

**ENVIRONMENTAL IMPACT ASSESSMENT FOR RÖSSING URANIUM'S PROPOSED
DESALINATION PLANT NEAR SWAKOPMUND**

MARINE ECOLOGY SPECIALIST STUDY

Prepared for

SLR Environmental Consulting (Namibia) (Pty) Ltd

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ABBREVIATIONS, UNITS AND GLOSSARY

Abbreviations

ANZECC	Australian and New Zealand Environment and Conservation Council
CD	Chart Datum
CIP	Clean in Place
CITES	Convention on International Trade in Endangered Species
CMS	Convention on Migratory Species
CSIR	Council for Scientific and Industrial Research
DWAF	Department of Water Affairs and Forestry
EIA	Environmental Impact Assessment
HAB	Harmful Algal Blooms
IUCN	International Union for Conservation of Nature
MPA	Marine Protected Area
PIM	Particulate Inorganic Matter
POM	Particulate Organic Matter
RO	Reverse Osmosis
RSA DWAF	Republic of South Africa, Department of Water Affairs and Forestry
RSA	Republic of South Africa
SACW	South Atlantic Central Water
SWRO	Seawater Reverse Osmosis
TSPM	Total Suspended Particulate Matter
US-EPA	United States Environmental Protection Agency

Units used in the report

$\mu\text{g}/\ell$	micrograms per litre
cm	centimetres
cm/s	centimetres per second
$\text{g C}/\text{m}^2/\text{day}$	grams Carbon per square metre per day
h	hours
kg	kilogram
km	kilometres
km^2	square kilometres
m	metres
m/s	metres per second
mm	millimetres
m^2	square metres
m^3/day	cubic metres per day
m/s	metres per second
mg/ℓ	milligrams per litre
$\text{mg Chl a}/\text{m}^3$	milligrams Chlorophyll a per cubic metre
psu	practical salinity units which in the normal oceanic salinity ranges are the same as $^{\circ}/_{00}$
s	seconds
%	percentage
~	approximately
<	less than
>	greater than
$^{\circ}\text{C}$	degrees centigrade

Glossary

Barotropic	a fluid whose density is a function of only pressure
Bathymetry	measurements of the depths of the ocean relative to mean sea level.

Benthic	Referring to organisms living in or on the sediments of aquatic habitats (lakes, rivers, ponds, etc.).
Benthos	The sum total of organisms living in, or on, the sediments of aquatic habitats.
Benthic organisms	Organisms living in or on sediments of aquatic habitats.
Biodiversity	The variety of life forms, including the plants, animals and micro-organisms, the genes they contain and the ecosystems and ecological processes of which they are a part.
Biomass	The living weight of a plant or animal population, usually expressed on a unit area basis.
Biota	The sum total of the living organisms of any designated area.
Bivalve	A mollusk with a hinged double shell.
Community structure	All the types of taxa present in a community and their relative abundance.
Community	An assemblage of organisms characterized by a distinctive combination of species occupying a common environment and interacting with one another.
Dilution	The reduction in concentration of a substance due to mixing with water.
Dissolved oxygen (DO)	Oxygen dissolved in a liquid, the solubility depending upon temperature, partial pressure and salinity, expressed in milligrams/litre or millilitres/litre.
Eckman transport	the net motion of fluid as the result of a balance between Coriolis and turbulent drag forces.
Ecosystem	A community of plants, animals and organisms interacting with each other and with the non-living (physical and chemical) components of their environment.
Effluent	A complex waste material (e.g. liquid industrial discharge or sewage) that may be discharged into the environment.
Epifauna	Organisms, which live at or on the sediment surface being either attached (sessile) or capable of movement.
Environmental impact	A positive or negative environmental change (biophysical, social and/or economic) caused by human action.
Environmental quality objective	A statement of the quality requirement for a body of water to be suitable for a particular use (also referred to as Resource Quality Objective).
Guideline trigger values	These are the concentrations (or loads) of the key performance indicators measured for the ecosystem, below which there exists a low risk that adverse biological (ecological) effects will occur. They indicate a risk of impact if exceeded and should 'trigger' some action, either further ecosystem specific investigations or implementation of management/remedial actions.
Habitat	The place where a population (e.g. animal, plant, micro-organism) lives and its surroundings, both living and non-living.
Infauna	Animals of any size living within the sediment. They move freely through interstitial spaces between sedimentary particles or they build burrows or tubes.
Inter-specific stress	biological stress between co-existing species
Intertidal	the area of a seashore which is covered at high tide and uncovered at low tide

Macrofauna	Animals >1 mm.
Macrophyte	A member of the macroscopic plant life of an area, especially of a body of water; large aquatic plant.
Meiofauna	Animals <1 mm.
Mariculture	Cultivation of marine plants and animals in natural and artificial environments.
Marine discharge	Discharging wastewater to the marine environment either to an estuary or the surf zone or through a marine outfall (<i>i.e.</i> to the offshore marine environment).
Marine environment	Marine environment includes estuaries, coastal marine and near-shore zones, and open-ocean-deep-sea regions.
Offshore blinder	a subtidal reef that reaches almost to the sea surface.
Pollution	The introduction of unwanted components into waters, air or soil, usually as result of human activity; e.g. hot water in rivers, sewage in the sea, oil on land.
Population	Population is defined as the total number of individuals of the species or taxon.
Purse seine	a large fishing net designed to be set by two boats around a school of fish and arranged such that after the ends have been brought together the bottom can be closed, the net thereby acting as a bag.
Recruitment	The replenishment or addition of individuals of an animal or plant population through reproduction, dispersion and migration.
Sediment	Unconsolidated mineral and organic particulate material that settles to the bottom of aquatic environment.
Sessile	attached directly by its base to the substratum without a stalk or peduncle
Species	A group of organisms that resemble each other to a greater degree than members of other groups and that form a reproductively isolated group that will not produce viable offspring if bred with members of another group.
Sludge	Residual sludge, whether treated or untreated, from urban wastewater treatment plants.
Subtidal	The zone below the low-tide level, <i>i.e.</i> it is never exposed at low tide.
Supralittoral	The supralittoral zone is situated above the high water spring tide level.
Surf zone	Also referred to as the 'breaker zone' where water depths are less than half the wavelength of the incoming waves with the result that the orbital pattern of the waves collapses and breakers are formed.
Suspended material	Total mass of material suspended in a given volume of water, measured in mg/ℓ.
Suspended matter	Suspended material.
Suspended sediment	Unconsolidated mineral and organic particulate material that is suspended in a given volume of water, measured in mg/ℓ.
Taxon (Taxa)	Any group of organisms considered to be sufficiently distinct from other such groups to be treated as a separate unit (e.g. species, genera, families).
Trophodynamics	The dynamics of nutrition or metabolism, also nutritional energy.SBMS
Turbidity	Measure of the light-scattering properties of a volume of water, usually measured in nephelometric turbidity units.

Vulnerable

A taxon is vulnerable when it is not Critically Endangered or Endangered but is facing a high risk of extinction in the wild in the medium-term future.

TABLE OF CONTENTS

ABBREVIATIONS, UNITS AND GLOSSARY	I
1. GENERAL INTRODUCTION	1
1.1 Background.....	1
1.2 Scope of Work	1
1.3 Approach to the Study	1
1.4 Limitations and Assumptions.....	2
1.5 Structure of the Report.....	2
2. PROJECT DESCRIPTION IN RELATION TO THE MARINE ENVIRONMENT	4
2.1 Description of the Desalination Plant Facilities	4
2.2 Identification and Selection of Alternatives	5
2.2.1 Seawater Intake Structure	5
2.2.2 Discharge of Brine Effluent	6
2.2.3 Other project alternatives.....	9
3. METHODOLOGY	10
4. LEGISLATIVE AND PERMITTING REQUIREMENTS.....	14
4.1 National Legislation	15
4.2 International Standards and Guidelines.....	15
4.3 Water Quality Guidelines	16
4.4 Mixing Zones	20
5. DESCRIPTION OF THE MARINE ENVIRONMENT.....	20
5.1 Introduction.....	21
5.2 Physical Environment.....	22
5.2.1 Seabed Topography, Bathymetry and Sediments	22
5.2.2 Waves.....	24
5.2.3 Tides	24
5.2.4 Coastal Currents	26
5.2.5 Surf zone Currents.....	26
5.2.6 Upwelling	26
5.2.7 Water Masses and Temperature	27
5.2.8 Turbidity	27
5.2.9 Organic Inputs.....	30
5.2.10 Low Oxygen Events	31
5.2.11 Sulphur Eruptions	32
5.3 Biological Environment.....	33
5.3.1 Rocky Habitats and Biota.....	33
5.3.2 Sandy Substrate Habitats and Biota	35
5.3.3 Pelagic Communities.....	39
5.4 Other Uses of the Area	49
5.4.1 Mariculture Activities	49
5.4.2 Fishing	51
5.4.3 Ecotourism	54
5.4.4 Industrial Uses	55
5.4.5 Existing Environmental Impacts.....	55
5.4.7 Potentially Threatened Habitats.....	56

6. IDENTIFICATION OF KEY ISSUES AND SOURCES OF POTENTIAL ENVIRONMENTAL IMPACT	57
6.1 Identification of Key Issues.....	57
6.1.1 Construction Phase	57
6.1.2 Commissioning Phase	57
6.1.3 Operational Phase	58
6.1.4 Decommissioning Phase	58
7. ASSESSMENT OF ENVIRONMENTAL IMPACTS	59
7.1 Construction of Intake and Discharge Structures.....	59
7.1.1 Disturbance of the Coastal Zone	59
7.1.2 Pollution and Accidental Spills	60
7.1.3 Increased Turbidity.....	62
7.1.4 Construction Noise and Blasting	64
7.1.5 Installation of Structures.....	67
7.2 Operational Phase	68
7.2.1 Impingement and Entrainment of organisms at intake	68
7.2.2 Flow Distortion.....	70
7.2.3 Desalination Plant Effluents	71
7.2.4 “No go” Alternative	92
7.2.5 Cumulative Impacts	92
8. RECOMMENDATIONS AND CONCLUSIONS	94
8.1 Comparison of Alternatives	94
8.2 Environmental Acceptability	94
8.3 General Recommendations and Mitigation Measures	95
8.3.1 General Recommendations relevant to Desalination Plants	95
8.3.2 Mitigation Measures	96
8.4 Monitoring Recommendations	98
8.4.1 Recommendations for Validation	98
8.4.2 Recommendations for Operational Monitoring	98
8.4.3 Recommendations for Surveillance Reviews	99
8.5 Conclusions and Impact Statement	100
9. REFERENCES	101
APPENDIX A:	114
A.1 Potential Effects of Blasting	115
A.2 Seawater Chlorine Chemistry and Associated Potential Impacts	117
A.3 Environmental Fate of Cleaning Chemicals used in the CIP Process	119
A.4 References	122

1. GENERAL INTRODUCTION

1.1 Background

Rössing Uranium Limited (Rössing Uranium) is investigating ways to improve its economic resilience and meet the challenges of current low uranium market prices. Since November 2013, Rössing Uranium and other mines in the region have been purchasing desalinated water for mining operations from the Areva desalination plant near Wlotzkasbaken, at significant cost. This was intended to be an interim measure until the new NamWater desalination plant at Mile 6 came into operation.

However, as the outcome, timelines and commercial aspects to the NamWater project remain uncertain, and an agreement with Areva to secure water on a long-term basis at economically feasible terms could not be reached, Rössing Uranium is considering an alternate source for desalinated seawater by way of a new desalination plant, ±6 km north of Swakopmund. By designing, constructing and operating a desalination plant to supply their water needs, Rössing Uranium hopes to reduce the overhead costs of its mining operations and enhance its commercial sustainability.

With respect to the proposed project, and in line with Namibia's Environmental Assessment Policy (1994) and the provisions contained in the Environmental Management Act (2007), Rössing Uranium requires the compilation of a Social and Environmental Impact Assessment (SEIA). SLR Environmental Consulting (Namibia) (Pty) Limited (SLR) and Aurecon Namibia (Pty) Ltd (Aurecon), have jointly been appointed to manage the SEIA process for the proposed desalination plant. SLR in turn has appointed Pisces Environmental Services (Pisces) to provide the Marine Specialist Report.

1.2 Scope of Work

SLR/Aurecon requested the compilation of a Marine Ecology Specialist Report based on:

- a review and verification of the Marine Specialist Study undertaken for the NamWater Mile 6 Desalination Plant EIA; and
- an assessment of impacts on the marine ecology of the project area based on the results of a hydrodynamic modelling study undertaken specifically for the proposed Rössing Uranium desalination plant.

The Marine Ecology Specialist Report is to provide input to the project-specific SEIA, and shall further contribute to the marine component of a Social Environmental Management Plan (SEMP), which will be submitted to the Ministry of Environment and Tourism with an application for environmental clearance.

Detailed Terms of Reference for the Marine EIA Report are to:

- provide a baseline description of the local marine environment and the main ecological processes;
- describe the impacts of the proposed development on the marine biology of the project area during the construction and operational phases of the desalination plant;
- identify and assess all factors resulting from the construction and operation of the desalination plant and associated infrastructure that may influence the marine and coastal environments in the region, based on existing information and the results of the hydrodynamic modelling study;
- recommend mitigation measures and management actions; and
- provide inputs into the SEM for the marine aspects of the construction and operational phases of the intake structure and brine disposal system.

1.3 Approach to the Study

Although the Marine Ecology Specialist Study largely adopts a desktop approach, qualitative information on the intertidal and shallow subtidal environments collected during the site visit and during exploratory dives at the proposed discharge location will be included in the description of

the baseline environment. This combined approach is deemed adequate for placing into context the potential impacts associated with a desalination plant of the capacity proposed for this development. Furthermore, the assessment of impacts associated with the brine discharge will be based on the results of a numerical modelling study undertaken by WSP (hydrodynamic modelling specialist), thereby adding confidence to the assessment of the likely extent and duration of the hypersaline effluent footprints under different seasonal oceanographic scenarios.

1.4 Limitations and Assumptions

The following are the assumptions and limitations of the study:

- The study is based on the project description made available to the specialist at the time of the commencement of the study (plant capacities, discharge locations, constituents, volumes, etc.). The impact assessment is restricted to only those constituents specified by Rössing Uranium as being contained within the effluents from the desalination plant.
- The ecological assessment is limited to a “desktop” approach and thus relies on existing information only. However, site-specific descriptions for three sites spanning the coastline between the seawater intake and brine discharge locations were provided by divers, who swam transects through the surf zone perpendicular to the shoreline, photographically recording seabed type, notable features and representative marine biota.
- The modelling study comprises semi-empirical methods and an analytical near-field model. This approach was adopted because sophisticated numerical models typically used for modelling of brine discharges in deep water are incapable of numerically simulating discharges in the turbulent beach and nearshore zone. Some important conclusions and associated assessments and recommendations made in the marine ecology assessment are based on the modelling results. The predictions of these models, whilst considered to be robust in terms of the major discharge constituent, need to be validated by field observations and subsequent monitoring. If field observations and monitoring, however, fail to mirror predicted results, the forecasted impacts will need to be confirmed and /or reviewed through a post-commissioning monitoring programming.
- Potential changes in the marine environment such as sea level rise and/or increases in the severity and frequency of storms related to climate change are not explicitly considered here. Such scenarios are difficult to assess due to the uncertainties surrounding climate change. Should evidence or more certain predictions of such changes become available, Rössing Uranium should re-assess their development and management plans to include the impacts of these anticipated macroscale changes. However, it is not expected that these climate changes will affect the effluent plume behaviour to the extent that the conclusions of this study will be altered.

1.5 Structure of the Report

This Marine Specialist Study Report describes the effects of the construction and operation of the proposed Rössing Uranium desalination plant on the marine environment (*i.e.* the coastal zone below the high water mark), and its significance within the context of the receiving environment in the vicinity of Mile 6 north of Swakopmund. The report outlines the approach to the study, assesses impacts identified by a marine specialist consultant, and makes recommendations for mitigation, monitoring and management of these impacts. The report is structured as follows:

Section 1: General Introduction - provides a general overview to the proposed project, and outlines the Scope of Work and objectives of the study and the report structure. Assumptions and limitations to the study are also given.

Section 2: Project Description relative to the Marine Environment - gives a brief overview of the marine components of the proposed RO Plant, giving some technical detail on the alternative project designs considered and the volume, nature and water quality of the proposed discharges from the RO Plant.

Section 3: Methodology - provides details of the assessment methodology applied to the study.

Section 4: Legislative and Permitting Requirements - details the regulatory requirements, as well as other guidelines that are applicable to the marine aspects of the desalination project.

Section 5: Description of the Marine Environment - describes the receiving biophysical environment that could be impacted by the RO Plant. Existing impacts on the environment are discussed and sensitive and/or potentially threatened habitats or species are identified;

Section 6: Identification of Key Issues and Sources of Potential Environmental Impact - here key issues identified during the public consultation and environmental screening process for the proposed Desalination Project are identified and summarised in terms of the construction phase, operational phase and decommissioning phase;

Section 7: Assessment of Environmental Impact - identifies and assesses the significance of potential direct, indirect and cumulative environmental impacts on the marine environment associated with the construction and operation of the RO Plant and associated infrastructure, based on information provided by the client and the results of the modelling studies.

Section 8: Recommendations and Conclusions - the environmental acceptability of the development alternatives are discussed, and the environmentally preferred alternative is identified. A comparison between the “no development” alternative and the proposed development alternatives is also included. Mitigation measures and monitoring recommendations are presented.

Section 9: References - provides a full listing of all information sources and literature cited in this chapter.

2. PROJECT DESCRIPTION IN RELATION TO THE MARINE ENVIRONMENT

The site for the proposed desalination plant is approximately 6 km north of Swakopmund, at the existing Swakopmund Salt Works.

The project will comprise the following components:

- A seawater intake system and associated infrastructure located in the vicinity of the existing Swakopmund Salt Works intake.
- Either an open channel or pipeline to transport water to the desalination plant.
- A new seawater intake buffer pond.
- The pre-treatment plant with a Dissolved Air Flotation (DAF) system to remove sediments, solids and organic matter from the feed-water.
- A modular Seawater Reverse Osmosis (SWRO) desalination plant with a peak capacity of 10,000 m³/day and an average production of ~3 million m³/year (or 8,200 m³/day). The desalination plant will be housed in an enclosure and fenced off area together with the post- and pre-treatment infrastructure. The plant will be situated on the Site option 1, nearest the existing salt works processing facility.
- The brine outlet system and associated infrastructure located within the Mining Licence area of the Salt Works and discharging brine into the surf zone.
- Construction of a new 6-km long, 11 kV power cable (buried) and a new substation at the plant.
- A water supply line of ~850 m length transporting desalinated water to the existing NamWater pipeline.
- Related services and structures i.e. offices, access road, etc.

The above describes the “preferred alternative”, however other alternatives will be considered and are described further on.

This study will focus only on the potential impacts associated with construction and operation of those aspects of the proposed development relevant to the marine environment, namely

- the seawater intake system, and
- the brine discharge system.

Impacts relating to plant location, power supply and the required linear routes between the desalination plant and the existing NamWater pipeline will be dealt with by other specialists and will not be further discussed here.

2.1 Description of the Desalination Plant Facilities

The basic process for the treatment of water in the proposed desalination plant is summarised here for the sake of completeness. Reverse Osmosis (RO) is a membrane filtration process utilised to reduce the salinity of seawater (feed-water). The feed-water is supplied through a seawater intake and appropriately treated before being pumped to a holding pond. To overcome the natural osmotic pressure of seawater, it is then pumped at high pressure through to the RO membranes. This process retains the brine (high salinity) on one side of the membranes and allows the water containing very low salinity to pass to the other side. The desalinated water is piped to the potable water tank and the brine is released back in to the ocean through discharge pipes. The recovery rate of product water through the process is typically approximately 40%.

The proposed output capacity of treated water is 3 Mm³/year (10,000 m³/day). The seawater intake structure will be designed for a gross instantaneous seawater abstraction capacity of 26,067 m³/day and a brine disposal system through which a maximum of 15,000 m³/day of brine will be continuously discharged back into the sea. The design life of the desalination plant is set at 10 years, which correlates the the currently Rossing Uranium Life of Mine plan.

The engineering technologies to be applied at the desalination plant will be flocculation (possibly bioflocculation), Dissolved Air Flotation (DAF), cartridge filtration, and reverse osmosis. A maximum of 60% of the sea water abstracted will be returned to the sea as concentrate (Brine plus

various filter backwash products and sludge from the pretreatment process). The selected design temperature of the water in the plant is 15 °C, but the plant will be capable of performing over a range of temperatures, with the RO feed pressure decreasing if the temperature is above 15 °C and the required feed pressure increasing when the water temperature is below 15 °C.

The use of a biocide (chlorine) may be required to inhibit biological growth in the pipelines, on the screens and in the media filters. To avoid damage to the RO membranes, the chlorinated water needs to be neutralised before it can pass through the membranes. Furthermore, a pipe 'pigging' system for regular maintenance and cleaning of the seawater supply lines (only intake) will be installed. This involves the use of a 'pig' (bullet-shaped device with bristles), which is introduced into the pipeline which transfers the feed-water from the pump station to the desalination plant to mechanically clean out the structure. Depending on the quality of the feed-water, the RO membranes will need to be cleaned at intervals of six to eight weeks. Residual streams from the cleaning process, DAF and membrane filtration systems would be neutralised with Sodium Metabisulphite (SMBS) before being added to the brine and discharged back to the ocean. An option to treat the sludge in purpose-designed handling facilities, with the solids recycled or disposed of to an accredited landfill site, is being considered.

