge/ sail/discharge
Slurry Pump/cycle Period	18.4	hrs	Includes initial hopper fill period of 142 minutes and 964 minutes of overflow
Dredge cycles/week	2.87		
Slurry pumped/week	6.86E+05	m^3	
Water pumped/week	4.12E+05	m^3	
Slurry pump/year	3.57E+07	m^3	
Water pump/year	**2.14E+07**	**m^3**	**Only number used below**
			Assumed input values

Model inputs	**Min**	**Max**		
Water pumped per year				
Temperature BBL	11	11	oC	NMP
Temperature SL	12	18	oC	NMP
Salinity	35	35	psu	NMP
Total Alkalinity (AT)	2384	2388	umol/kg	Monteiro, PMS, 1996, PhD Thesis Table 5.1

Total CO2 (CT)	2317	2360	umol/kg	Monteiro, PMS, 1996, PhD Thesis Table 5.1
Atmospheric CO2 2021	410	415	ppm	

Model Outputs

pCO2 in the BML	948	1354	uatm	Calculated using CO2SYS (Orr et al., 2018)
pCO2 at the SL 12oC	987	1406	uatm	Ditto
pCO2 at the SL 18oC	1247	1757	uatm	Ditto
TCO2 before equilibration	2317	2360	umol/kg	Ditto
TCO2 after equilibration 12oC	2177	2180	umol/kg	Ditto
TCO2 at the SL 18oC	2129	2133	umol/kg	Ditto
TCO2 loss after equilibration 12oC	140	180	umol/kg	
TCO2 loss after equilibration 18oC	188	227	umol/kg	
Total CO2 lost to the atmosphere per year assuming equilibration - 12oC	2.92E+09	3.76E+09	mmol C	corrected by 1.025 to convert to a volume concentration
Total CO2 lost to the atmosphere per year assuming equilibration - 18oC	3.93E+09	4.74E+09	mmol C	
Estimated emissions 12oC	126.3	162.4	tons CO2	corrected from mol to tons and from carbon to CO2
Estimated emissions 18oC	169.6	204.8	tons CO2 / year	

8.1 VESSEL: DREDGING OPERATIONS:

Commencement of dredging operations (includes recovery of sediments, fine tailings return, water column and marine fauna, but excludes all general ship activities, steaming, bunkering, and discharging of hopper contents) – Refer to Section 8.9 for activity plans.

Performance Objectives	
The management objectives are: • Through mitigation and monitoring, minimise direct effects of the operation on the marine environment. • Manage dredging related impacts on the marine environment to avoid compromising future exploitation of renewable marine resources. • Establish and maintain an information base that will assist in evaluating cumulative impacts. • Establish recovery rates of marine habitats impacted during the dredging. • Through communication minimise potential conflict with other marine users. • Promote information exchange with scientific institutions and I&APs. • Protect heritage sites (shipwrecks) if encountered. • Ensure for safety of the operation, by applying all relevant safe vessel operations	
Standards of compliance	
• Minerals (Prospecting and Mining) Act (33 of 1992). • Environmental Management Act (Act 7 of 2007). • Marine Resources Act 27 of 2000 • MARPOL. • Petroleum products and energy amendment Act (2000). • Draft Pollution Control and Waste Management Bill. • Merchant Shipping Act 57 of 1951. • Wreck and Salvage Act (2004). • United Nation Convention on Law of the Sea.	• The London Convention. • Convention of Biological Diversity. • Territorial Sea and Exclusive Economic Zone of Namibia Act, No. 3 of 1990. • Namibian Ports Authority Act 2 of 1994. • Marine Traffic Act, No. 2 of 1981 as amended by the Marine traffic Amendment Act 15 of 1991. • Prevention and Combating of Pollution of the Sea by Oil Act No 6 of 1981 (as amended by Act 24 of 1991). • Dumping at Sea Control Act 73 of 1980. • Environmental Statement Policy