Consideration is also being given to the use of the IDE ProGreen system. This modular, energy efficient, desalination technology is practical to deploy for small-to-mid-sized projects such as the proposed Rössing Uranium desalination plant. The ProGreen concept involves effective filtration at the pre-treatment phase thereby minimising biofouling and eliminating the need for membrane cleaning chemicals (i.e. no coagulants or disinfectants required). The chemical-free bio-filtration pre-treatment process maintains a low Silt Density Index while effectively eliminating bio-fouling. In combination with the complimentary chemical-free membrane Direct Osmosis Cleaning (DOC) system, the IDE ProGreen system offers an environmentally-friendly solution for the optimization of product flow, salt rejection, recovery rate, and operating costs. Each of the technologies can be deployed on a standalone basis, with recovery rates of 45% being achievable.

2.2 Identification and Selection of Alternatives

The number and size of seawater reverse osmosis desalination (SWRO) plants, as well as the environmental awareness related to such projects is increasing, resulting in increased emphasis on intake (and outfall) design and economics. Intake (and outfall) structures can be the 'fatal flaw issue' of new seawater desalination facilities. Whereas the design and manufacturing of the desalination unit itself are well established and approved technical solutions are available, the intake and pre-treatment of the seawater, and the discharge of brine need to be specifically adapted to the particular conditions at the plant site. These can differ over a wide range, as not only the raw water quality, but also aspects such as the geological situation, existing infrastructure and logistics need to be taken into consideration during the design, construction and operation of a desalination plant. The proponents engineering design team investigated numerous design options in a series of Trade-off studies undertaken as part of the project. With environmental input, consensus was gained on preferred and feasible alternatives for both seawater intake and brine discharge structures, and these are being assessed during the impact assessment phase.

2.2.1 Seawater Intake Structure

Intake alternatives for the supply of seawater to a desalination plant can largely be grouped into direct (open water) and indirect (water filtered through seabed) intakes. A submerged inlet structure situated in the vicinity of the existing Swakopmund Salt Works intake is proposed for the Rössing Uranium desalination plant. The intake would be a jetty situated within the intertidal zone with vertical turbine pumps located a set-back distance from its seaward end. The pumps would be protected by a fixed screen system. The fixed screen opening has been specified to be 100 mm with a maximum intake velocity of 0.15 m/sec to minimize impingement/entrainment. A velocity cap at the intake is also recommended. The abstracted seawater will be conveyed to the jetty's elevated land side section from where it will discharge into gravity flow pipelines and flow towards the head of an existing overland channel. The channel will lead to an inland seawater pond system located at the desalination plant.

Three potential intake locations were considered (Figure 1), namely:

Site 1: At the existing jetty - directly south of the existing salt works intake jetty;

Site 2: Outer shelf - approximately 100 m south;

Site 3: Yellow shelf - approximately 160 m south.

Following an assessment of the sites from a technical, financial, operation & maintenance and health & safety / environmental compliance perspective, the Trade-off Study (Royal HaskoningDHV & WSP 2014a) identified that Site 3: Yellow Shelf was the preferred alternative and that the other options would not be carried forward as feasible alternatives as they offered no tangible environmental benefits.

Open water intakes necessitate the need for extensive pre-treatment of intake water. Behind the screening and sand trap, the pre-treatment of the seawater would involve dissolved air flotation, which gives rise to a waste stream containing the filtered solids and any coagulant used. If this water cannot be returned to the feed-water source (typically the case if a coagulant has been added), the derived sludge must be dewatered either by settlement or by some mechanical process. To eliminate biological growth in the intake system, chlorination is widely used.

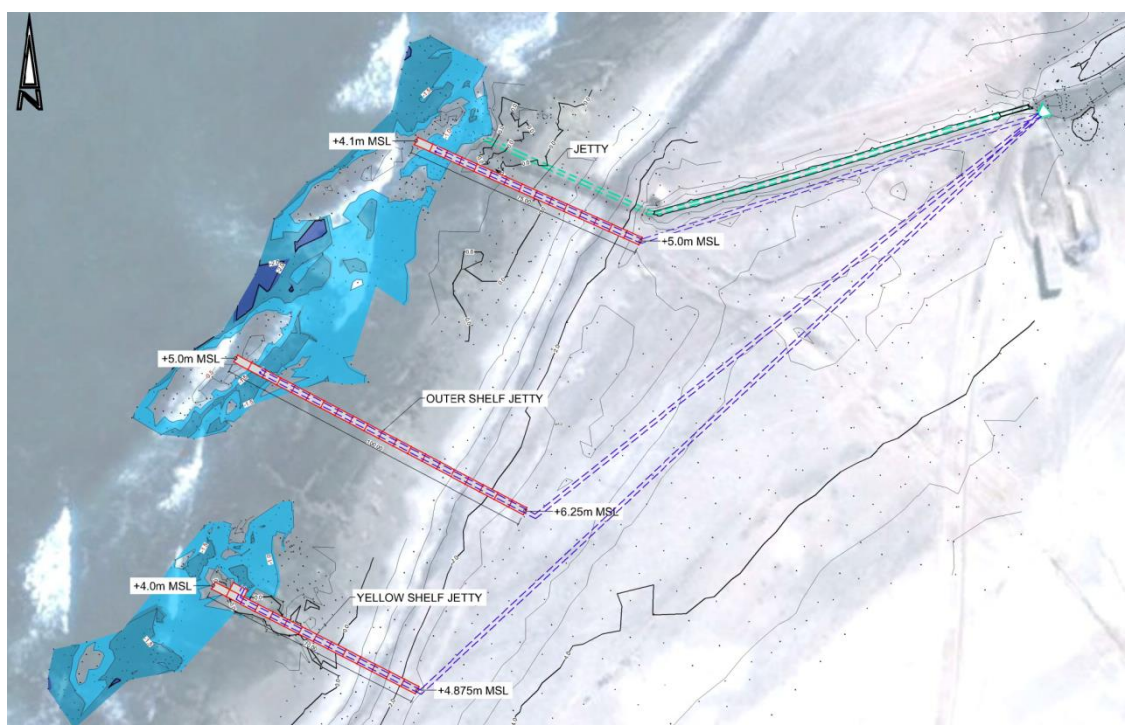


Figure 1: The three potential locations considered for the desalination plant's seawater intake (Royal HaskoningDHV & WSP 2014a).

2.2.2 Discharge of Brine Effluent

Various discharge alternatives exist for the disposal of effluent from a desalination plant, namely single-point or diffuser effluent pipelines, beach discharge wells, open-channel surf zone discharges, and evaporation ponds. As part of the proposed development of the Rössing Uranium desalination plant, a Trade-off Study was undertaken, which considered four methods of brine disposal, namely: offshore discharge, surf zone discharge, infiltration pond and beach channel disposal. Following an assessment of the discharge options from a technical, financial, operation & maintenance and health & safety / environmental compliance perspective, the Trade-off Study (Royal HaskoningDHV & WSP 2014b) identified that a single open surf zone outfall below the spring low water mark was the preferred discharge alternative as it would ensure the best brine disposal solution for the life of mine and magnitude of flow by achieving acceptable brine dilutions and dispersion through engineering design in conjunction with the local wave energy, at the lowest installation capital cost. The other discharge options have not been carried forward as feasible

alternatives. The offshore discharge would prove too expensive for the plant scale and the onshore discharge options were deemed to carry too great an environmental risk.

Typically, ocean outfalls for large seawater desalination plants extend beyond the surf zone, and are equipped with diffusers in order to provide the mixing necessary to prevent the heavy saline discharge plume from accumulating at the seabed in the immediate vicinity of the discharge. For smaller capacity plants such as being considered by Rössing Uranium, surf zone discharges are, however, a feasible alternative to the high-cost offshore option. Although the surf zone carries a significant amount of turbulent energy and usually provides much better mixing than the end-of-pipe type of diffuser outfall system, this zone has a limited capacity to transport the saline discharge load to the open ocean. If the mass of the saline discharge exceeds the threshold of the surf zone's salinity load transport capacity, the excess salinity would begin to accumulate in the surf zone and could ultimately result in a long-term salinity increment in this zone beyond the level of tolerance of the aquatic life (WHO 2007). This salinity threshold mixing/transport capacity of the surf zone can be determined using hydrodynamic modelling. A hydrodynamic modelling exercise was commissioned as part of the SEIA process and its results will inform the impact significance ratings of the marine ecology aspects.

In addition to the brine discharge methodology options, five potential brine outfall locations were identified (Figure 2) and a further trade-off study (Royal HaskoningDHV & WSP 2014c) undertaken to provide a comparative evaluation of the five options by qualitatively comparing the detailed attributes for each of the potential sites. This Trade-off Study recommended Outfall 5 as the preferred brine outfall location. However, Outfall 1 is also being considered due to the reduced construction efforts afforded by the existing concrete-encased pipeline structure. Both these options were thus considered in the hydrodynamic modelling study and in this marine ecological impact assessment. Outfall locations 2, 3 and 4, situated in between Options 1 and 5, were found not to bring any tangible benefits and have therefore not been carried forward into the SEIA as feasible alternatives.

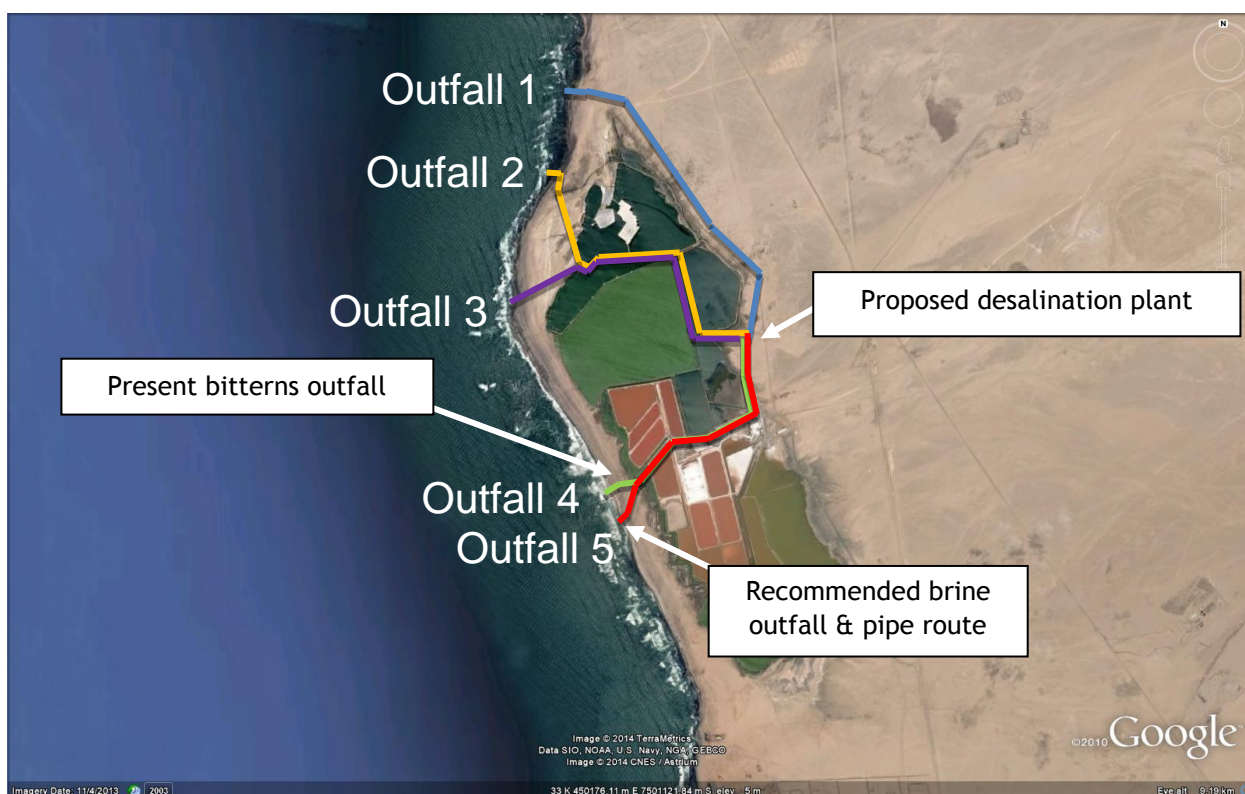


Figure 2: The five alternatives outfall location considered for the desalination plant's brine discharge (Royal HaskoningDHV & WSP 2014c).

Regardless of the final location, the outfall would consist of a pump station, a 2.25 km 400 mm diameter HDPE pipe, which would be trenched along its entire length and with the final ~40 m of pipeline seaward of the high-water mark being encased in concrete and either partially buried and incorporated with a natural rock outcrop 230 m south of the present bitterns discharge (Outfall 5) or laid adjacent to the existing concrete-encased pipeline structure (Outfall 1). The pipeline with diffuser would terminate in a natural deep area below the mean low water level.

Table 1-1 lists the expected composition of the brine effluent and the typical cleaning reagents and pre-treatment chemicals to be used should standard conventional RO technology be implemented. The brine effluent at the maximum plant capacity is anticipated to have a temperature of between 2-4° Celsius above the ambient average seawater temperature, a salinity of 66 g/ℓ or psu (based on the maximum feed-water salinity of 34.2 g/ℓ or psu), a density of 1,049 kg/m³, and with a maximum effluent flow of ~10 Mℓ/day.

Table 1-1: Expected composition and flow of the brine discharge from the proposed desalination plant assuming a capacity of 10 Mℓ/day. Discharges for preferred option (IDE Progreen™ system) are provided for comparison.

Description	Units	Quantity
Feed-water Intake (average)	m ³ /d	23,600
Feed-water Intake (instantaneous)	m ³ /d	26,100
Average brine discharge (average)	m ³ /d	13,550
Average brine discharge (instantaneous)	m ³ /d	15,000
Average Co-discharge (Pre-treatment and Media Filtration Backwash - intermittent and discharged over 24 h)	m ³ /d	1,338 (in 24 h)
Instantaneous Co-discharge (Pre-treatment and Media Filtration Backwash - intermittent and discharged over 24 h)	m ³ /d	1,483 (in 24 h)
Co-discharge (CIP rinse water for conventional RO system - 6 x per year only and assumed to be discharged over 12 h)	m ³ /d	202 (in 12 h)
Discharge velocity	m/s	~6
Salinity	mg/ℓ psu	66,000 66
Change in temperature	°C	2 - 4
pH		7.3 - 8.2
Suspended Solids (average)	mg/ℓ	8 - 12
Phosphonate antiscalant for conventional RO system	mg/ℓ	4 - 5
Chlorine - for conventional RO system	mg/ℓ	0.002
Sodium bisulphate (SMBS)	mg/ℓ	3 - 3.5
Spent CIP solution in waste flow (6 x per year and blended in over 12 hours)		
Peroxyacetic acid	mg/ℓ	0.003
Low pH cleaner		0.01
High pH cleaner		0.01
Preservative (sodium metabisulfite) in waste flow (twice a year)	mg/ℓ	6.0
Coagulant: Ferric Chloride (FeCl ₃) will precipitate into Ferric Hydroxide, which will be removed as a solid.	mg/ℓ	3-4
Discharges from preferred option (IDE Progreen™ system)		
CIP rinse water Co-discharge		
Phosphonate antiscalant	n/a	n/a
Chlorine		
Sodium bisulphate (SMBS)		

2.2.3 Other project alternatives

Several other trade-off studies were conducted and feasible alternatives to be assessed in the SEIA were identified. These include alternatives for the location of the desalination plant (Preferred Site 1 and two alternative sites are being considered). The preferred alternative includes a buried cable from the Tamarisk substation to the plant, however, an alternative of running this power line overhead along the C34, and then underground to the plant, is being assessed as a feasible alternative. In line with best practice, the “no go” alternative should also be considered and assessed during an SEIA process, however pursuing the “no go” alternative from the perspective of impacts to marine ecology is mute in this case as there will be no deviation from the *status quo*. These “other” project alternatives have been considered and it is determined that they will have no bearing on marine ecology and so have not been assessed as part of this study.

3. METHODOLOGY

Assessment of predicted significance of impacts for a proposed development is by its nature, inherently uncertain - environmental assessment is thus an imprecise science. To deal with such uncertainty in a comparable manner, standardised and internationally recognised methodology¹ has been developed. Such accepted methodology is applied in this study to assess the significance of the potential environmental impacts of the proposed development, outlined as follows:

For each impact, the EXTENT (spatial scale), MAGNITUDE (size or degree scale) and DURATION (time scale) are described. These criteria are used to ascertain the SIGNIFICANCE of the impact, firstly in the case of no mitigation and then with the most effective mitigation measure(s) in place. Tables 3-1 to 3-5 show the scale used to assess these variables, and define each of the rating categories.

Table 3-1: Assessment criteria for the evaluation of impacts

CRITERIA	CATEGORY	DESCRIPTION
Extent or spatial influence of impact	National	Within the country
	Regional	Within the province/recognised region
	Site-specific/ Local	On site or within 1,000 m of the impact site
Magnitude of impact (at the indicated spatial scale)*	High	Social and/or natural functions and/ or processes are severely altered (i.e. function is severely hampered and processes are unlikely to function)
	Medium	Social and/or natural functions and/ or processes are notably altered (i.e. function is affected to a noticeable degree and processes struggle to function effectively)
	Low	Social and/or natural functions and/ or processes are slightly altered (i.e. while function is affected in a measurable way, processes are likely to function, albeit sub-optimally)
	Very Low	Social and/or natural functions and/ or processes are negligibly altered (i.e. function is slightly affected and processes are likely to function effectively)
	Zero	Social and/or natural functions and/ or processes remain unaltered
Duration of impact	Long Term	More than 10 years
	Medium Term	Up to 10 years
	Short term (construction period)	Up to 3 years

*NOTE: Where applicable, the magnitude is related to the relevant standard (threshold value specified and source referenced).

The magnitude of impact is based on specialist knowledge of that particular field.

The SIGNIFICANCE of an impact is derived by taking into account the temporal and spatial scales and magnitude. Such significance is also informed by the context of the impact, i.e. the character and identity of the receptor of the impact. The means of arriving at the different significance ratings is

¹ As described, inter alia, in the South African Department of Environmental Affairs and Tourism's Integrated Environmental Management Information Series (Gov. of SA, 2002).

detailed in the tables below, developed by Ninham Shand in 1995 as a means of minimising subjectivity in such evaluations, i.e. to allow for replicability in the determination of significance.

Table 3-2: Definition of significance ratings

SIGNIFICANCE RATINGS	LEVEL OF CRITERIA REQUIRED
High	<ul style="list-style-type: none"> • High magnitude with a regional extent and long term duration • High magnitude with either a regional extent and medium term duration or a local extent and long term duration • Medium magnitude with a regional extent and long term duration
Medium	<ul style="list-style-type: none"> • High magnitude with a local extent and medium term duration • High magnitude with a regional extent and construction period or a site specific extent and long term duration • High magnitude with either a local extent and construction period duration or a site specific extent and medium term duration • Medium magnitude with any combination of extent and duration except site specific and construction period or regional and long term • Low magnitude with a regional extent and long term duration
Low	<ul style="list-style-type: none"> • High magnitude with a site specific extent and construction period duration • Medium magnitude with a site specific extent and construction period duration • Low magnitude with any combination of extent and duration except site specific and construction period or regional and long term • Very low magnitude with a regional extent and long term duration
Very low	<ul style="list-style-type: none"> • Low magnitude with a site specific extent and construction period duration • Very low magnitude with any combination of extent and duration except regional and long term
Neutral	<ul style="list-style-type: none"> • Zero magnitude with any combination of extent and duration

Once the significance of an impact has been determined, the PROBABILITY of this impact occurring as well as the CONFIDENCE in the assessment of the impact is determined using the rating systems outlined in the Tables 3-3 and 3-4. The significance of an impact should always be considered in concert with the probability of that impact occurring.

Table 3-3: Definition of Probability Ratings

PROBABILITY RATINGS	CRITERIA
Definite	Estimated greater than 95% chance of the impact occurring.
Probable	Estimated 5% to 95% chance of the impact occurring.
Unlikely	Estimated less than 5% chance of the impact occurring.

Table 3-4: Definition of Confidence Ratings

CONFIDENCE RATINGS*	CRITERIA
Certain	Wealth of information on and sound understanding of the environmental factors potentially influencing the impact.
Sure	Reasonable amount of useful information on and relatively sound understanding of the environmental factors potentially influencing the impact.
Unsure	Limited useful information on and understanding of the environmental factors potentially influencing this impact.

* The level of confidence in the prediction is based on specialist knowledge of that particular field and the reliability of data used to make the prediction.

Lastly, the REVERSIBILITY of the impact is estimated using the rating system outlined in Table 3-6.

Table 3-5: Definition of reversibility ratings

REVERSIBILITY RATINGS	CRITERIA
Irreversible	The activity will lead to an impact that is permanent.
Reversible	The impact is reversible, within a period of 10 years.

Despite attempts at providing a completely objective and impartial assessment of the environmental implications of development activities, environmental assessment processes can never escape the subjectivity inherent in attempting to define significance. The determination of the significance of an impact depends on both the context (spatial scale and temporal duration) and intensity of that impact. Since the rationalisation of context and intensity will ultimately be prejudiced by the observer, there can be no wholly objective measure by which to judge the components of significance, let alone how they are integrated into a single comparable measure.

This notwithstanding, in order to facilitate informed decision-making, environmental assessments must endeavour to come to terms with the significance of the potential environmental impacts associated with particular development activities. Recognising this, SLR/Aurecon have attempted to address potential subjectivity in the current SEIA process by:

- Being explicit about the difficulty of being completely objective in the determination of significance, as outlined above;
- Developing an explicit methodology for assigning significance to impacts and outlining this methodology in detail. Having an explicit methodology not only forces the assessor to come to terms with the various facets contributing towards the determination of significance, thereby avoiding arbitrary assignment, but also provides the reader of the SEIA Report with a clear summary of how the assessor derived the assigned significance;
- Differentiating, wherever possible, between the likely significance of potential environmental impacts as experienced by the various affected parties; and
- Utilising a team approach and internal review of the assessment to facilitate a more rigorous and defensible system.

Although these measures may not totally eliminate subjectivity, they provide an explicit context within which to review the assessment of impacts.

Environmental Assessment Policy in Namibia requires that, “as far as is practicable”, cumulative environmental impacts should be taken into account in all environmental assessment processes. SEIAs have traditionally, however, failed to come to terms with such impacts, largely as a result of the following considerations:

- Cumulative effects may be local, regional or global in scale and dealing with such impacts requires coordinated institutional arrangements; and
- Environmental assessments are typically carried out on specific developments, whereas cumulative impacts result from broader biophysical, social and economic considerations, which typically cannot be addressed at the project level.

However, when assessing the significance of the project level impacts, cumulative effects have been considered as far as it is possible in striving for best practice. The sustainability of the project is closely linked to assessment of cumulative impacts. The Uranium Rush SEA provides a strategic context against which to measure elements of sustainability and cumulative impacts and impacts in this SEIA will be measured against the desired states identified in the SEA.

4. LEGISLATIVE AND PERMITTING REQUIREMENTS

The Republic of Namibia follows an Environmental Assessment Policy that aims to achieve and maintain sustainable development, particularly the wise utilisation of natural resources and the responsible management of the biophysical environment. Accordingly, there are several regulatory requirements pertaining specifically to the marine environment, at international, national and regional level, to which the proposed Rössing Uranium desalination project will have to conform. These are briefly outlined below and summarised in Table 4-1.

Table 4-1: Relevant acts and the regulations for industrial activities in and adjacent to Namibian waters.

Law/Ordinance	Applicability
Article 95 (1) of the Constitution of the Republic of Namibia (1990)	<ul style="list-style-type: none"> • Preservation of Namibia's ecosystems, essential ecological processes and biological diversity • Sustainable use of natural resources
Environmental Assessment Policy of 1995	<ul style="list-style-type: none"> • Prescribes Environmental Impact Assessments for developments with potential negative impacts on the environment
Environmental Management and Assessment Act (2007)	<ul style="list-style-type: none"> • Establishes principles for environmental management and promoting integrated environmental management
Sea Birds and Seals Protection Act 46 of 1973	<ul style="list-style-type: none"> • No disturbance of seabirds and seals
Seashore Ordinance 37 of 1958	<ul style="list-style-type: none"> • Removal of living and non-living resources from seashore or seabed and depositing of rubbish within 3 nautical miles of the shore
Sea Fisheries Act 29 of 1992	<ul style="list-style-type: none"> • Dumping at sea • Discharge of wastes in marine reserves • Disturbance of rock lobsters, marine invertebrates or aquatic plants • Prohibited areas for catching/disturbing fish, aquatic plants or disturbing/damaging seabed
Nature Conservation Ordinance 4 of 1975	<ul style="list-style-type: none"> • Protection of various species
Marine Resources Act 27 of 2000 (and accompanying regulations)	<ul style="list-style-type: none"> • Discharges into the sea
Convention of Biological Diversity	<ul style="list-style-type: none"> • Protection of various species
Atmospheric Pollution Prevention Ordinance No. 11 of 1976	<ul style="list-style-type: none"> • Pollution prevention
Hazardous Substances Ordinance 14 of 1974, and amendments	<ul style="list-style-type: none"> • Pollution prevention
Petroleum Products and Energy amendment Act of 2000	<ul style="list-style-type: none"> • Disposal of used oil
Territorial Sea and Exclusive Economic Zone of Namibia Act 3 of 1990	<ul style="list-style-type: none"> • Exploitation of natural resources in the EEZ
Draft Pollution Control and Waste Management Bill (1999)	<ul style="list-style-type: none"> • Protection for particular species, resources or components of the environment
Water Resources Management Act 24 of 2004	<ul style="list-style-type: none"> • Water related pollution and abstraction
SADC Protocol on Shared Water Systems	<ul style="list-style-type: none"> • Water related pollution and abstraction
National Monuments Act 28 of 1969	<ul style="list-style-type: none"> • Disturbance of shipwrecks, archaeological and cultural sites
United Nations Law of the Sea Convention of 1982	<ul style="list-style-type: none"> • Marine pollution from seabed activities and land-based sources

4.1 National Legislation

The Namibian Environmental Management Act, No. 7 of 2007 provides a set of principles for environmental management in Section 3, and in Schedule 1 specifies a list of 35 activities that require an Environmental Impact Assessment (EIA). The Act promotes public participation, and makes provision for external review by the Environmental Commissioner, where required, at the proponent's expense. Following the Environmental Management Act, an environmental clearance certificate is required from the Directorate of Environmental Affairs (DEA) prior to commencement of operations for the desalination plant project.

Under Namibian legislation both the abstraction and discharge of water from and into the sea requires a permit under Sections 32 and 56 of the Water Resources Management Act (No. 24 of 2004), respectively. This Act is based on the National Water Policy (Ministry of Agriculture, Water and Rural Development 2000). The effluent water discharged (and potential associated co-discharges) from the proposed desalination plant is classified as "industrial effluent" and thus requires a license under the Act. A combined licence to abstract and use water and to discharge effluent is covered under Section 38 of the Act. Licenses are provided through the Ministry of Agriculture, Water Affairs and Forestry (MAWF).

The Ministry of Environment and Tourism (MET) is the custodian of Namibia's natural environment and discharges this duty via environmental regulations. The MET is thus the lead agent for EIAs; however licensing of water abstraction, use and disposal is only considered once an EIA acceptable to MAWF has been submitted and a Record of Decision has been handed down. Rossing Uranium will therefore apply for these licenses independently of this SEIA process.

A draft version of the Pollution Control and Waste Management Bill (1999) has amalgamated a variety of Acts and Ordinances that provide protection for particular species, resources or components of the environment. These include, but are not limited to, the Nature Conservation Ordinance No.4 of 1975, the Sea Fisheries Act 29 of 1992, the Sea Birds and Seals Protection Act 46 of 1973, Seashore Ordinance No. 37 of 1958, Hazardous Substances Ordinance No. 14 of 1974 and amendments, and the Atmospheric Pollution Prevention Ordinance No. 11 of 1976. All construction, disturbance, effluent and pollution resulting from the RO Plant project will be required to be in accordance with the requirements outlined in the Pollution Control and Waste Management Bill.

4.2 International Standards and Guidelines

In addition to the regional, national and international legislative requirements, there are international standards, protocols and guidelines applicable to a desalination plant project:

- In August 2007, the Department of Water Affairs & Forestry (DWAF 2007) of South Africa published the "*Guidelines for the evaluation of possible environmental impacts during the development of seawater desalination processes*". This document gives general guidance on the assessment procedure, lists possible environmental impacts which can be expected during implementation of seawater desalination, and provides recommendations for specialist and monitoring studies.
- The International Finance Corporation, a member of the World Bank Group, has developed operational policies (IFC 1998) that, inter alia, require that an impact assessment is undertaken within the country's overall policy framework and national legislation, as well as international treaties, and that natural and social aspects are to be considered in an integrated way. IFC has further published Environmental, Health, and Safety Guidelines (known as the 'EHS Guidelines') containing guidelines and standards applicable to projects discharging industrial wastewater (IFC 2007). The EHS Guidelines contain the performance levels and measures that are normally acceptable to IFC and are generally considered to be achievable in new facilities at reasonable costs by existing technology. The EHS Guidelines are technical reference documents with general and industry-specific examples of Good International Industry Practice (GIIP), as defined in IFC's Performance Standard 3 on Pollution Prevention and Abatement (IFC 2006). This Performance Standard has the objective to avoid and minimize adverse impacts on human health and the environment by avoiding or minimizing pollution from project activities. It outlines a project approach to pollution prevention and abatement in line with internationally disseminated pollution prevention and

control technologies and practices. In addition, Performance Standard 3 promotes the private sector's ability to integrate such technologies and practices as far as their use is technically and financially feasible and cost-effective in the context of a project that relies on commercially available skills and resources.

- Other guidance documents are those by the California Coastal Commission (Seawater Desalination and the California Coastal Act, 2004), the United Nations Environmental Programme (UNEP 2008) and the World Health Organisation (WHO, 2008) that include international best practices and principles such as the precautionary approach and describe how design and construction approaches can mitigate likely impacts.
- The Rio Declaration on Environment and Development (1992), which calls for use of EIA as an instrument of national decision making (Principle 17). Moreover, it establishes important principles for sustainable development that should be reflected in EIAs, such as the application of the precautionary principle (Principle 15, whereby, where there is uncertainty in the nature and severity of a potential impact, conservative assumptions are made with respect to the significance and potential severity of the impact being assessed).

As signatory to the Convention of Biological Diversity and Convention to Combat Desertification, Namibia is committed to the preservation of rare and endemic species, and to provide protection for ecosystems and natural life-support processes within the country's boundaries. As a signatory of the United Nations Law of the Sea Convention of 1982, Namibia is required to adopt legislation to reduce marine pollution from seabed activities in the Exclusive Economic Zone (EEZ) and on the continental shelf, and from land-based sources.

4.3 Water Quality Guidelines

The Water Resources Management Act does not contain any target values for water quality associated with brine effluents discharged into the marine environment. These will form part of the regulations associated with the new Water Act and will be implemented at a future date. As far as can be established, South Africa is the only southern African country that currently has an official set of water quality guidelines for coastal marine waters. In terms of policy, legislation and practice South Africa's operational policy for the disposal of land-derived wastewater to the marine environment (DWAF 2004 a-c) is thus of relevance. Specifically, environmental quality objectives need to be set for the marine environment, based on the requirements of the site-specific marine ecosystems, as well as other designated beneficial uses (both existing and future) of the receiving environment. The identification and mapping of marine ecosystems and the beneficial uses of the receiving marine environment provide a sound basis from which to derive site-specific environmental quality objectives (Taljaard *et al.* 2006). To ensure that environmental quality objectives are practical and effective management tools, they need to be set in terms of measurable target values, or ranges for specific water column and sediment parameters, or in terms of the abundance and diversity of biotic components. The South African Water Quality Guidelines for Coastal Marine Waters (DWAF 2005) provide recommended target values (as opposed to standards) for a range of substances, but these are not exhaustive. Therefore, in setting site-specific environmental quality objectives, the information contained in the DWAF guideline document is supported by additional information obtained from published literature and best available international guidelines (e.g. ANZECC 2000; World Bank 1998; EPA 2006). Recommended target values are also reviewed and summarized in the Benguela Current Large Marine Ecosystem (BCLME) document on water quality guidelines for the BCLME region (CSIR 2006). Recommended target values extracted from these guidelines are provided in Table 4-2.

As required by the Water Resources Management Act 24 of 2004, the Namibian Department of Water Affairs and Forestry is in the process of compiling regulations for water quality standards for effluent disposal to ground, groundwater and surface waters, including territorial coastal marine waters. To meet this objective, a set of Special Water Quality Standards for effluents has been proposed. Although not specifically stipulated as such, these appear applicable to effluent discharges into fresh water sources only. Nonetheless, for the sake of completeness, the proposed Special Water Quality Standards are presented in Table 4-3, with values for the combined brine and waste stream (before dilution) from the proposed Rössing Uranium desalination plant being provided for comparison. Should DWAF enforce these standards at a point discharge in the marine environment without taking cognizance of the dilution of the brine effluent in the mixing zone (see Section 4.4 below), an application for an exemption permit will need to be submitted.

Table 4-2: Water quality guidelines for the discharge of a high-salinity brine into the marine environment.

VARIABLE	SOUTH AFRICA (DWAf 2005)	AUSTRALIA/NEW ZEALAND (ANZECC 2000)	WORLD BANK ^a (World Bank 1998)	US ENVIRONMENTAL PROTECTION AGENCY (EPA 2006)
Zone of impact / mixing zone	To be kept to a minimum, the acceptable dimensions of this zone informed by the EIA and requirements of licensing authorities, based on scientific evidence.		100 m radius from point of discharge for temperature	
Temperature	The maximum acceptable variation in ambient temperature is $\pm 1^\circ\text{C}$	Where an appropriate reference system is available, and there are sufficient resources to collect the necessary information for the reference system, the median (or mean) temperature should lie within the range defined by the 20%ile and 80%ile of the seasonal distribution of the ambient temperature for the reference system.	$< 3^\circ\text{C}$ above ambient at the edge of the zone where initial mixing and dilution take place. Where the zone is not defined, use 100 meters from the point of discharge when there are no sensitive aquatic ecosystems within this distance.	
Salinity^b	33 - 36 psu	Low-risk trigger concentrations for salinity are that the median (or mean) salinity should lie within the 20%ile and 80%ile of the ambient salinity distribution in the reference system(s). The old salinity guideline (ANZECC 1992) was that the salinity change should be $<5\%$ of the ambient salinity.	-	
Total residual Chlorine	No guideline, however, deleterious effects recorded for concentrations as low as 2 - 20 $\mu\text{g}/\ell$. A conservative trigger value is $<2 \mu\text{g}/\ell$.	3 $\mu\text{g Cl}/\ell$ measured as total residual chlorine (low reliability trigger value at 95% protection level, to be used only as an indicative interim working level) (ANZECC 2000) ^c	0.2 mg/ℓ at the point of discharge prior to dilution	Long-term and short-term water quality criteria for chlorine in seawater are 7.5 $\mu\text{g}/\ell$ and 13 $\mu\text{g}/\ell$, respectively

VARIABLE	SOUTH AFRICA (DWA 2005)	AUSTRALIA/NEW ZEALAND (ANZECC 2000)	WORLD BANK ^a (World Bank 1998)	US ENVIRONMENTAL PROTECTION AGENCY (EPA 2006)
Dissolved oxygen	For the west coast, the dissolved oxygen should not fall below 10 % of the established natural variation. For the south and east coasts the dissolved oxygen should not fall below 5 mg/ℓ (99 % of the time) and below 6 mg/ℓ (95 % of the time)	Where an appropriate reference system is available, and there are sufficient resources to collect the necessary information for the reference system, the median lowest diurnal DO concentration for the period for DO should be >20 th ile of the ambient dissolved oxygen concentration in the reference system(s) distribution. The trigger value should be obtained during low flow and high temperature periods when DO concentrations are likely to be at their lowest.		
Nutrients	Waters should not contain concentrations of dissolved nutrients that are capable of causing excessive or nuisance growth of algae or other aquatic plants or reducing dissolved oxygen concentrations below the target range indicated for dissolved oxygen (see above)	Default trigger values of PO ₄ -P: 100 µg/ℓ NO _x -N: 50 µg/ℓ NH ₄ ⁺ -N: 50 µg/ℓ for the low rainfall southern Australian region (Table 3.3.8 in ANZECC 2000)		
Chromium	8 µg/ℓ (as total Cr)	Marine moderate reliability trigger value for chromium (III) of 10 µg./ℓ with 95% protection Marine high reliability trigger value for chromium (VI) of 4.4 µg/ℓ at 95% protection.	0.5 mg/ℓ (total Cr) for effluents from thermal power plants	1 100 µg/ℓ for highest concentration at brief exposure without unacceptable effect 50 µg/ℓ highest concentration at continuous exposure without unacceptable effect
Iron		Insufficient data to derive a reliable trigger value. The current Canadian guideline level is 300 µg/ℓ	1.0 mg/ℓ for effluents from thermal power plants	

VARIABLE	SOUTH AFRICA (DWAf 2005)	AUSTRALIA/NEW ZEALAND (ANZECC 2000)	WORLD BANK ^a (World Bank 1998)	US ENVIRONMENTAL PROTECTION AGENCY (EPA 2006)
Molybdenum		Insufficient data to derive a marine trigger value for molybdenum. A low reliability trigger value of 23 µg/ℓ was adopted to be used as indicative interim working levels.		
Nickel	25 µg/ℓ (as total Ni)	7 µg/ℓ at a 99% protection level is recommended for slightly-moderately disturbed marine systems.		74 µg/ℓ for highest concentration at brief exposure without unacceptable effect 8.2 µg/ℓ highest concentration at continuous exposure without unacceptable effect

- ^a The World Bank guidelines are based on maximum permissible concentrations at the point of discharge and do not explicitly take into account the receiving environment, *i.e.* no cognisance is taken of the fact of the differences in transport and fate of pollutants between, for example, a surf zone, estuary or coastal embayment with poor flushing characteristics and an open and exposed coastline. It is for this reason that we include in this study other generally accepted Water Quality guidelines that take the nature of the receiving environment into account.
- ^b The ANZECC (2000) Water Quality guideline for salinity is less stringent than, but roughly approximates, the South African Water Quality guideline that requires that salinity should remain within the range of 33 psu to 36 psu (=ΔS of approximately 1 psu). Scientific studies have shown that effects on marine biota are primarily observed for increases of >4 psu above ambient level. ΔS 1 psu and 4 psu have been chosen for assessment purposes.
- ^c In case of chlorine “shocking”, which involves using high chlorine levels for a short period of time rather than a continuous low-level release, the target value is a maximum value of 2 mg/ℓ for up to 2 hours, not to be repeated more frequently than once in 24 hours, with a 24-hour average of 0.2 mg/ℓ (The same limits would apply to bromine and fluorine.).

Table 4-3: Proposed Special Water Quality Standards for Effluents (DWAF 2014) and expected values *before dilution* in the brine effluent from the proposed Rössing Uranium desalination plant.

Determinant	Unit	Proposed Special Water Quality Standards for Effluents	Combined Brine and Waste stream (before dilution)
Turbidity	NTU	<5	10
Colour		<10%	
Suspended solids	mg/l	<25 mg/l	50
TDS	mg/l	<500 mg/l above the intake potable water quality	63,000
pH		6.5- 9.5	7.5 - 8
Temp	C	± 1°C of ambient	(+) 2 - 4
Nitrate as N		< 15 mg/l (as N)	
Nitrite as N	mg/l	<2 mg/l	0.4 - 0.75
Fluoride (F)	mg/l	< 1 mg/l	2
Na	mg/l	<50mg/l above the intake potable water quality	20,000
Ca	mg/l	Not specified	750
Mg	mg/l	Not specified	2,500
K	mg/l	Not specified	900
Chloride as Cl	mg/l	<40mg/l above the intake potable water quality	38,000
Alkalinity as CaCO ₃	mg/l	Not specified	
Hardness as CaCO ₃	mg/l	Not specified	
Sulphate as SO ₄	mg/l	<20mg/l above the intake potable water quality	5,000
Iron as Fe	mg/l	<200 µg/l	4 - 5

4.4 Mixing Zones

A mixing zone is the area around an effluent discharge point where the effluent is actively diluted with the water of the receiving environment. This zone usually encompasses the near-field and mid-field regions of dilution to allow for the plume to mix throughout the water column. No water quality criteria for physical and chemical stressors are defined within the mixing zone. Instead, these water quality criteria ('trigger values') are defined at the boundary of the mixing zone to ensure the quality of nearby waters does not deteriorate as a result of the effluent discharge. The boundaries of a proposed mixing zone are typically defined according to an estimated distance from the discharge point at which point defined water quality guidelines will be met, as predicted by numerical modelling of the discharge.

5. DESCRIPTION OF THE MARINE ENVIRONMENT

5.1 Introduction

This environmental description encompasses the coastal zone and shallow nearshore waters (< 40 m depth) extending from Walvis Bay north to Henties Bay (Figure 3). Some of the data presented is, however, more regional in nature, e.g. the wave climate, nearshore currents, etc. The purpose of this environmental description is to provide the marine baseline environmental context within which the desalination plant project proposed by Rössing Uranium will take place. Certain aspects of the biophysical environment (e.g. climate, wind, etc.) are not only applicable to the marine context and are thus described in the overarching EIA and omitted here.



Figure 3: The location of the salt works and associated existing and proposed infrastructure in relation to the coastal zone between Walvis Bay and Henties Bay. The two alternative outfall options are shown (yellow lines).

5.2 Physical Environment

5.2.1 Seabed Topography, Bathymetry and Sediments

The coastal strip around Swakopmund is covered by a 2-3 m thick layer of very loose, medium to fine grained sea sand, which stretches ~200 m inland. Only in the vicinity of Henties Bay is the shore backed by low sandy cliffs.