Activity	Aspect	Impact	Management Requirement	Duration
Dredging of the deposit	Biodiversity	Destruction of Sediment profile	For each target area, conduct geophysical survey prior to dredging and post dredging, to determine / provide information on: • Area and volume (thickness) of sediment removed versus predicted; • Residual volume (thickness) of sediment covering the footwall; • Morphological character of the seafloor; • Seabed areas within the dredged target area that are undisturbed by the dredging process; • Readjust frequency and scope of further post dredging geophysical surveys if required as a function of interpreted information against the primary objectives of the survey; • Integrate output information into adjustments for the next period of dredging; and • Retain data and interpretations to a GIS system.	Before the start of dredging & at 3-year intervals during the dredging programme.
			• Through geophysical techniques record the morphological characteristics of a completed dredge zone; • Integrate output information into subsequent recovery surveys; and • Retain data and interpretations to an environmental GIS system.	During dredging
		Removal of sediments, loss of benthos and disturbance to the environment	For each of the target areas (SP1, SP2, SP3) prior to dredging commencing the following initial sample / data collection and assessments are required to update the existing baseline information and provide the before component of MBACI: • Collection of representative macro faunal assemblages • Collection of sediment for the evaluation of grain size, organics, dissolved nutrients & H_2S and whole sediment toxicity testing. (surficial and internal samples) • Determination of the presence and distribution of thiobacterial mats • Integrate output information into adjustments of the planned recovery surveys; and • Retain data and interpretations to an environmental GIS system • Benthic large epifauna biodiversity survey • Acoustic survey in dredge area to characterise the underwater sound field as preparation for acoustic monitoring during dredge operations • Marine mammal, seabird & other marine fauna observations	Pre dredging assessment:

Activity	Aspect	Impact	Management Requirement	Duration
Dredging of the deposit		Recovery of benthos and environment	Conduct for the target dredge areas: • Collection and analyses of representative macrofauna samples from impact and reference sites • Sediment collection for grain size, organics, dissolved nutrients, trace metals & H_2S and whole sediment toxicity testing (surficial and internal samples) • Disturbance, distribution, recovery of thiobacteria mats • From the results of the simulation modelling confirm areas of actual and potential high sedimentation (that, in dredged areas may enhance recolonisation and/or compromise benthic communities in adjacent areas due to smothering/burial) • Determine benthos recovery / re-colonisation rates • Revise future recovery monitoring programmes in context of the EIA accordingly, and • Retain data and interpretations to an environmental GIS system	At 3-year intervals over the SP1 dredging programme. This can be revised according to apparent benthos recovery rates.
			• Marine mammal, seabird & other marine fauna observations and acoustic monitoring	During dredging
	Discharge of 'lean' water overboard: Turbidity plumes	Water column quality	For each of the target areas prior to dredging being undertaken the following initial sample / data collection and assessments are required to update the existing baseline data: • Collection of targeted seabed sediment for the evaluation of grain size, organics, dissolved nutrients, trace metals & H_2S and whole sediment toxicity (surficial and internal samples) • Integrate the biogeochemical output information into the data and assumptions of the simulation model; • Integrate the biogeochemical output information into adjustments of the planned in-plume verification surveys; • If warranted revise EIA assumptions in context accordingly, and • Retain data and interpretations to an environmental GIS system	Pre dredging assessments:

Activity	Aspect	Impact	Management Requirement	Duration
			During dredging in each target area undertake in-plume surveys designed to: • Verify plume behaviour, distribution and impacts determined from simulation modelling based on collected site specific information; • Integrate in-plume survey output information into the simulation modelling to refine model algorithms and/or assumptions; • Based on outputs develop plans for the next in-plume survey(s); and • Retain data and interpretations to an environmental GIS system	During years 1-3 of the SP1 dredging programme for each target area conduct surveys in winter/spring and summer/autumn to cover periods of bottom water ventilation and hypoxic flows from the north. Thereafter conduct surveys at3-year intervals to confirm recorded distributions at additional dredging areas.
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