As part of the pre-feasibility phase for the NamWater Desalination Plant Project, the CSIR (2008a) conducted a geophysical and hydrographical survey of the area directly north of the salt works using sub-bottom profiling and echosounding. Although the salt works are located on a slight promontory, it is expected that the bathymetry offshore will be very similar to that recorded during the NamWater Study. These bathymetric data showed a gently sloping seabed reaching the -10 m depth contour at around 1,700 m offshore. No bathymetric data are available for depths inshore of -3 m to -4 m contour, but existing information suggests a rock plate sloping very gently into the intertidal area. This rocky shelf is prominent between the old concrete intake structure and the current seawater intake for the saltworks, and to the western-most point of the salt works (Figure 4a), becoming patchy further south (Figure 4b). Offshore blinders occur to the west and south of the old bittens disposal site. There is a prominent berm on the upper beach along much of the coastline (Figure 4c). From there, the beach slopes steeply to the low water mark.



Figure 4: The coastline west of the salt works is characterised by a rocky intertidal shelf in the north, and pebble beaches with a prominent berm further south (Photos: C. Soltau, WSP Coastal Engineers).

The surficial sediments in the intertidal and low-shore areas are generally dominated by moderately to well-sorted fine to medium sand with median particle sizes of 200- 400 µm and heavy minerals present in the sediments. In the south of the study area, the sediments become coarser and can contain substantial proportions of gravel and pebbles, with occasional extensive pebble beds in the mid- and low-shore (Figure 4c).

Further offshore, the seafloor is dominated by undulated rock or hard sediment with occasional rock outcrops or reefs running either parallel or at an angle to the coastline. The rock surface appears rough with a micro relief of approximately 0.5-1.0 m. Sandy areas are sparse, and generally occur in small isolated patches scattered over the area. The sediment accumulations are thin with a maximum observed thickness of 1.8 m.

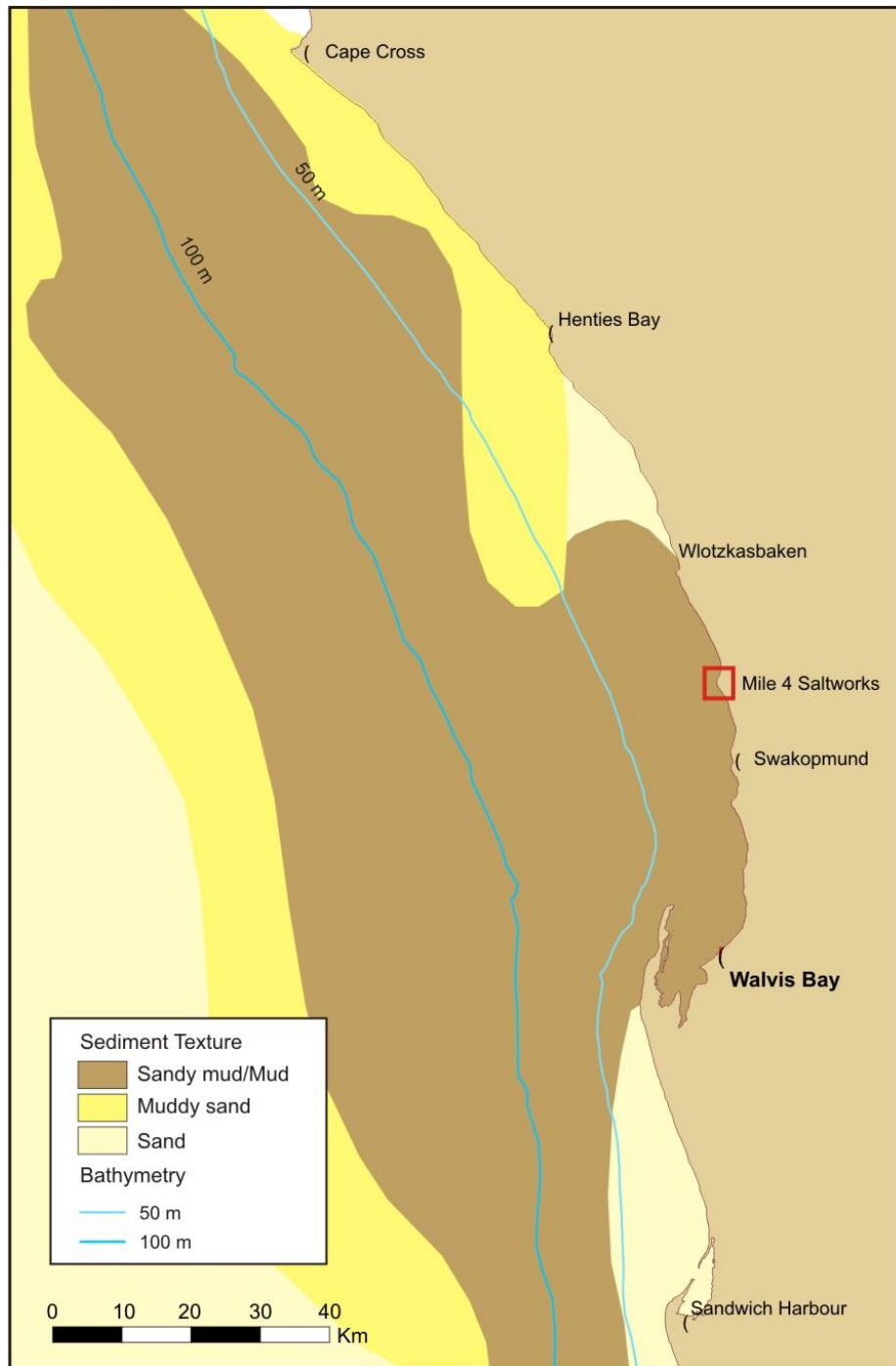


Figure 5: The location of the proposed desalination plant (red rectangle) in relation to the offshore seabed sediments in the region.

Transects perpendicular to the shore were swum by divers in four localities between the Outfall 1 and Outfall 5 alternatives. These investigations identified that off Outfall 1 the seabed to 200 m offshore comprises smooth, flat bedrock with sparse patches of sand. Off Outfall 5 the seabed to 200 m offshore is characterised by small, shallow gullies of 0.5 - 1 m wide and 0.3 m deep orientated perpendicular to the coast. Inshore bedrock is covered by a thin veneer of sand, with rocks protruding through the unconsolidated sediments in places. Further offshore the sand is replaced by patches of cobbles.

Beyond the 100-m depth contour, the seabed is dominated by a tongue of sandy mud, which extends from south of Sandwich Harbour to the north past Henties Bay (Figure 5). These biogenic muds, which comprise organically rich diatomaceous oozes originating from planktonic detritus, are the main determinants of the formation of low-oxygen waters and sulphur eruptions off central Namibia.

The central Namibian continental shelf is covered by layers of sediments primarily of biogenic (biological) origin as a result from the high productivity in the upwelled waters (see below for upwelling). A significant feature of the central Namibian middle shelf is an extensive mud belt.

5.2.2 Waves

The central Namibian coastline is influenced by major swells generated in the Roaring Forties, as well as significant sea waves generated locally by the persistent south-westerly winds. Apart from Walvis Bay and Swakopmund, wave shelter in the form of west to north-facing embayments, and coast lying in the lee of headlands are extremely limited.

No measured wave data are available for the Swakopmund - Henties Bay area. However, data collected by Voluntary Observing Ships indicate that wave heights in the range of 1.5 m to 2.5 m occur most frequently, with a mean wave height of 2.14 m and mean wave periods in the range of 8 s to 13 s (Figure 6). Longer period swells with mean periods of 11 s to 15 s and generated by mid-latitude cyclones occur about 30% of the time.

Wind-induced waves on the other hand have shorter wave periods (~8 seconds), and are generally steeper than swell-induced waves. Storms occur frequently with significant wave heights over 3 m occurring 10% of the time. The largest waves recorded originate from the S-SW sectors and may attain 4-6 m.

The annual distribution indicates that 75% of the waves come from the SSW and SW, with ~18% coming from the S. There is no strong seasonal variation in the wave regime except for slight increases in swell from WSW-W direction in winter.

5.2.3 Tides

In common with the rest of the southern African coast, tides in the study area are regular and semi-diurnal. The maximum tidal variation is approximately 2 m, with a typical tidal variation of ~1 m. Variations of the absolute water level as a result of meteorological conditions such as wind and waves can however occur adjacent to the shoreline and differences of up to 0.5 m in level from the tidal predictions are not uncommon. Tidal currents are minimal with measurements of 0.1 m/s reported at Walvis Bay. Table 5-1 lists mean tidal levels for Walvis Bay.

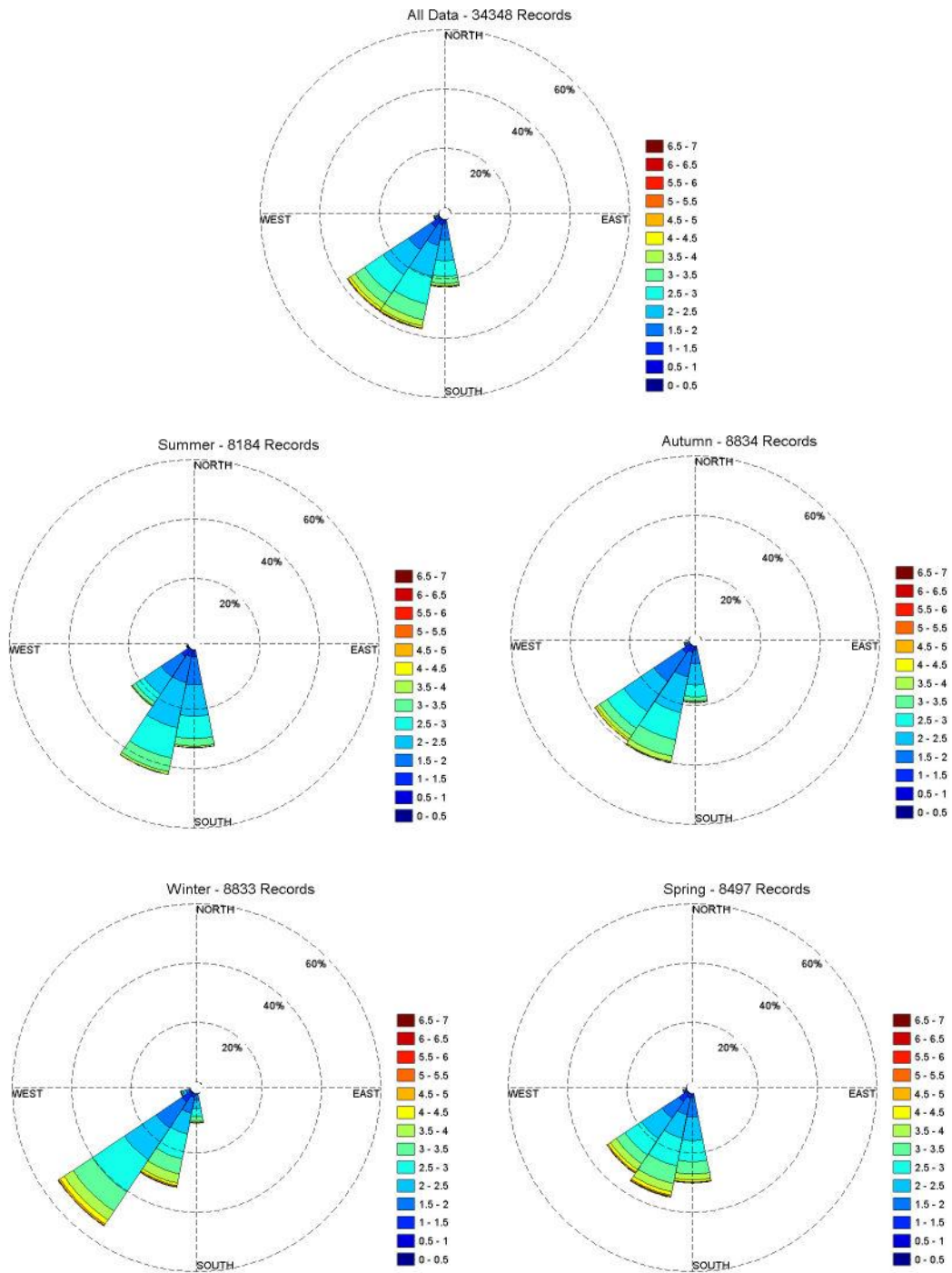


Figure 6: Seasonal offshore wave conditions for a data point located at 23° S, 13.75° E (Source: CSIR modelling study for the NamWater desalination plant EIA (CSIR, 2009)).

Table 5-1: Tide statistics for Walvis Bay from the SA Tide Tables (SAN 2007). All levels are referenced to Chart Datum.

Description		Level in m
Highest Astronomical Tide	HAT	+1.97
Mean High Water of Spring Tide	MHWS	+1.69
Mean High Water of Neap Tide	MHWN	+1.29
Mean Level	ML	+0.98
Mean Sea Level	MSL	+0.966
Mean Low Water of Neap Tide	MLWN	+0.67
Mean Low Water of Spring Tide	MLWS	+0.27
Lowest Astronomical Tide	LAT	0.00

5.2.4 Coastal Currents

Current velocities in continental shelf areas of the Benguela region range generally between 10 - 30 cm/s (Boyd & Oberholster 1994). The flows are predominantly wind-forced, barotropic and fluctuate between poleward and equatorward flow (Shillington *et al.* 1990; Nelson & Hutchings 1983). Fluctuation periods of these flows are 3 - 10 days, although the long-term mean current residual is in an approximate NW (alongshore) direction. Currents in the nearshore environment along the coastline of the study area have not been well studied, but some surface-current measurements were done at Swakopmund between 1971 and 1972 (CSIR 2005). Surface currents in the area appear to be quite variable, with flows primarily <30 cm/s and an average velocity of 14 cm/s. Current speeds in reverse flows observed between Walvis Bay and Henties Bay range between 2 - 17 cm/s. Near bottom shelf flow is mainly poleward (Nelson 1989) with low velocities of typically 5 cm/s.

5.2.5 Surf zone Currents

Typically wave-driven flows dominate in the surf zone (characteristically 150 m to 250 m wide), with the influence of waves on currents extending out to the base of the wave effect (~40 m; Rogers 1979). The influence of wave-driven flows extends beyond the surf zone in the form of rip currents. Longshore currents are driven by the momentum flux of shoaling waves approaching the shoreline at an angle, while cross-shelf currents are driven by the shoaling waves. The magnitude of these currents is determined primarily by wave height, wave period, angle of incidence of the wave at the coast and bathymetry. Surf zone currents have the ability to transport unconsolidated sediments along the coast in the northward littoral drift.

Nearshore velocities have not been reported and are difficult to estimate because of acceleration features such as surf zone rips and sandbanks. However, computational model estimates using nearshore profiles and wave conditions representative of this coastal region suggest time-averaged northerly longshore flows which have a cross-shore mean of between 0.2 to 0.5 m/s. Instantaneous measurements of cross-shore averaged longshore velocities are often much larger. Surf zone-averaged longshore velocities in other exposed coastal regions commonly peak at between 1.0 m/s to 1.5 m/s, with extremes exceeding 2 m/s for high wave conditions (CSIR 2002). The southerly longshore flows are considered to remain below 0.5 m/s.

5.2.6 Upwelling

The major feature of the Benguela system is upwelling and the consequent high nutrient supply to surface waters leads to high biological production and large fish stocks. The prevailing longshore, equatorward winds move nearshore surface water northwards and offshore. To balance the

displaced water, cold, deeper water wells up inshore. Although the rate and intensity of upwelling fluctuates with seasonal variations in wind patterns, the most intense upwelling tends to occur where the shelf is narrowest and the wind strongest. The largest and most intense upwelling cell is in the vicinity of Lüderitz, and upwelling can occur there throughout the year (Figure 7). Off northern and central Namibia, several secondary upwelling cells occur. Upwelling in these cells is perennial, with a late winter maximum (Shannon 1985).

5.2.7 Water Masses and Temperature

South Atlantic Central Water (SACW) comprises the bulk of the seawater in the study area, either in its pure form in the deeper regions, or mixed with previously upwelled water of the same origin on the continental shelf (Nelson & Hutchings 1983). Salinities range between 34.5‰ and 35.5‰ (Shannon 1985). For the Swakopmund area an ambient salinity of 34.2‰ has been reported (CSIR 2009). Data recorded over a ten year period at Swakopmund (1988 - 1998) show that seawater temperatures vary between 10°C and 23°C, averaging 14.9°C. They show a strong seasonality with lowest temperatures occurring during winter when upwelling is at a maximum (Figure 8).

During the non-upwelling season in summer, daily seawater temperature fluctuations of several degrees are common along the central Namibian nearshore coast. It appears that the thermal regime of the surf zone is controlled by the locally-forced Ekman offshore transport, which leads the associated temperature fluctuations by one day (Bartholomae & Hagen 2007). This time-lag suggests the existence of a persistent recirculation cell in nearshore waters in this region.

The continental shelf waters of the Benguela system are characterised by low oxygen concentrations, especially on the bottom. SACW itself has depressed oxygen concentrations (~80% saturation value), but lower oxygen concentrations (<40% saturation) frequently occur (Visser 1969; Bailey *et al.* 1985; Chapman & Shannon 1985).

Nutrient concentrations of upwelled water of the Benguela system attain 20 µM nitrate-nitrogen, 1.5 µM phosphate and 15-20 µM silicate, indicating nutrient enrichment (Chapman & Shannon 1985). This is mediated by nutrient regeneration from biogenic material in the sediments (Bailey *et al.* 1985). Modification of these peak concentrations depends upon phytoplankton uptake which varies according to phytoplankton biomass and production rate. The range of nutrient concentrations can thus be large but, in general, concentrations are high.

5.2.8 Turbidity

Turbidity is a measure of the degree to which the water loses its transparency due to the presence of suspended particulate matter. Total Suspended Particulate Matter (TSPM) is typically divided into Particulate Organic Matter (POM) and Particulate Inorganic Matter (PIM), the ratios between them varying considerably. The POM usually consists of detritus, bacteria, phytoplankton and zooplankton, and serves as a source of food for filter-feeders. Seasonal microphyte production associated with upwelling events will play an important role in determining the concentrations of POM in coastal waters. PIM, on the other hand, is primarily of geological origin consisting of fine sands, silts and clays. PIM loading in nearshore waters is strongly related to natural inputs from rivers or from 'berg' wind events, or through resuspension of material on the seabed.

Concentrations of suspended particulate matter in shallow coastal waters can vary both spatially and temporally, typically ranging from a few mg/ℓ to several tens of mg/ℓ (Bricelj & Malouf 1984; Berg & Newell 1986; Fegley *et al.* 1992). Field measurements of TSPM and PIM concentrations in the Benguela current system have indicated that outside of major flood events, background concentrations of coastal and continental shelf suspended sediments are generally <12 mg/ℓ, showing significant long-shore variation (Zoutendyk 1992, 1995). Considerably higher concentrations of PIM have, however, been reported from southern African west coast waters under stronger wave conditions associated with high tides and storms, or under flood conditions.

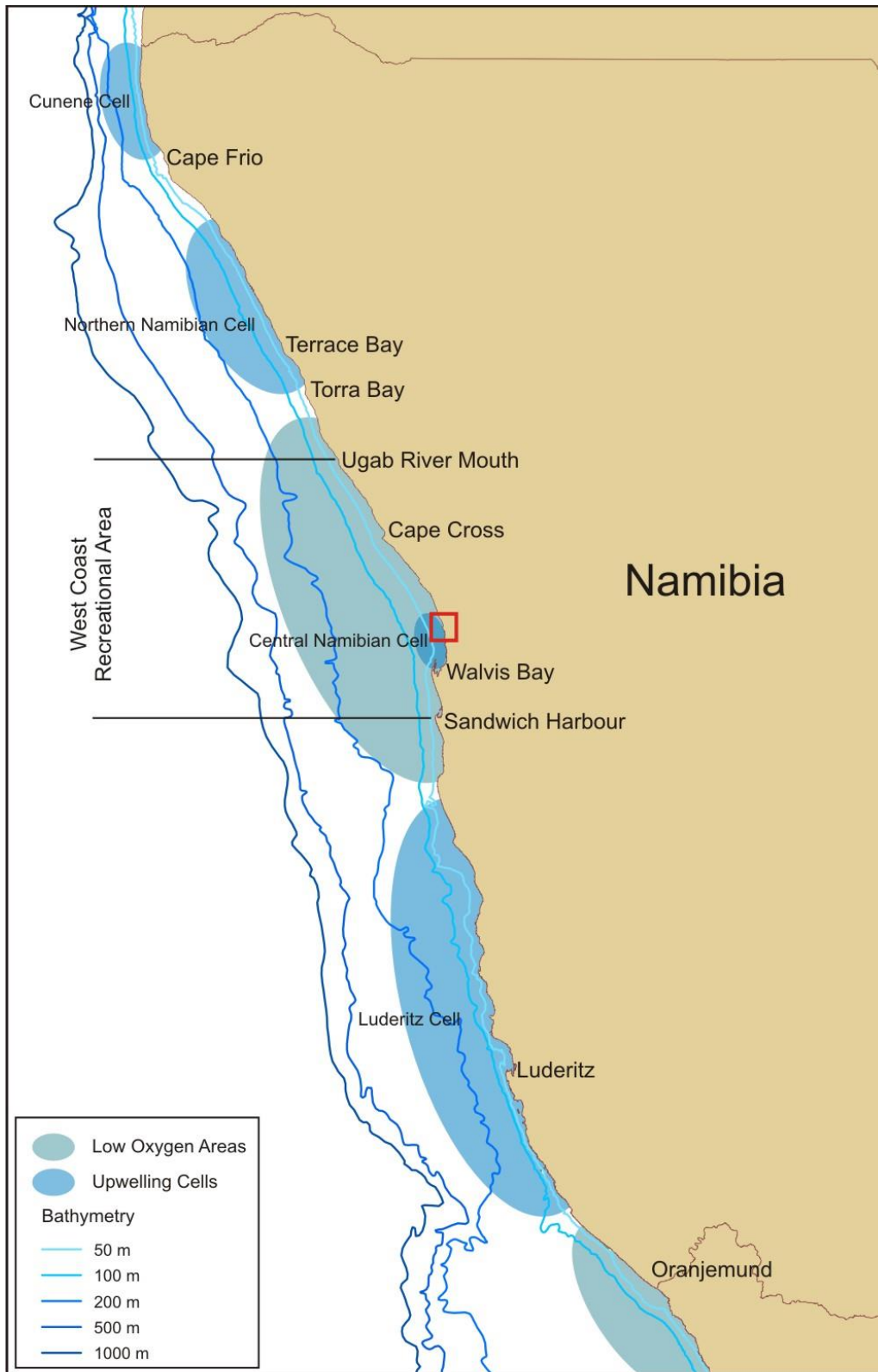


Figure 7: The location of the proposed desalination plant (red rectangle) in relation to the upwelling cells and the formation zones of low oxygen water.

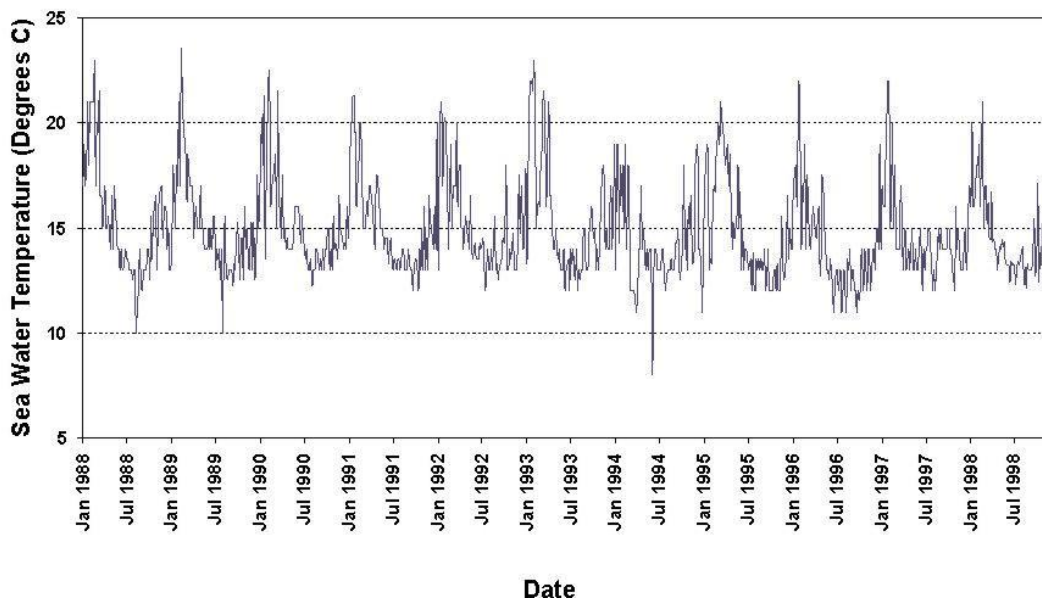


Figure 8: Seawater temperatures at Swakopmund recorded between 1988 and 1998.

The major source of turbidity in the swell-influenced nearshore areas off Namibia is the redistribution of fine inner shelf sediments by long-period Southern Ocean swells. The current velocities typical of the Benguela (10-30 cm/s) are capable of resuspending and transporting considerable quantities of sediment equatorwards. Under relatively calm wind conditions, however, much of the suspended fraction (silt and clay) that remains in suspension for longer periods becomes entrained in the slow poleward undercurrent (Shillington *et al.* 1990; Rogers & Bremner 1991).

Superimposed on the suspended fine fraction, is the northward littoral drift of coarser bedload sediments, parallel to the coastline. This northward, nearshore transport is generated by the predominantly southwesterly swell and wind-induced waves. Longshore sediment transport, however, varies considerably in the shore-perpendicular dimension. Sediment transport in the surf zone is much higher than at depth, due to high turbulence and convective flows associated with breaking waves, which suspend and mobilise sediment (Smith & Mocke 2002).

On the inner and middle continental shelf, the ambient currents are insufficient to transport coarse sediments, and resuspension and shoreward movement of these by wave-induced currents occur primarily under storm conditions (see also Drake *et al.* 1985; Ward 1985).

The powerful easterly 'berg' winds occurring along the Namibian coastline in autumn and winter also play a significant role in sediment input into the coastal marine environment (Figure 9), potentially contributing the same order of magnitude of sediment input as the annual estimated input of sediment by the Orange River (Zoutendyk 1992; Shannon & O'Toole 1998; Lane & Carter 1999). For example, for a single 'berg'-wind event it was estimated that 50 million tons of dust were blown into the sea by extensive sandstorms along much of the coast from Cape Frio, Namibia in the north to Kleinsee, South Africa in the south (Shannon & Anderson 1982) with transport of the sediments up to 150 km offshore.



Figure 9: Satellite image showing dust plumes being blown offshore from the Namibia coast (Image source: www.intute.ac.uk).

5.2.9 Organic Inputs

The Benguela upwelling region is an area of particularly high natural productivity, with extremely high seasonal production of phytoplankton and zooplankton. These plankton blooms in turn serve as the basis for a rich food chain up through pelagic baitfish (anchovy, pilchard, round-herring and others), to predatory fish (snoek), mammals (primarily seals and dolphins) and seabirds (jackass penguins, cormorants, pelicans, terns and others). All of these species are subject to natural mortality, and a proportion of the annual production of all these trophic levels, particularly the plankton communities, die naturally and sink to the seabed.

Balanced multispecies ecosystem models have estimated that during the 1990s the Benguela region supported biomasses of 76.9 tons/km² of phytoplankton and 31.5 tons/km² of zooplankton alone (Shannon *et al.* 2003). Thirty six percent of the phytoplankton and 5% of the zooplankton are estimated to be lost to the seabed annually. This natural annual input of millions of tons of organic material onto the seabed off the southern African west coast has a substantial effect on the ecosystems of the Benguela region. It provides most of the food requirements of the particulate and filter-feeding benthic communities that inhabit the sandy-muds of this area, and results in the high organic content of the muds in the region. As most of the organic detritus is not directly consumed, it enters the seabed decomposition cycle, resulting in subsequent depletion of oxygen in deeper waters overlying these muds and the generation of hydrogen sulphide and sulphur eruptions along the coast.

An associated phenomenon ubiquitous to the Benguela system are red tides (dinoflagellate and/or ciliate blooms) (see Shannon & Pillar 1985; Pitcher 1998). Also referred to as Harmful Algal Blooms (HABs), these red tides can reach very large proportions, with sometimes spectacular effects. Toxic dinoflagellate species can cause extensive mortalities of fish and shellfish through direct poisoning, while degradation of organic-rich material derived from both toxic and non-toxic blooms results in oxygen depletion of subsurface water. Periodic low oxygen events associated with massive algal blooms in the nearshore can have catastrophic effects on the biota (see below).

5.2.10 Low Oxygen Events

The low oxygen concentrations are attributed to nutrient remineralisation in the bottom waters of the system (Chapman & Shannon 1985). The absolute rate of this is dependent upon the net organic material build-up in the sediments, with the carbon rich mud deposits playing an important role. As the mud on the shelf is distributed in discrete patches, there are corresponding preferential areas for the formation of oxygen-poor water, the main one being off central Namibia (Chapman & Shannon 1985) (see Figure 7). The distribution of oxygen-poor water is subject to short (daily) and medium term (seasonal) variability in the volumes of oxygen depleted water that develops (De Decker 1970; Bailey & Chapman 1991). Subsequent upwelling processes can move this low-oxygen water up onto the inner shelf, and into nearshore waters, often with devastating effects on marine communities.

Oxygen deficient water can affect the marine biota at two levels. It can have sub-lethal effects, such as reduced growth and feeding, and increased intermolt period in the rock-lobster population (Beyers *et al.* 1994). The oxygen-depleted subsurface waters characteristic of the southern and central Namibian shelf are an important factor determining the distribution of rock lobster in the area. During the summer months of upwelling, lobsters show a seasonal inshore migration (Pollock & Shannon 1987), and during periods of low oxygen become concentrated in shallower, better-oxygenated nearshore waters.

On a larger scale, periodic low oxygen events in the nearshore region can have catastrophic effects on the marine communities. Low-oxygen events associated with massive algal blooms can lead to large-scale stranding of rock lobsters, and mass mortalities of other marine biota and fish (Newman & Pollock 1974; Matthews & Pitcher 1996; Pitcher 1998; Cockcroft *et al.* 2000). Very recently, in March 2008, a series of red tide or algal blooms dominated by the (non-toxic) dinoflagellate *Ceratium furca* occurred along the central Namibian coast (MFMR 2008). These bloom formations ended in disaster for many coastal marine species and resulted in what was possibly the largest rock lobster walkout in recent memory (Figure 10). Other fish mortalities included those of rock suckers, rock fish, sole, eels, shy sharks, and other animals such as octopuses and red bait, which were trapped in the low oxygen area below the surf zone (Louw 2008). The main cause for these mortalities and walkouts is oxygen starvation that results from the decomposition of huge amounts of organic matter. The blooms developed during a time where high temperatures combined with a lack of wind. These anoxic conditions were further exacerbated by the release of hydrogen sulphide - which is highly toxic to most marine organisms. Algal blooms usually occur during summer-autumn (February to April) but can also develop in winter during the 'bergwind periods', when similar warm windless conditions occur for extended periods.



Figure 10: 'Walk-outs' and mass mortalities of rock lobsters at the central Namibian coast (Image source: Louw 2008).

3.2.11 Sulphur Eruptions

Closely associated with seafloor hypoxia, particularly off central Namibia between Cape Cross and Conception Bay, is the generation of toxic hydrogen sulphide and methane within the organically-rich, anoxic muds following decay of expansive algal blooms. Under conditions of severe oxygen depletion, hydrogen sulphide (H_2S) gas is formed by anaerobic bacteria in anoxic seabed muds (Brüchert *et al.* 2003). This is periodically released from the muds as 'sulphur eruptions', causing upwelling of anoxic water and formation of surface slicks of sulphur discoloured water (Emeis *et al.* 2004), and even the temporary formation of floating mud islands (Waldron 1901). Such eruptions are accompanied by a characteristic pungent smell along the coast and the sea takes on a lime green colour (Figure 11). These eruptions strip dissolved oxygen from the surrounding water column. Such complex chemical and biological processes are often associated with the occurrence of harmful algal blooms, causing large-scale mortalities to fish and crustaceans (see above).

Sulphur eruptions have been known to occur off the Namibian coast for centuries (Waldron 1901), and the biota in the area are likely to be naturally adapted to such pulsed events, and to subsequent hypoxia. However, satellite remote sensing has recently shown that eruptions occur more frequently, are more extensive and of longer duration than previously suspected, and that resultant hypoxic conditions last longer than thought (Weeks *et al.* 2004).



Figure 11: Satellite image showing discoloured water offshore the Namib Desert resulting from a nearshore sulphur eruption (satellite image source: www.intute.ac.uk). Inset shows a photograph taken from shore at Sylvia Hill, north of Lüderitz, during such an event in March 2002 (photograph by J. Kemper, Lüderitz).

Recently, the role of micro-organisms in the detoxification of sulphidic water was investigated by a collaborative group of German and Namibian scientists (http://www.mpi-bremen.de/Projekte_9.html; <http://idw-online.de/pages/de/news/292832>). During a research cruise in January 2004, the scientists hit upon a sulphidic water mass off the coast off Namibia covering 7,000 km² of coastal seafloor. The surface waters, however, were well oxygenated. In the presence of oxygen, sulphide is oxidized and transformed into non-toxic forms of sulphur. Surprisingly though, there was an intermediate layer in the water column, which contained neither hydrogen sulphide nor oxygen. Further investigation indicated that sulphide diffusing upwards from

the anoxic bottom water is consumed by autotrophic denitrifying bacteria below the oxic zone. The intermediate water layer is the habitat of detoxifying microorganisms, which by using nitrate transform sulphide into finely dispersed particles of sulphur that are non-toxic. Thus, the microorganisms create a buffer zone between the toxic deep water and the oxygenated surface waters. These results, however, also suggest that animals living on or near the seafloor in coastal waters may be affected by sulphur eruptions more often than previously thought. Up to now, sulphidic water masses were monitored with the help of satellites, taking pictures of the sea surface while orbiting the earth, as they show up as whitish/turquoise discolorations of surface water (Figure 11). However, many of these sulphidic events may go unnoticed by satellite because bacteria consume the hydrogen sulphide before it reaches the surface.

5.3 Biological Environment

Biogeographically the central Namibian coastline falls into the warm-temperate Namib Province which extends northwards from Lüderitz into southern Angola (Emanuel *et al.* 1992). The coastal, wind-induced upwelling characterising the Namibian coastline, is the principle physical process which shapes the marine ecology of the central Benguela region.

The coastline of central Namibia is dominated by sandy beaches, with rocky habitats being represented only by occasional small rocky outcrops. Consequently, marine ecosystems along the coast comprise a limited range of habitats that include:

- sandy intertidal and subtidal substrates,
- intertidal rocky shores and subtidal reefs, and
- the water body.

The benthic communities within these habitats are generally ubiquitous throughout the southern African West Coast region, being particular only to substratum type, wave exposure and/or depth zone. They consist of many hundreds of species, often displaying considerable temporal and spatial variability. The biological communities 'typical' of each of these habitats are described briefly below, focussing both on dominant, commercially important and conspicuous species, as well as potentially threatened or sensitive species, which may be affected by the proposed project.

5.3.1 Rocky Habitats and Biota

Intertidal Rocky Shores

The central and northern coasts of Namibia are bounded to the east by the Namib Desert and are characterised primarily by gravel plains and shifting dunes. In common with most semi-exposed to exposed coastlines on the southern African west coast, the rocky shores that occur in the region are strongly influenced by sediments, and include considerable amounts of sand intermixed with the benthic biota. This intertidal mixture of rock and sand is referred to as a mixed shore, and constitutes 40 % of the coastline between the Kunene River and Walvis Bay (Bally *et al.* 1984). In the study area, mixed shores are limited to small low-shore outcrops that are exposed only at low water spring, which alternate with stretches of low-shore platform reefs and extensive pebble and sandy beaches.

Typically, the intertidal area of rocky shores can be divided into different zones according to height on the shore. Each zone is distinguishable by its different biological communities, which is largely a result of the different exposure times to air. The level of wave action is particularly important on the low shore. Generally, biomass is greater on exposed shores, which are dominated by filter-feeders. Sheltered shores support lower biomass, and algae form a large portion of this biomass (McQuaid & Branch 1984; McQuaid *et al.* 1985).

Mixed shores incorporate elements of the trophic structures of both rocky and sandy shores. As fluctuations in the degree of sand coverage are common (often adopting a seasonal affect), the fauna and flora of mixed shores are generally impoverished when compared to more homogenous shores. The macrobenthos is characterized by sand-tolerant species whose lower limits on the shore are determined by their abilities to withstand physical smothering by sand (Daly & Mathieson 1977;

Dethier 1984; van Tamelen 1996). The rocky shores along the coastline of the salt works appear to be heavily influenced by mobile sediments as large expanses of rock are barren of biota and appear scoured. Patchy dominance in the mid- and low-shore by ephemeral green algae (*Ulva* spp., *Cladophora* spp.) also suggest that these shores are periodically smothered by sands, as these algae proliferate as soon as sediments are eroded away.

The published data on rocky intertidal biota is restricted to the areas south of Lüderitz (Penrith & Kensley 1970a; Pulfrich *et al.* 2003a, 2003b; Pulfrich 2004b, 2005, 2006, 2007a; Clark *et al.* 2004, 2005, 2006; Pulfrich & Atkinson 2007), and north of Rocky Point (Penrith & Kensley 1970b; Kensley & Penrith 1980), with only a single published study documenting the area between Walvis Bay and Swakopmund (Nashima 2013). The information sourced from these publications, is complemented by unpublished data on rocky biota in the Wlotzkasbaken area supplied by MFMR (B. Currie, MFMR, unpublished data), an unpublished student report on invertebrate macrofauna occurring at three shores between Walvis Bay and Swakopmund (Ssemakula 2010) and visual observations by the author.

Typical species in the high shore include the tiny snail *Afrolittorina knysnaensis*, the false limpet *Siphonaria capensis*, the limpet *Scutellastra granularis*, and often dense stands of the barnacle *Chthamalus dentatus*. Further down the shore the mytilid mussels, *Semimytilus algosus*, *Choromytilus meridionalis*, and *Perna perna* occur. The invasive alien Mediterranean mussel *Mytilus galloprovincialis* is also present. Foliose algae are represented primarily by the red algae *Caulacanthus ustulatus*, *Ceramium* spp., *Plocamium* spp. and *Mazzaella capensis* and the ephemeral green algae *Ulva* spp. and *Cladophora* spp (Figure 12). In sand influenced areas the sand-tolerant algae *Nothogenia erinacea* and *Gelidium capense* and the anemone *Aulactinia reynaudi* also occur (A. Pulfrich, pers. obs.). The species encountered at the rocky outcrops in the study area were similar to those recorded from rocky intertidal areas in southern Namibia, and further to the north (see references above).



Figure 12: Intertidal rocky communities in the vicinity of the proposed desalination plant area showing intertidal zonation (left) and inundation by mobile sediments (right).

Although not directly harbouring any rare faunal or floral species, rocky intertidal shores are food-rich habitats for seabirds and wetland birds, attracting higher numbers of birds than the surrounding sandy beaches. Rocky intertidal fauna most sensitive to disturbance are the large limpet species. They tend to be the first ones eliminated by disturbance and the last to recover because of possible

narrow tolerance limits to changes in environmental conditions. They act as keystone species on rocky shore, controlling the abundance of foliose algae and hence many other species (Branch 1981).

Rocky Subtidal Reefs

Reports on the benthic biota of nearshore reefs are restricted primarily to research undertaken in the vicinity of Lüderitz (Beyers 1979; Tomalin 1995; Pulfrich 1998; Pulfrich & Penney 1998, 1999, 2001), and information on rocky subtidal habitats in central Namibia is lacking. No scientific surveys have been undertaken of rocky subtidal habitats in the study area, and no information exists on the faunal and floral communities (J. Basson, MFMR, pers. com.).

A hydrographical and geophysical survey conducted in the area indicates that the area is characterised by gently sloping, low-relief rock outcrops intersected by sandy gullies and depressions (CSIR 2008). The flat and featureless nature of the reefs suggests that they may intermittently be covered by a veneer of unconsolidated sediments. Although kelp occurs sparsely for up to 100 m offshore, the benthic communities inhabiting these reefs can be expected to be dominated by sand-tolerant and deposit feeding species.

As a component of the Marine Specialist Study for the NamWater Desalination Plant EIA, a diving survey with the purpose of investigating the sea floor communities in the vicinity of the proposed brine discharge points was conducted (Pulfrich & Steffani 2008). Unfortunately only limited information on the benthic communities in the area could be gathered due to poor underwater visibility, however, it was ascertained that the seabed in the area was primarily bedrock covered by sand of various thickness. Benthic organisms present included tube worms, which had constructed compact sandy reefs of 0.75 - 1.0 m in diameter and up to 0.6 m in height, inhabited by various rocky bottom species including polychaetes, amphipods, isopods, rock boring bivalves and sea anemones. Sparse clumps of large mussels (*Perna perna*) were interspersed among the tube-worm colonies. Rocky outcrops or larger boulders were densely covered by red filamentous and foliose algae, with clumps of very large *Perna* (up to 135 mm in length) occurring between the algal patches. The predatory gastropod *Thais haemastoma*, which apparently can occur in large numbers, was also recorded.

As part of the proposed Rössing Uranium Desalination Plant project transects perpendicular to the shore were swum by divers in four localities between the Outfall 1 and Outfall 5 alternatives with the objective of identifying seabed features and their associated marine communities. These investigations identified that rocks protruding through the sand veneer were typically covered in encrusting coralline algae, with foliose species represented by the red algae *Rhodymenia obtusa*, *Rhodymenia natalensis*, *Ceramium capense* and *Polyopes constrictus*, and with *Hypnea ecklonii* and *Carpoblepharis flaccida* growing epiphytically on the canopy-forming kelps *Laminaria pallida*. Green algae were primarily represented by *Cladophora flagelliformis*.

5.3.2 Sandy Substrate Habitats and Biota

The benthic biota of soft bottom substrates constitutes invertebrates that live on (epifauna), or burrow within (infauna), the sediments, and are generally divided into megafauna (animals >10 mm), macrofauna (>1 mm) and meiofauna (<1 mm).

Intertidal Sandy Beaches

Sandy beaches are one of the most dynamic coastal environments. The composition of their faunal communities is largely dependent on the interaction of wave energy, beach slope and sand particle size, which is called beach morphodynamics. Three morphodynamic beach types are described: dissipative, reflective and intermediate beaches (McLachlan *et al.* 1993, Defeo & McLachlan 2005). Generally, dissipative beaches are relatively wide and flat with fine sands and high wave energy. Waves start to break far from the shore in a series of spilling breakers that 'dissipate' their energy along a broad surf zone. This generates slow swashes with long periods, resulting in less turbulent conditions on the gently sloping beach face. These beaches usually harbour the richest intertidal faunal communities. Reflective beaches have low wave energy, and are coarse grained (>500 µm

sand) with narrow and steep intertidal beach faces. The relative absence of a surf zone causes the waves to break directly on the shore causing a high turnover of sand. The result is depauperate faunal communities. Intermediate beach conditions exist between these extremes and have a very variable species composition (McLachlan *et al.* 1993; Jaramillo *et al.* 1995). This variability is mainly attributable to the amount and quality of food available. Beaches with a high input of e.g. kelp wrack have a rich and diverse drift-line fauna, which is sparse or absent on beaches lacking a drift-line (Branch & Griffiths 1988; Field & Griffiths 1991).

In the area between Walvis Bay and the Kunene River, beaches make up 44 % of the coastline (Bally *et al.* 1984). A number of studies have been conducted on sandy beaches in central Namibia, including Sandwich Harbour (Stuart 1975; Kensley & Penrith 1977), the Paaltjies (McLachlan 1985) and Langstrand (McLachlan 1985, 1986; Donn & Cockcroft 1989), beaches near Walvis Bay and Cape Cross (Donn & Cockcroft 1989), and recently a beach survey was conducted near Wlotzkasbaken as part of the baseline study for the Areva desalination plant (Pulfrich 2007b). A further study by Tarr *et al.* (1985) investigated the ecology of three beaches further north on the Skeleton Coast. The results of these studies are summarised below.

Most beaches on the central Namibian coastline are open ocean beaches receiving continuous wave action. They are classified as exposed to very exposed on the 20-point exposure rating scale (McLachlan 1980), and intermediate to reflective and composed of well-sorted medium to coarse sands. The beaches tend to be characterised by well-developed berms, and are well-drained and oxygenated.

Numerous methods of classifying beach zonation have been proposed, based either on physical or biological criteria. The general scheme proposed by Branch & Griffiths (1988) is used below, supplemented by data from central Namibian beach studies (Stuart 1975; Kensley & Penrith 1977; McLachlan 1985, 1986; Donn 1986; Donn & Cockcroft 1989) (Figure 13).

Supralittoral zone - The supralittoral zone is situated above the high water spring (HWS) tide level, and receives water input only from large waves at spring high tides or through sea spray. The supralittoral is characterised by a mixture of air breathing terrestrial and semi-terrestrial fauna, often associated with and feeding on kelp deposited near or on the driftline. Terrestrial species include a diverse array of beetles and arachnids and some oligochaetes, while semi-terrestrial fauna include the oniscid isopod *Tylos granulatus*, and the talitrid amphipod (Amphipoda, Crustacea) *Talorchestia quadrispinosa*. Community composition depends on the nature and extent of wrack, in addition to the physical factors structuring beach communities, as described above.

Midlittoral zone - The intertidal zone, also termed the mid-littoral zone, has a vertical range of about 2 m. This mid-shore region is characterised by the cirrolanid isopods *Pontogeloides latipes*, *Eurydice (longicornis=) kensleyi*, and *Excirrolana natalensis*, the deposit-feeding polychaete *Scolecopsis squamata* (Polychaeta) and various species of the polychaete genus *Lumbrineris*, and the amphipods of the families Lysianassidae and Phoxocephalidae. In some areas, juvenile and adult sand mussels *Donax serra* (Bivalvia, Mollusca) may also be present in considerable numbers.

Inner turbulent zone - The inner turbulent zone extends from the low water spring tide level to about -2 m depth, and is characterised by highly motile species. The benthic planktic mysids *Gastrosaccus namibensis* and *G. psammodytes* (Mysidacea, Crustacea), the ribbon worm *Cerebratulus fuscus* (Nemertea) and the cumacean *Cumopsis robusta* (Cumacea) are typical of this zone, although they generally extend partially into the midlittoral above. In areas where a suitable swash climate exists, the gastropod *Bullia digitalis* (Gastropoda, Mollusca) may also be present in considerable numbers.

Transition zone - The transition zone spans approximately 2-3 m depth and marks the area to which the break point might move during storms. Extreme turbulence is experienced in this zone, and as a consequence this zone typically harbours the lowest diversity on sandy beaches. Typical fauna of this zone include the polychaetes *Nephtys hombergi*, *Diopatra neopolitana* and *Glycera convoluta*, nemertean worms, amphipods such as *Urothoe elegans* and *Mandibulophoxus stimpsoni*, and the isopods *Cirolana hirtipes* and *Eurydice (longicornis=) kensleyi*.

Outer turbulent zone - Below 3 m depth extends the outer turbulent zone, where turbulence is significantly decreased and which is marked by a sudden increase in species diversity and biomass. In addition to the polychaetes found in the transition zone, other polychaetes in this zone include *Diopatra neopolitana* and *Glycera convoluta*. The abundance of nemertean worms increases significantly from that in the transition zone. Amphipods such as *Urothoe elegans* and *Mandibulophoxux stimpsoni* are also more abundant, as are the isopods *Cirolana hirtipes* and *Pontogeloides latipes*, the mysid *G. namibensis*, the decapods *Diogenes extricatus* and *Ogyrides saldanhae*, and the three spot swimming crab *Ovalipes punctatus*, as well as the gastropods *Bullia laevis* and *Natica forata*.

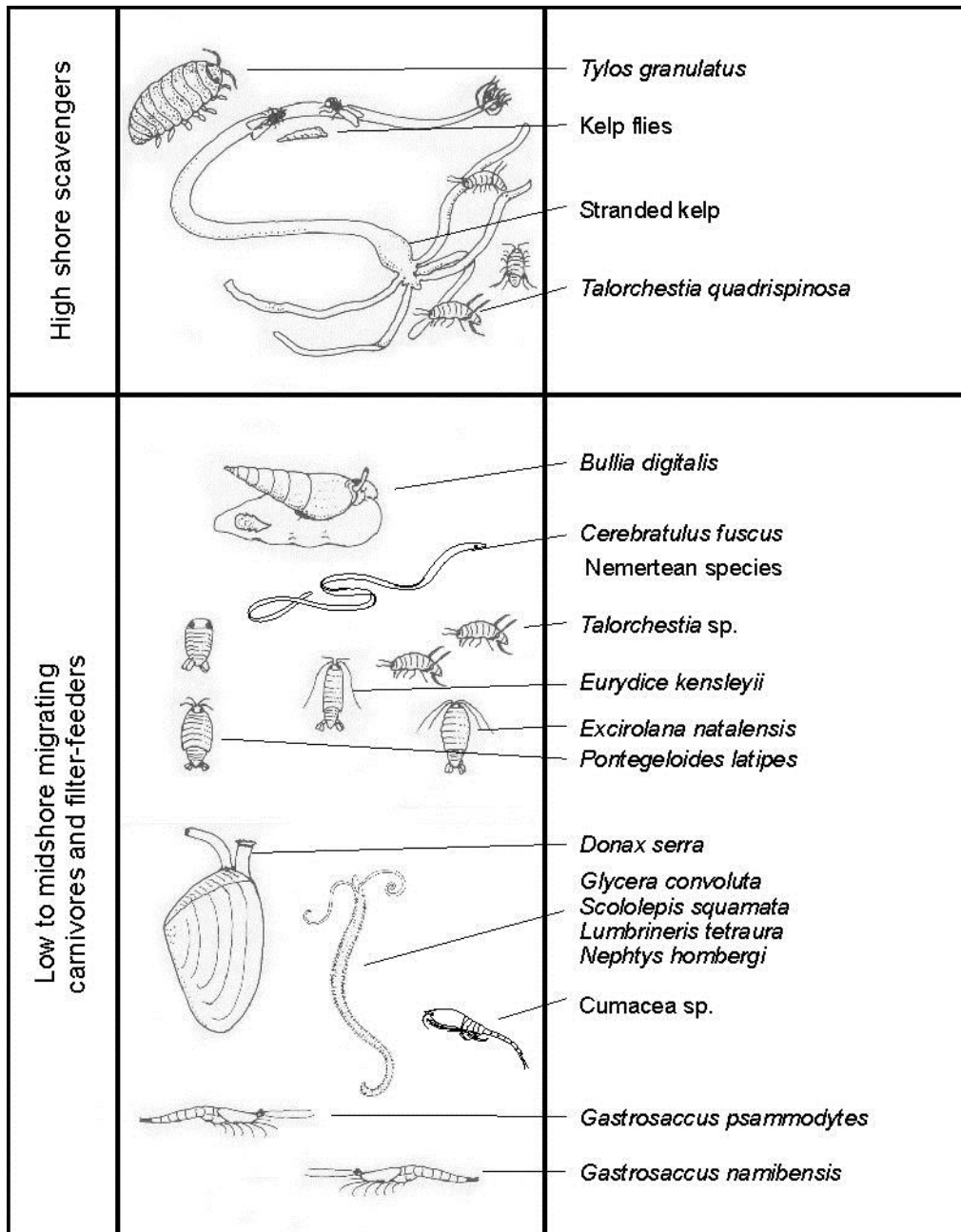


Figure 13: Schematic representation of the West Coast intertidal beach zonation (adapted from Branch & Branch 1981). Species commonly occurring on the central Namibian beaches are listed.

The surf zone in the study area is rich in phytoplankton (primarily dinoflagellates and diatoms) and zooplankton. Particulate organic matter is commonly deposited on the beaches as foam and scum. The organic matter, both in suspension and deposited on the sand, is thought to represent the main food input into these beaches, thereby accounting for the dominance of filter-feeders in the macrofaunal biomass (McLachlan 1985).

Most of the macrofaunal species recorded from beaches in central Namibia are ubiquitous throughout the biogeographic province, and **no rare or endangered species are known**. The invertebrate communities are similar to those recorded from beaches in southern Namibia (McLachlan & De Ruyck 1993; Nel *et al.* 1997; Meyer *et al.* 1998; Clark & Nel 2002; Clark *et al.* 2004; Pulfrich 2004a; Clark *et al.* 2005, 2006; Pulfrich & Atkinson 2007; Pulfrich *et al.* 1013). The beaches are characterised by a relatively depauperate invertebrate fauna, both with regard to species diversity and biomass, which is typical of high-energy west coast beaches.

Subtidal Sandy Habitats

In the subtidal region, the structure and composition of benthic soft bottom communities is primarily a function of water depth and sediment grain size, but other factors such as current velocity, organic content, and food abundance also play a role (Snelgrove & Butman 1994; Flach & Thomsen 1998; Ellingsen 2002).

With the exception of numerous studies on the benthic fauna of Walvis Bay lagoon (Kensley 1978; CSIR 1989, 1992; COWI 2003; Tjipute & Skuuluka 2006), there is a noticeable scarcity of published information on the subtidal soft sediment biota along the rest of the central Namibian coast. The only reference sourced was that of Donn & Cockcroft (1989) who investigated macrofauna to 5 m depth at Langstrand (see description for outer-turbulent zone above). In general, almost no scientific work on subtidal benthic communities has been done in the vicinity of the study area, or within the general region (J. Basson, MFMR, pers. comm.) and no further information could be obtained.

Beyond the outer turbulent zone to 80 m depth, species diversity, abundance and biomass generally increases with communities being characterised equally by polychaetes, crustaceans and molluscs. The midshelf mudbelt is a particularly rich benthic habitat where biomass can attain 60 g/m² dry weight (Christie 1974; see also Steffani 2007b). The comparatively high benthic biomass in this mudbelt region represents an important food source to carnivores such as the mantis shrimp, cephalopods and demersal fish species (Lane & Carter 1999). In deeper water beyond this rich zone biomass declines to 4.9 g/m² at 200 m depth and then is consistently low (<3 g/m²) on the outer shelf (Christie 1974).

Typical species occurring at depths of up to 60 m included the snail *Nassarius* spp., the polychaetes *Orbinia angrapequensis*, *Nephtys sphaerocirrata*, several members of the spionid genera *Prionospio*, and the amphipods *Urothoe grimaldi* and *Ampelisca brevicornis*. The bivalves *Tellina gilchristi* and *Dosinia lupinus orbigny* are also common in certain areas. All these species are typical of the southern African West Coast (Christie 1974; 1976; McLachlan 1986; Parkins & Field 1998; Pulfrich & Penney 1999b; Goosen *et al.* 2000; Steffani & Pulfrich 2004a; 2007; Steffani, unpublished data) (Figure 14).

Whilst many empirical studies related community structure to sediment composition (e.g. Christie 1974; Warwick *et al.* 1991; Yates *et al.* 1993; Desprez 2000; van Daltsen *et al.* 2000), other studies have illustrated the high natural variability of soft-bottom communities, both in space and time, on scales of hundreds of metres to metres (e.g. Kenny *et al.* 1998; Kendall & Widdicombe 1999; van Daltsen *et al.* 2000; Zajac *et al.* 2000; Parry *et al.* 2003), with evidence of mass mortalities and substantial recruitments (Steffani & Pulfrich 2004a). It is likely that the distribution of marine communities in the mixed deposits of the coastal zone is controlled by complex interactions between physical and biological factors at the sediment-water interface, rather than by the granulometric properties of the sediments alone (Snelgrove & Butman 1994; Seiderer & Newell 1999). For example, off central Namibia it is likely that periodic intrusion of low oxygen water masses is a major cause of this variability (Monteiro & van der Plas 2006; Pulfrich *et al.* 2006). Although there is a poor understanding of the responses of local continental shelf macrofauna to low

oxygen conditions, it is safe to assume that in areas of frequent oxygen deficiency the communities will be characterised by species able to survive chronic low oxygen conditions, or colonising and fast-growing species able to rapidly recruit into areas that have suffered complete oxygen depletion. Local hydrodynamic conditions, and patchy settlement of larvae, will also contribute to small-scale variability of benthic community structure.

It is evident that an array of environmental factors and their complex interplay is ultimately responsible for the structure of benthic communities. Yet the relative importance of each of these factors is difficult to determine as these factors interact and combine to define a distinct habitat in which the animals occur. However, it is clear that water depth and sediment composition are two of the major components of the physical environment determining the macrofauna community structure off southern Namibia (Steffani & Pulfrich 2004a, 2004b, 2007; Steffani 2007a, 2007b, 2009a, 2009b, 2009c, 2010).



Figure 14: Benthic macrofaunal genera commonly found in nearshore sediments include: (top: left to right) *Ampelisca*, *Prionospio*, *Nassarius*; (middle: left to right) *Callianassa*, *Orbinia*, *Tellina*; (bottom: left to right) *Nephtys*, hermit crab, *Bathyporeia*.

5.3.3 Pelagic Communities

The pelagic communities are typically divided into plankton and fish, and their main predators, marine mammals (seals, dolphins and whales), seabirds and turtles. Seabirds are dealt with in a separate specialist study and will thus not be discussed further here.

Plankton

Plankton is particularly abundant in the shelf waters off Namibia, being associated with the upwelling characteristic of the area. Plankton range from single-celled bacteria to jellyfish of 2-m diameter, and include bacterio-plankton, phytoplankton, zooplankton, and ichthyoplankton (Figure 15).

Off the Namibian coastline, phytoplankton are the principle primary producers with mean annual productivity being comparatively high at 2 g C/m²/day. The phytoplankton is dominated by diatoms, which are adapted to the turbulent sea conditions. Diatom blooms occur after upwelling events, whereas dinoflagellates are more common in blooms that occur during quiescent periods, since they can grow rapidly at low nutrient concentrations (Barnard 1998). A study on phytoplankton in the surf zone off two beaches in the Walvis Bay and Cape Cross area showed relatively low primary production values of only 10-20 mg C/m²/day compared to those from oceanic waters. This was attributed to the high turbidity in this environment (McLachlan 1986). In the surf zone, diatoms and dinoflagellates are nearly equally important members of the phytoplankton, and some silicoflagellates are also present. Characteristic species belong to the genus *Gymnodinium*, *Peridinium*, *Navicula*, and *Thalassiosira* (McLachlan 1986).

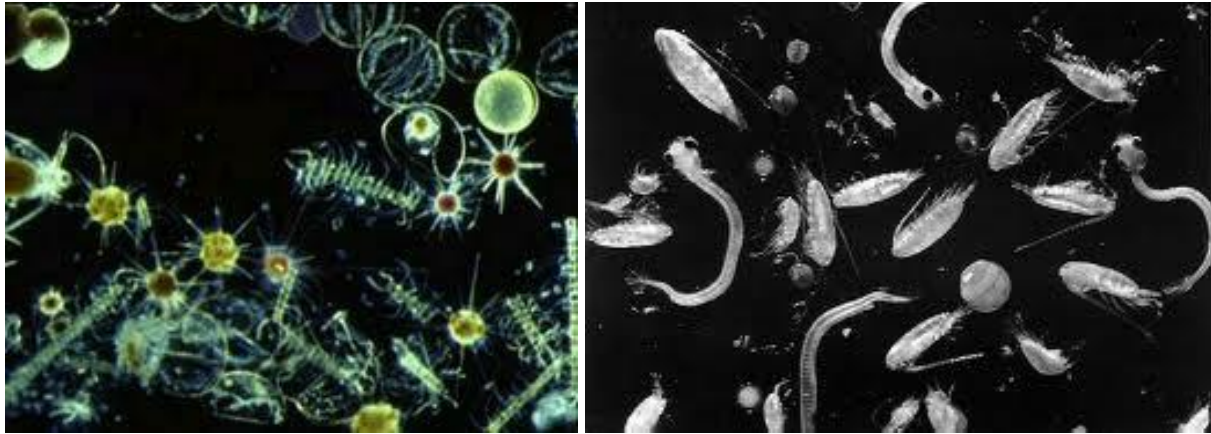


Figure 15: Phytoplankton (left, photo: hymagazine.com) and zooplankton (right, photo: mysciencexbox.org) is associated with upwelling cells on the Namibian shelf.

Namibian zooplankton reaches maximum abundance in a belt parallel to the coastline and offshore of the maximum phytoplankton abundance. Samples collected over a full seasonal cycle (February to December) along a 10 to 90-nautical-miles transect offshore Walvis Bay showed that the mesozooplankton (<2 mm body width) community included egg, larval, juvenile and adult stages of copepods, cladocerans, euphausiids, decapods, chaetognaths, hydromedusae and salps, as well as protozoans and meroplankton larvae (Hansen *et al.* 2005). Copepods are the most dominant group making up 70-85% of the zooplankton. The four dominant calanoid copepod species, in order of abundance, are *M. lucens*, *C. carinatus*, *R. nasutus* and *Centropages* spp. During the period of intense upwelling, the two herbivorous species, *C. carinatus* and *R. nasutus*, increase in abundance inshore, leading to a shift in dominance from *C. carinatus* to *M. lucens* with increasing distance offshore. Seasonal patterns in copepod abundance, with low numbers during autumn (March-June) and increasing considerably during winter/early summer (July-December), appear to be linked to the period of strongest coastal upwelling in the northern Benguela (May-December), allowing a time lag of about 3-8 weeks, which is required for copepods to respond and build up large populations (Hansen *et al.* 2005). This suggests close coupling between hydrography, phytoplankton and zooplankton. Timonin *et al.* (1992) described three phases of the upwelling cycle (quiescent, active and relaxed upwelling) in the northern Benguela, each one characterised by specific patterns of zooplankton abundance, taxonomic composition and inshore-offshore distribution. It seems that zooplankton biomass closely follows the changes in upwelling intensity and phytoplankton standing crop. Consistently higher biomass of zooplankton occurs offshore to the west and northwest of Walvis Bay (Barnard 1998).

Ichthyoplankton constitutes the eggs and larvae of fish. As the preferred spawning grounds of numerous commercially exploited fish species are located off central and northern Namibia (Figure 16), their eggs and larvae form an important contribution to the ichthyoplankton in the region.

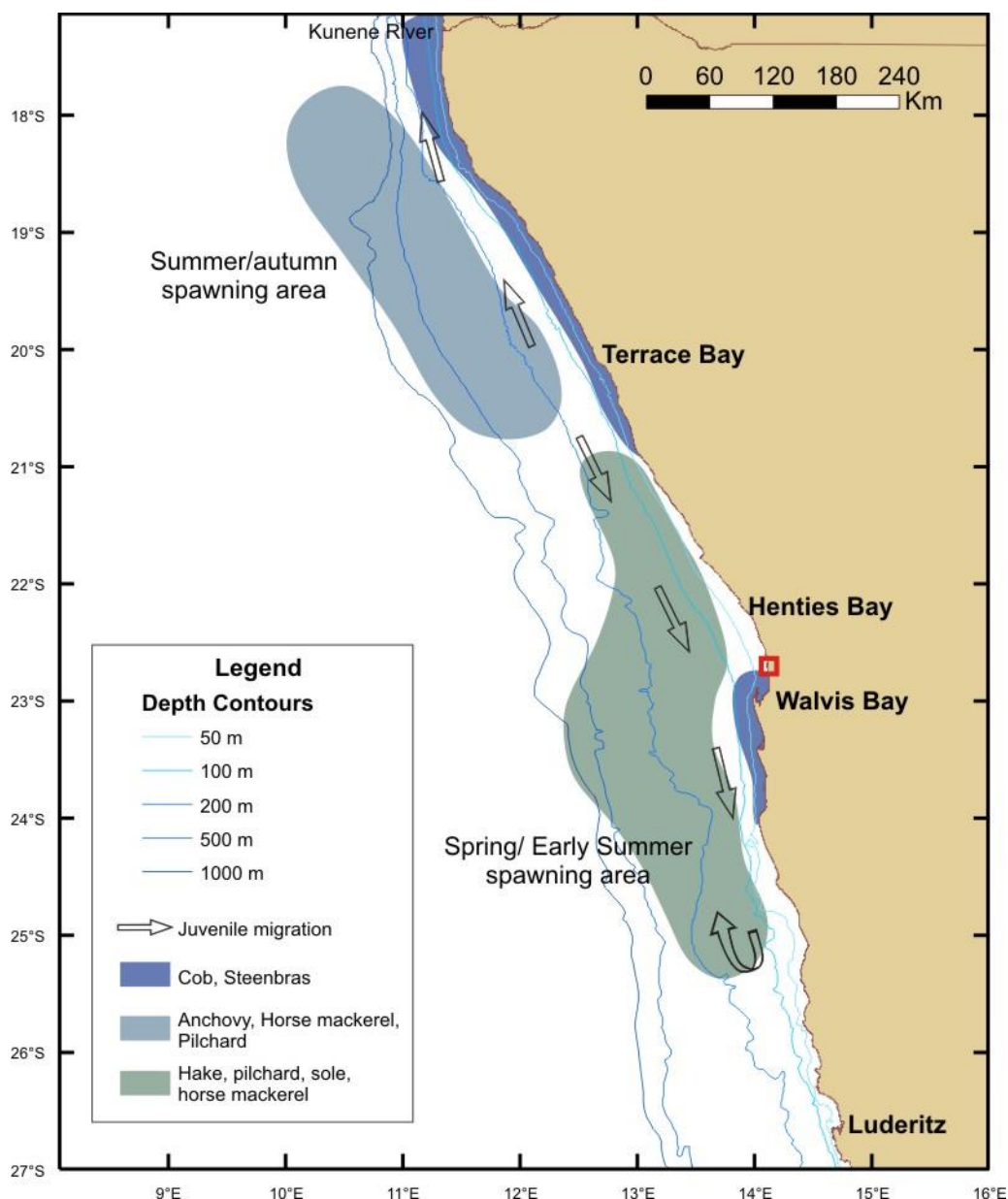


Figure 16: Major spawning areas in the central Benguela region (adapted from Cruikshank 1990) in relation to the study area (red rectangle - not to scale).

Fish

The surf zone and outer turbulent zone habitats of sandy beaches are considered to be important nursery habitats for marine fishes (Modde 1980; Lasiak 1981; Kinoshita & Fujita 1988; Clark *et al.* 1994). However, the composition and abundance of the individual assemblages seems to be heavily dependent on wave exposure (Blaber & Blaber 1980; Potter *et al.* 1990; Clark 1997a, b). Surf zone fish communities off the coast of southern Namibia have been studied by Clark *et al.* (1998) and Meyer *et al.* (1998), who reported only five species occurring off exposed and very exposed beaches, these being southern mullet/harders (*Liza richardsonii*), white stumpnose (*Rhabdosargus globiceps*), False Bay klipfish (*Clinus latipennis*), Super klipvis (*C. superciliosus*) and galjoen (*Dichistius capensis*). Linefish species common off the central Namibian coastline include snoek (*Thyrsites atun*), silver kob (*Argyrosomus inodorus*), West Coast Steenbras (*Lithognathus aureti*), blacktail (*Diplodus sargus*), white stumpnose, Hottentot (*Pachymetopon blochii*) and galjoen

(*Dichistius capensis*). From the surf zone off Langstrand beach near Walvis Bay, McLachlan (1986) recorded galjoen, West Coast steenbras, flathead mullet (*Mugil cephalus*), and southern mullet. Off Cape Cross only two species were recorded, these being sandsharks (*Rhinobatos annulatus*) and West Coast steenbras.

No systematic surveys of the fish fauna of Walvis Bay, the lagoon and surrounding areas appear to have been undertaken. Glasson & Branch (1997) refer to the presence of the sandshark in Walvis Bay. Both mullet species enter the lagoon in large shoals, often pursued by flocks of Great White Pelicans or Cape Cormorants. Other fish species reported as occurring in the lagoon include silver kob, barbel (*Galeichthys feliceps*) and west coast steenbras. However, angling competition records for the lagoon indicate that no bony fishes have been caught since 2000 (Walvis Bay Angling Club), with only sandsharks, bull rays (*Pteromyllacus bovinus*), blue sting rays (*Dasyatis pastinaca*) and hound sharks (*Mustelis mustelis*) being caught.

The biological, behavioural and life-history characteristics of the three most important linefish species in Namibian coastal waters are summarised below.

Silver kob *Argyrosomus inodorus* are distributed from northern Namibia to the warm temperate / subtropical transition zone on South Africa's east coast (Griffiths & Heemstra 1995). Four stocks have been identified, one in Namibia, with its core distribution from Cape Frio in the north to Meob Bay in the south, a distance of 850 km (Kirchner 2001). Maturity is reached at a length of 35 cm and age of 1.5 years with a maximum recorded size of 36 kg (Kirchner *et al.* 2001). Spawning occurs throughout the year but mostly in the warmer months from October to March when water temperatures are above 15°C and large adult fish occur in the nearshore, particularly in the identified spawning areas of Sandwich Harbour and Meob Bay. Adults are migratory whereas juveniles are resident in the surf zone.

The Namibian stock of *A. inodorus* is exploited by the commercial linefishery (deck and skiboats) and recreational shore angling with, until recently, a mean annual catch of 500 t and 350 t respectively. There is also a small recreational boat fishery (Kirchner 2001). The stock is regarded as overexploited and near collapse with less than 25% of pristine spawner biomass remaining. The availability of *A. inodorus* and other fish species to shore and boat fishers is driven by environmental conditions. For example, strong south-westerly winds, large swells and upwelling all have a negative impact on catches. Warm-water events and sulphur eruptions inhibit feeding and the catchability of most species (Holtzhausen *et al.* 2001).

West coast dusky kob *Argyrosomus coronus* are distributed from northern Namibia to northern Angola (Griffiths & Heemstra 1995), but do occur as far south as St Helena Bay in South Africa (Lamberth *et al.* 2008). Maturity is reached at a total length of 87 cm and 4.5 years of age and a maximum size of 80 kg attained (Potts *et al.* 2012). Early juveniles frequent muddy sediments in 50-100 m depth, moving inshore once they reach 300 mm total length. These juveniles and adolescents are resident in the nearshore, and are especially abundant in the turbid plume off the Cunene River Mouth and in selected surf zones of northern and central Namibia (Potts *et al.* 2010). The adults are migratory according to the movement of the Angola-Benguela frontal zone, moving northwards as far as Gabon in winter and returning to southern Angola in spring where spawning occurs in the offshore (Potts *et al.* 2010).

In Angola and Namibia, *A. coronus* are exploited by the shore- and boat-based commercial, artisanal and recreational line fisheries. The Angolan beach-seine, gillnet and purse-seine fisheries also land this species. Overexploitation in its northern range is likely exacerbated by a distributional shift of adult fish out of Angolan waters. Ten years ago, the ratio of *A. inodorus* to *A. coronus* in the Namibian fishery was 10:1 (Kirchner & Beyer 1999) compared to 10:15 in the present day (Potts *et al.* in prep). This is largely due to a distributional shift southwards also evidenced by a 58% reduction in relative abundance and a 27% reduction in mean length in Angolan waters (Potts *et al.* in prep). The overall driver is thought to be warmer coastal waters in the northern Benguela coastal zone.

The populations of both kob species are under stress from fishing, climate change, distributional shifts and an increase in inter-specific interactions. Inter-specific stress has also become a factor. *A. inodorus* and *A. coronus* now overlap in distribution and hybridisation, which may at least partly

be due to a stress-induced breakdown in mate recognition, has occurred. In fish, hybridisation is usually associated with increased resistance to disease and physiological tolerance of environmental stresses, and often allows species to expand their ranges to invade new niches. However, molecular support for potential reduced fitness in hybridized fish under environmental stress exists (David *et al.* 2004), providing a plausible explanation for the relatively rare occurrence of interspecies hybridisation in sympatric environments. Behavioural and biological responses such as distributional shifts and hybridisation make it clear that some population thresholds have already been reached and that even low-level anthropogenic forcings may precipitate further change.

Similar to the kob species described above, white steenbras *Lithognathus lithognathus* and west coast steenbras *Lithognathus aureti* are sister species and sympatric from St Helena Bay to the Orange River Estuary. White steenbras occur from the Orange River to the Umtamvuna River on South Africa's eastern seaboard, but spawning habitat appears to be restricted to less than 50 hectares throughout its range (Sink *et al.* 2011). Adults undertake an annual spawning migration to the edge of the species's distribution on the east coast. There is, however, circumstantial evidence for the "extinction" of a separate west coast spawning population due to overexploitation in the last century (Lamberth *et al.* 2011).

West coast steenbras *Lithognathus aureti* are endemic to the west coast of southern Africa, but rarely found outside Namibia's territorial waters (Holtzhausen 2000). However, they do occur as far south as St Helena Bay and historical abundance in South African waters is thought to have been a lot higher prior to the advent of the commercial beach-seine fishery (Lamberth *et al.* 2008). In Namibia, *L. aureti* are exploited by commercial and recreational boat-based linefishers, as well as by recreational shore-anglers with a total landed catch of approximately 600 t per annum (Holtzhausen & Mann 2000). Overexploitation in the early 1990s was arrested by the closure of the gillnet fishery for this species. Tagging studies have indicated that *L. aureti* comprise two separate closed populations; one in the vicinity of Meob Bay and one from central Namibia northwards (Holtzhausen *et al.* 2001). Spawning localities are as yet unknown but tagging evidence suggests that males migrate considerable distances in search of gravid females (Holtzhausen 2000).

The parallels between *L. aureti* and *L. lithognathus* suggest that the spawning habitat of West coast steenbras may also be limited. The bulk of the population exists in the nearshore at <10 m depth, with juveniles occurring in the intertidal surf zone (McLachlan 1986). By inference, spawning occurs in the surf zone and eggs and larvae from both populations drift northwards (Holtzhausen 2000). The fact that both populations of *L. aureti* exist entirely in the nearshore would make them susceptible to any coastal development that lies in the path of alongshore movement. Whereas juveniles occur in the surf zone throughout its range, spawning habitat may be extremely limited and has yet to be clearly identified.

Small pelagic species include the sardine/pilchard (*Sardinops ocellatus*) (Figure 17, left), anchovy (*Engraulis capensis*), chub mackerel (*Scomber japonicus*), horse mackerel (*Trachurus capensis*) (Figure 17, right) and round herring (*Etrumeus whiteheadi*). These species typically occur in mixed shoals of various sizes (Crawford *et al.* 1987), and generally occur within the 200 m contour, although they may often be found very close inshore, just beyond the surf zone. They spawn downstream of major upwelling centres in spring and summer, and their eggs and larvae are subsequently carried up the coast in northward flowing waters. Recruitment success relies on the interaction of oceanographic events, and is thus subject to spatial and temporal variability. Consequently, the abundance of adults and juveniles of these small pelagic fish is highly variable both within and between species. The Namibian pelagic stock is currently considered to be in a critical condition due to a combination of over-fishing and unfavourable environmental conditions as a result of Benguela Niños.

Since the collapse of the pelagic fisheries, jellyfish biomass has increased and the structure of the Benguelan fish community has shifted, making the bearded goby (*Sufflogobius bibarbatus*) the new predominant prey species. However, despite increased predation pressure, the gobies are thriving. Recent research has shown that gobies have a very high tolerance of low oxygen and high H₂S levels, which enables them to feed on benthic fauna within hypoxic waters during the day, and then move to oxygen-richer pelagic waters at night, when predation pressure is lower, to feed on live jellyfish (Utne-Palm *et al.* 2010; van der Bank *et al.* 2011).



Figure 17: Cape fur seal preying on a shoal of pilchards (left). School of horse mackerel (right) (photos: www.underwatervideo.co.za; www.delivery.superstock.com).

Turtles

Five of the eight species of turtle worldwide occur off Namibia (Bianchi *et al.* 1999). Turtles that are occasionally sighted off central Namibia, include the Leatherback Turtle (*Dermochelys coriacea*), the largest living marine reptile. Limited information is available on marine turtles in Namibian waters, although leatherback turtles, which are known to frequent the cold southern ocean, are the most commonly-sighted turtle species in the region. Observations of Green (*Chelonia mydas*), Loggerhead (*Caretta caretta*), Hawksbill (*Eretmochelys imbricata*) and Olive Ridley (*Lepidochelys olivacea*) turtles in the area are rare.

Leatherbacks turtles inhabit deeper waters and are considered a pelagic species, travelling the ocean currents in search of their prey (primarily jellyfish). While hunting they may dive to over 600 m and remain submerged for up to 54 minutes (Hays *et al.* 2004). Their large size allows them to maintain a constant core body temperature and consequently they can penetrate colder temperate waters.

The South Atlantic population of leatherback turtles is the largest in the world, with as many as 40,000 females thought to nest in an area centred on Gabon, yet the trajectory of this population is currently unknown (Witt *et al.* 2011). Namibia is gaining recognition as a feeding area for leatherback turtles that are either migrating through the area or undertaking feeding excursions into Namibian waters. The turtles are thought to be attracted by the large amount of gelatinous plankton in the Benguela ecosystem (Lynam *et al.* 2006). Based on tag returns from animals found dead in Namibia, these turtles are thought to come mainly from Gabonese and Brazilian nesting grounds (R. Braby, pers. comm., Namibia Coast Conservation and Management Project - NACOMA, 25 August 2010).

Although they tend to avoid nearshore areas, they may be encountered in the area around Walvis Bay between October and April when prevailing north wind conditions result in elevated seawater temperatures. Elwen & Leeney (2011) reported 21 sightings of leatherback turtles in Walvis Bay between 2009 and 2010. Anecdotal evidence suggests that sightings of leatherback turtles have been fewer in the past two years (R. Leeney, pers. comm. with tourism industry operators). Leatherback turtles have recently washed up in significant numbers on the central Namibian shore (Figure 18), with some being recorded as far south as Mining Area 1 in the Sperrgebiet (28° 27'S)(A. Pulfrich, pers. obs.). During the past five years 200 to 300 dead turtles were found (www.nacoma.org.na). The shell of a green turtle was found in Sandwich Harbour in March 2012 (NDP data).

Several anthropogenic factors threaten sea turtle populations including entanglement in fishing gear, incidental catches in fisheries, vessel strikes, ingestion of marine debris, pollution, decline of habitat along the Western Atlantic coast and loss of nesting habitat (Carr 1987; National Research Council (NRC) 1990; Lutz & Alfaro-Shulman 1991; Lutcavage *et al.* 1997; Witzell 1999; Witherington

& Martin 2000; Dwyer *et al.* 2003; James *et al.* 2005). Anthropogenic noise is also thought to be detrimental to sea turtles (Samuel *et al.* 2005), with likely effects on their behaviour and ecology.

Leatherback Turtles are listed as “Critically Endangered” worldwide by the IUCN and are in the highest categories in terms of need for conservation in CITES (Convention on International Trade in Endangered Species), and CMS (Convention on Migratory Species). Although Namibia is not a signatory of CMS, Namibia has endorsed and signed a CMS International Memorandum of Understanding specific to the conservation of marine turtles. Namibia is thus committed to conserve these species at an international level.



Figure 18: Dead Leatherback Turtle washed up at a beach north of Swakopmund, March 2008.

Marine Mammals

Marine mammals occurring off the Namibian coastline include cetaceans (whales and dolphins) and seals. The cetacean fauna of the Namibian coast comprises between 22 and 31 species (Cetus Projects 2008; Currie *et al.* 2009), the diversity reflecting both species recorded from the waters of Namibia (Williams *et al.* 1990; Rose & Payne 1991; Findlay *et al.* 1992; Griffin & Coetzee 2005) and species expected to be found in the region based on their distributions elsewhere along the southern African West coast (Best 2007; Elwen *et al.* 2011a). The diversity is comparatively high, reflecting the cool inshore waters of the Benguela Upwelling system and the occurrence of warmer oceanic water offshore of this. The species confirmed to be present in Namibian waters are listed in Table 5-2.

Of the species recorded the endemic Heaviside’s Dolphin *Cephalorhynchus heavisidii* (Figure 19, left) is found in the extreme nearshore region of the project area. Although there are no population estimates for Heaviside’s dolphins as a whole, the size of the population utilising Walvis Bay in 2009 was estimated at 505 (Elwen & Leeney 2009), and a degree of site fidelity of the species to Pelican Point was confirmed from images taken in 2008 and 2009. Sightings of this species in Walvis Bay occur mostly at Pelican Point; the few sightings in other parts of the bay occur more commonly in summer (January to March), when sightings at Pelican Point decrease, suggesting that these animals have a different primary habitat during those months. The range of the Heaviside’s dolphins in this area is unknown, although aerial surveys (Leeney in prep.) have revealed that they utilises nearshore habitat along much of the Namibian coastline including south of Walvis Bay, with a hotspot of abundance just south of Sandwich Harbour. Acoustic detections of the species at Pelican Point are most numerous during the night, decreasing to a minimum in the early afternoon

(Leeney *et al.* 2011). This pattern is likely linked with prey availability at this site. Although considered numerous in South African waters, Heaviside's dolphins are vulnerable due to their use of human-impacted coastal habitats, the small home ranges of individuals and the restricted geographic range of the species.

The bottlenose dolphin (*Tursiops truncatus*) is found in the extreme nearshore region between Lüderitz and Cape Cross (Elwen *et al.* 2011b; Leeney in prep.) (including the Sandwich Harbour lagoon), as well as offshore of the 200 m isobath along the Namibian coastline. This species has been a key element of the research conducted by the Namibian Dolphin Project in Walvis Bay, with the population in 2008 estimated (via photo-identification techniques) at 77 individuals. Since then there has been a 6-8% annual reduction in the number of animals identified in the bay (Elwen *et al.* 2011b), with 19 individuals identified in 2008 not been seen since. This suggests some degree of emigration from the population. The reduction in the population is a serious concern and suggests that the species is under pressure in at least part of its range. Roughly twice as many individuals are identified in Walvis Bay in winter than during the summer months, suggesting that other habitats are more frequently utilised during the summer. A number of mother-calf pairs have been observed in Walvis Bay between 2008 and 2011. The reef north of Bird Island has been identified as an area used by these animals primarily for resting (Elwen & Leeney 2009; Elwen *et al.* 2011b), and has informally been designated as a no-go zone for tour boats.



Figure 19: The endemic Benguela Dolphin *Cephalorhynchus heavisidii* (left) (Photo: De Beers Marine Namibia), and Southern Right whale *Eubalaena australis* (right) (Photo: www.divephotoguide.com; www.aad.gov.au).

Although common bottlenose dolphins are found worldwide, they often live in isolated populations that number up to a few hundred individuals only. If such localised populations decline due to human impacts they can potentially die out, as numbers are not supplemented by animals from elsewhere. The Namibian population is unique within the Benguela ecosystem as it occurs close inshore, with their nearest neighbours being in central Angola.

The dusky dolphin (*Lagenorhynchus obscurus*) is considered a pelagic species and often sighted by fishermen working in deeper waters. However, it is an occasional visitor to Walvis Bay, where they may beach (e.g. Elwen *et al.* 2011). Southern right-whale dolphins (*Lissodelphis peronii*) have an extremely localised year-round distribution associated with the continental shelf and the shelf-edge in the region between 24° and 28° S. A further 11 species are resident within the offshore area of the Namibian coastline in water depths of over 500 m (see Table 52). Killer whales (*Orcinus orca*) are found throughout Namibian waters and likely range along the entire coastline (Elwen & Leeney 2011). Pilot whales (*Globicephala* spp.) are commonly sighted by fishermen in considerable numbers, and have also frequently been observed during offshore seismic surveys.

Of the southern hemisphere migratory whale species, blue whales (*Balaenoptera musculus*), fin whales (*B. physalus*), sei whales (*B. borealis*), minke whales (*B. acutorostrata*), Bryde's whale (*B. edeni*) and humpback whales (*Megaptera novaeangliae*) (Figure 19, right), and two species of

Table 5-2: Cetacean species present in Namibian waters

Species name	Common name	Source
Mysticetes (baleen whales)		
<i>Eubalaena australis</i>	Southern right whale	Bianchi <i>et al.</i> 1999; Roux <i>et al.</i> 2001 ; Best 2007 ; Roux <i>et al.</i> 2010
<i>Caperea marginata</i>	Pygmy right whale	Bianchi <i>et al.</i> 1999; Best 2007 ; Leeney <i>et al.</i> in rev.
<i>Balaenoptera edonii</i>	Bryde's whale	Best 2007; NDP
<i>Balaenoptera bonaerensis</i>	Antarctic minke whale	Best 2007
<i>Balaenoptera acutorostrata subsp.</i>	Dwarf minke whale	Bianchi <i>et al.</i> 1999; Best 2007
<i>Megaptera novaeangliae</i>	Humpback whale	Bianchi <i>et al.</i> 1999; Best 2007; Barendse <i>et al.</i> 2011
<i>Balaenoptera physalus</i>	Fin whale	Bianchi <i>et al.</i> 1999; Best 2007
<i>Balaenoptera musculus</i>	Blue whale	Bianchi <i>et al.</i> 1999; Best 2007
<i>Balaenoptera borealis</i>	Sei whale	Best 2007
Odontocetes (toothed whales)		
<i>Physeter macrocephalus</i>	Sperm whale	Bianchi <i>et al.</i> 1999; Best 2007
<i>Kogia sima</i>	Dwarf sperm whale	Findlay <i>et al.</i> 1992; NDP
<i>Kogia breviceps</i>	Pygmy sperm whale	Ross 1984; Findlay <i>et al.</i> 1992; NDP
<i>Globicephala melas</i> & <i>Globicephala macrorhynchus</i>	Long-finned pilot whale & short-finned pilot whale	Findlay <i>et al.</i> 1992; Bianchi <i>et al.</i> 1999; Best 2007; NDP
<i>Cephalorhynchus heavisidii</i>	Heaviside's dolphin	Bianchi <i>et al.</i> 1999; Best 2007; Elwen & Leeney 2008
<i>Tursiops truncatus</i>	Bottlenose dolphin	Bianchi <i>et al.</i> 1999; Best 2007; Elwen & Leeney 2008
<i>Delphinus delphis</i>	Short-beaked common dolphin	Findlay <i>et al.</i> 1992; Best 2007
<i>Pseudorca crassidens</i>	False killer whale	Findlay <i>et al.</i> 1992; Best 2007; NDP
<i>Lagenorhynchus obscurus</i>	Dusky dolphin	Findlay <i>et al.</i> 1992; Bianchi <i>et al.</i> 1999 ; Best 2007
<i>Feresa attenuata</i>	Pygmy killer whale	Findlay <i>et al.</i> 1992; Best 2007
<i>Lissodelphis peronii</i>	Southern right whale dolphin	Rose & Payne 1991; Findlay <i>et al.</i> 1992; Bianchi <i>et al.</i> 1999; Best 2007
<i>Grampus griseus</i>	Risso's dolphin	Findlay <i>et al.</i> 1992
<i>Orcinus orca</i>	Killer whale/ orca	Bianchi <i>et al.</i> 1999; Findlay <i>et al.</i> 1992
<i>Ziphius cavirostris</i>	Cuvier's beaked whale	Findlay <i>et al.</i> 1992; Best 2007
<i>Hyperoodon planifrons</i>	Southern bottlenose whale	Best 2007
<i>Mesoplodon europaeus</i>	Gervais' beaked whale	Griffin & Coetzee 2005; Best 2007
<i>Mesoplodon grayi</i>	Gray's beaked whale	Findlay <i>et al.</i> 1992; Best 2007
<i>Mesoplodon layardii</i>	Layard's beaked whale	Findlay <i>et al.</i> 1992; Griffin 1998; Best 2007
<i>Mesoplodon densirostris</i>	(/strap-toothed whale) Blainville's beaked whale	Best 2007

Note: NDP refers to information collected and held, if not published, by the Namibian Dolphin Project, in reports or in strandings database.

balaenid whale, the southern right whale (*Eubalaena australis*) and the pygmy right whale (*Caperea marginata*) have been recorded in Namibian waters, primarily off the continental shelf during winter months. Humpback whales commonly have a summer distribution in polar waters (feeding grounds) and a winter distribution lower latitudes (breeding/calving grounds), and these whales have become frequent visitors to Walvis Bay during the austral winter (June to August). Barendse *et al.* (2011) identified 35 individual humpback whales from photo-identification images taken in Walvis Bay, comparing these whales with catalogues of humpbacks from Angola, South Africa, Gabon and the Antarctic Humpback Whale Catalogue. No matches were found, however. Humpback whales off southern Africa were seriously depleted during the whaling era, but have since recovered well (Collins *et al.* 2008).

Southern right whales have also been documented in coastal waters (Roux *et al.* 2001; Leeney in prep) and are known to frequent Walvis Bay, particularly during the winter (June-September). The population was seriously depleted during the whaling era, but has recovered well and been increasing at 7% per year, with the African population estimated at ~4,600 animals in 2008 (Brandão *et al.* 2011). More frequent sightings of right whales off Namibia suggest that right whales are extending back into their old range, although most sightings within Namibia are still in the southern 400 km of the country (Roux *et al.* 2010). In recent years a number of the sheltered bays between Chameis Bay (27°56'S) and Conception Bay (23°55'S) have become popular calving sites for Southern Right whales (Roux *et al.* 2010).

Minke whales are also commonly sighted in Namibian waters, but mostly in the Lüderitz area. Pygmy right whales have stranded on numerous occasions in Walvis Bay, both as live animals and as carcasses (Leeney *et al.* in rev), with the high proportion of juvenile animals in strandings records suggesting that a breeding ground or nursery area for this little-known, and possibly rare species may be located off the Namibian coast. Similarly, Pygmy right whales strand regularly along the Namibian coast, particularly in Walvis Bay. As the majority of strandings are juvenile individuals, there may likewise be a nursery ground offshore of the Walvis Bay area (Leeney *et al.* (in rev)). Stranding or skeletal records of southern bottlenose whales, rough toothed dolphin and Gervais' beaked whale have been recorded from the Namibian coast, although the level to which these may be extra-limital records is unknown. There are no data on the population status of these species off the southern African coast.

Of the migratory cetaceans, the blue, sei and fin whales are listed as "Endangered" and the Southern Right and Humpback whales as "Least Concern" in the International Union for Conservation of Nature (IUCN) Red Data book. All whales and dolphins are given absolute protection under the Namibian Law.

The Cape fur seal (*Arctocephalus pusillus pusillus*) (Figure 20) is common along the Namibian coastline, occurring at numerous breeding sites on the mainland and on nearshore islands and reefs. Currently the largest breeding site in Namibia is at Cape Cross north of Walvis Bay where about 51,000 pups are born annually (MFMR unpubl. Data). The colony supports an estimated 157,000 adults (Hampton 2003), with unpublished data from Marine and Coastal Management (South Africa) suggesting a number of 187,000 (Mecenero *et al.* 2006). A further colony of ~9,600 individuals exists on Hollamsbird Island south of Sandwich Harbour. The colony at Pelican Point is primarily a haul-out site. The mainland seal colonies present a focal point of carnivore and scavenger activity in the area, as jackals and hyena are drawn to this important food source.

Seals are highly mobile animals with a general foraging area covering the continental shelf up to 120 nautical miles offshore (Shaughnessy 1979), with bulls ranging further out to sea than females. The timing of the annual breeding cycle is very regular occurring between November and January. Breeding success is highly dependent on the local abundance of food, territorial bulls and lactating females being most vulnerable to local fluctuations as they feed in the vicinity of the colonies prior to and after the pupping season (Oosthuizen 1991). Namibian populations declined precipitously during the warm events of 1993/94 (Wickens 1995), as a consequence of the impacts of these events on pelagic fish populations. Population estimates fluctuate widely between years in terms of pup production, particularly since the mid-1990s (MFMR unpubl. Data; Kirkman *et al.* 2007).



Figure 20: Colony of Cape fur seals *Arctocephalus pusillus pusillus* (Photo: Dirk Heinrich).

There is a controlled annual quota, determined by government policy, for the harvesting of Cape fur seals on the Namibian coastline. The Total Allowable Catch (TAC) currently stands at 60,000 pups and 5,000 bulls, distributed among four licence holders. The seals are exploited mainly for their pelts (pups), blubber and genitalia (bulls). The pups are clubbed and the adults shot. These harvesting practices have raised concern among environmental and animal welfare organisations (Molloy & Reinikainen 2003).

5.4 Other Uses of the Area

5.4.1 Mariculture Activities

Mariculture (marine aquaculture) has gained considerable interest in Namibia over the last few years and is being conducted at an increasing scale in Walvis Bay. The current National Development Plan (NDP2), which calls for the promotion of aquaculture activities, and the national policy paper Vision 2030 both foresee a thriving aquaculture industry. Since 2003, the Aquaculture Act has provided a legislative context, and the policy paper *Towards the Responsible Development of Aquaculture* (MFMR 2001) and the *Aquaculture Strategy* (MFMR 2004) were developed to address the development of a sustainable aquaculture sector. A Strategic Environmental Assessment developed for the Erongo Region, indicated that suitable locations for sea-based and land-based aquaculture were limited and would primarily be associated with Walvis Bay and Swakopmund (Skov *et al.* 2008). Two plots between Walvis Bay and Swakopmund have been specifically zoned for aquaculture developments that propose to produce shrimp, finfish and abalone. A further area to the north of Swakopmund (north of the Mile 4 Saltworks) was identified as a potential development area for land-based aquaculture (Figure 21).

In Walvis Bay, several companies are currently engaged in cultivation of Pacific oyster (*Crassostrea gigas*) and European flat oyster (*Ostrea edulis*) in the lee of Pelican Point, using suspended baskets on long lines in deeper areas and platforms in shallower depths. An Aqua Park for oyster farming has been proposed for the shallow areas in the lee of Pelican Point (Skov *et al.* 2008). The ~1,200 ha area, which is under the jurisdiction of Namport, is located within the boundaries of the (proposed) Walvis Bay Nature Reserve and has been zoned for aquaculture. The Aqua Park is a large development and may accommodate 10-20 oyster farms.

Oyster cultivation is also conducted in the feed-water ponds of the Walvis Bay and Swakopmund salt works (Figure 22). Oyster spat is imported from Chile and South Africa, and are grown out in trays and mesh bags suspended from ropes or wooden structures within the ponds (Maurihungirire & Griffin 1998; <http://www.keetmanshoop.com/oysters.htm>; <http://www.ncp.co.za/WalvisHomeProducts/Oysters.asp>). In 2007, the industry exported oysters worth about N\$35 million. In March 2008, however, the oyster industry suffered huge losses with an estimated 80 - 90% of the

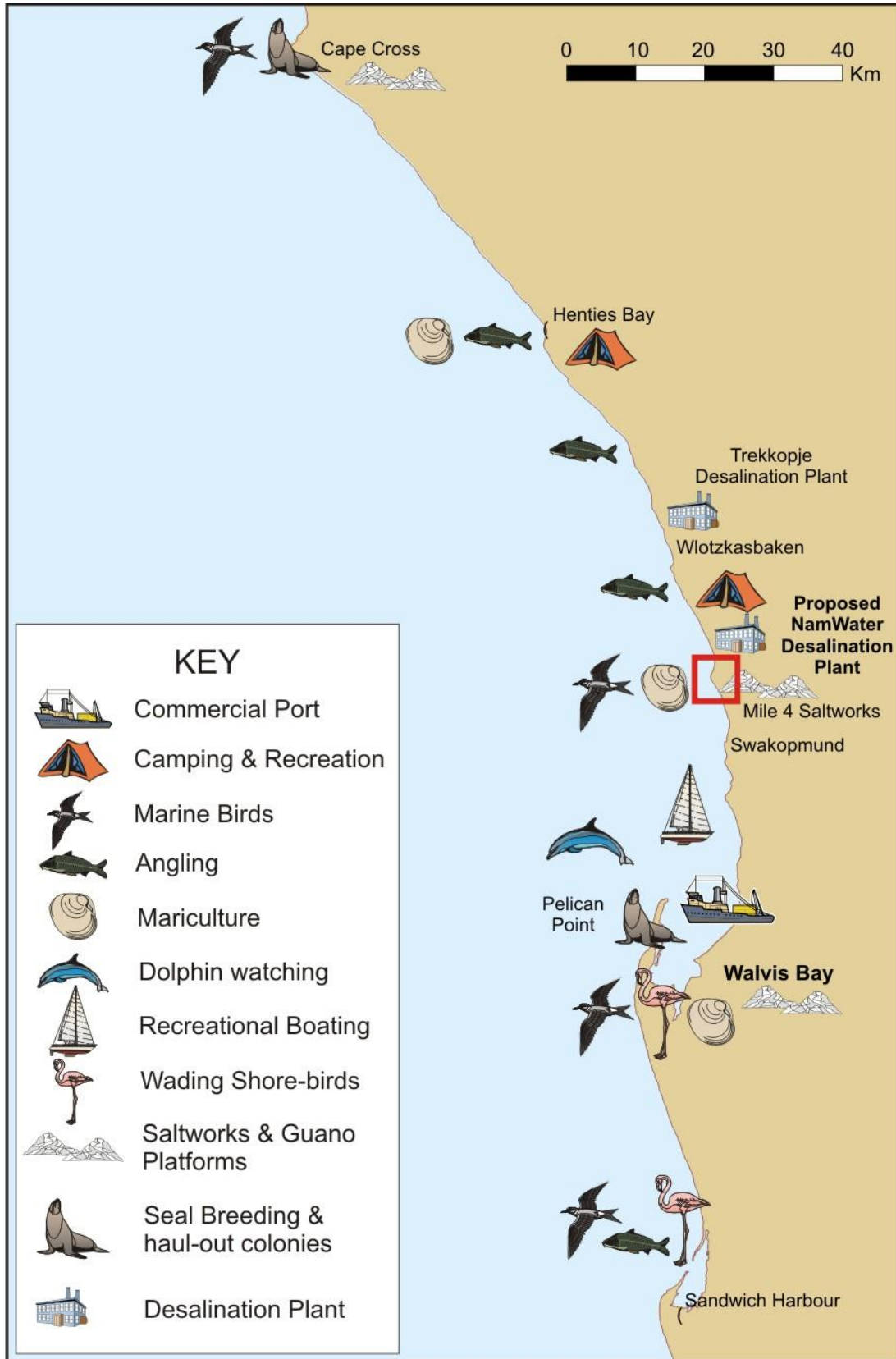


Figure 21: Location of the proposed desalination plant in relation to various project-marine environment interaction points.

stock being destroyed by a severe red tide event coupled with unusually high water temperatures (up to 25°C) and the release of hydrogen sulphide into the sea. The oyster farming industry is currently investigating ways to reduce the effects from any future algal bloom events. 'Toughening' oysters (e.g. taking the oysters out of the water so that they can adapt to surviving out of water or with little oxygen) and/or genetically modifying them to cope with harsh climatic and ocean conditions are some of the measures that may become important features in local oyster production (<http://allafrica.com> news from 28 April 2008, <http://africasciencenews.org> news from 28 March 2008). It has been suggested that long-term climatic trends in the Benguela ecosystem may lead to higher frequency of such low oxygen events (Monteiro *et al.* 2008). Mariculture in the region may thus become an increasingly risky prospect. The proposed on-land sites will face similar risks should suitable recirculation systems not be developed for these operations.

In 1999 experimental cultivation of the invasive alien mussel *Mytilus galloprovincialis* was undertaken at Pelican Point. The project was unsuccessful, however, due to the natural takeover by *Perna perna* and *Semimytilus algosus* (Molloy & Reinikainen 2003). The Sam Nujoma Marine and Coastal Resources Research Centre in Henties Bay is conducting research towards developing the fisheries and agriculture sectors, thereby complementing what is done by the MFMR. It includes the mariculture of algae (primarily *Gracilaria gracilis*) and White Mussel (*Donax serra*), as well as various finfish species (Blacktail, *Diplodus sargus capensis* and Silver Kob, *Argyrosomus inodorus*) (http://www.dlist-benguela.org/Hotspots/Hotspots/Erongo_Region). Some experiments are also being undertaken with South African abalone (*Haliotis midae*), and scallops.



Figure 22: Oyster cultivation in the salt works at Swakopmund (Photo: <http://www.ncp.co.za/WalvisHomeProductsOysters.asp>).

5.4.2 Fishing

Artisanal and recreational fishing

Artisanal subsistence fishing is not well developed in the region, being limited to the areas around Henties Bay, Swakopmund, and Walvis Bay. Shore angling is conducted by low-income residents to informally harvest fish for home consumption and for sale (Barnes & Alberts 2008). The formally recognized artisanal beach-seine fishery in Walvis Bay, targets mullet in the sheltered waters of the bay (Batty *et al.* 2005). By-catch species include barbel, blacktail, kob, steenbras and galjoen. Galjoen is commonly caught in the Swakopmund area (Barnes & Alberts 2008). Less than 150 individuals are involved in the artisanal fishery. Thus, although targetting the same resource, the artisanal sector is extremely small relative to the recreational angling sector, which has been

estimated at 8,800 recreational anglers making an annual landing of ~500 tons (Batty *et al.* 2005; Barnes & Alberts 2008).

The Namibian coast has a high reputation as a recreational angling destination, and was once legendary for the large catches made regularly by recreational anglers. Although only anecdotal evidence exists for the good catches made prior to the 1990s, the average catches have decreased considerably over the last two decades, (Holtzhausen & Kirchner 2001). Most angling is done from the shore, but some is also conducted from ski-boats beyond the surf zone. The recreational angling community is made up of three distinct segments: coastal Namibian residents (15%), inland Namibian residents (38%), and South African visitors (46%) (Kirchner *et al.* 2000). Recreational shore-angling in Namibia is primarily restricted to the coastline between Sandwich Harbour and the mouth of the Ugab River. Over 90% of shore-angling takes place in the vicinity of the coastal towns of Walvis Bay, Swakopmund and Henties Bay (Figure 23). Some limited angling also takes place farther north, at Terrace Bay and Torra Bay in the Skeleton Coast Park, and in the south towards Lüderitz, as well as near Oranjemund.



Figure 23: The area north of Walvis Bay is popular with rock- and surf-anglers (Photo: P. Tarr, from Molloy & Reinikainen 2003), resulting in heavy impact of the beaches by 4x4 traffic (Photo: Pisces Environmental Services).

Anglers target a variety of different species. The most important species is silver kob (*Argyrosomus inodorus*) constituting about 70% of all the recreational shore angling catches in Namibia (Kirchner *et al.* 2000). This species in particular has been heavily exploited in Namibian waters, and there is concern that the species is being depleted (Holtzhausen *et al.* 2001; Kirchner 2001). Other targeted

species include dusky kob (*A. coronus*), West Coast steenbras (*Lithognathus aureti*), galjoen (*Dichistius capensis*) and blacktail (*Diplodus sargus*). To a much lesser extent, sharks, including the bronze whaler shark (*Carcharhinus brachyurus*), the spotted gully shark (*Triakis megalopterus*) and the smoothhound (*Mustelus mustelus*), are targeted (Zeybrandt & Barnes 2001). Catches are made all year round, but are higher in summer. The bronze whaler, which is known in South Africa and Namibia as the copper shark or “bronzy”, is one of the focal points of a vibrant tourism industry (Holtzhausen & Camarada 2007). Shore anglers prize the bronze whaler for its legendary fighting ability and anglers from all over the world travel to Namibia in the hope of catching a bronze whaler from the beach, using rod and reel. An economic survey over the years 2003-2007 showed that 10 Namibian angling guides take out 3,600 clients (average 3 clients/day on a 6-day week for 200 days per year) specifically for bronze-whaler angling, annually generating at least US\$1 million to the economy of the country (excluding travelling costs).

In addition there is recreational harvest of shellfish in Namibia for personal consumption. Wild populations being harvested recreationally include the mussels *Perna perna*, *Mytilus galloprovincialis* and white mussel *Donax serra*.

Commercial Fisheries

The commercial linefishery, operates from Walvis Bay in inshore waters targeting similar species to those caught by the recreational shore anglers, namely silver kob and West Coast steenbras, and snoek (*Thyrsites atun*). Linefishing is conducted from skiboats as well as from larger vessels. Between April and August the boats primarily operate in the area between the Ugab River and Rocky Point, although they may occasionally also visit the area between Walvis Bay and Ugab River. During early summer, the fleet target the inshore areas off Walvis Bay for Snoek (Holtzhausen & Kirchner 1998).

The commercial linefishery catches kob in roughly equal numbers to those landed by shore anglers (Kirchner & Beyer 1999), but the kob caught by commercial linefish boats are on average older and larger than those caught by the recreational, leading to a total higher catch mass (Stage & Kirchner 2005). However, catches of kob have declined and commercial vessels are now increasingly catching sharks (Stage & Kirchner 2005), which were previously caught mostly by recreational skiboat anglers (Holtzhausen *et al.* 2001).

Apart from the requirement to hold a government permit, the commercial linefishery is not subject to any restrictions. Permits are freely available and the number of registered permit-holders has more than doubled in the past decade (Stage & Kirchner 2005). Following concern that fish stocks cannot support the current fishing pressure and that linefishing is becoming unprofitable, there have been discussions to introduce commercial linefishing restrictions, including reducing the number of permit-holders, introducing size limits, total allowable catches and/or closed seasons (Holtzhausen *et al.* 2001; Kirchner 2001).

Studies investigating the benefits of the linefishery to the Namibian economy showed that in total the linefishery contributed approximately N\$35 million to Namibia's Gross Domestic Product in 1996/1997 of which N\$29.7 million was direct expenditure by recreational shore-anglers (Kirchner *et al.* 2000). Using multiplier effects, it appears that the economic benefits are greatest in recreational angling, less in commercial fishing by large vessels, and least in commercial skiboat fishing (Zeybrandt & Barnes 2001; Barnes *et al.* 2004; Stage & Kirchner 2005) suggesting that further catch restrictions would do less harm to the economy if applied to the commercial linefishing sector rather than to recreational angling (Stage & Kirchner 2005).

The sardine *Sardinops sagax* and hake *Merluccius* spp. form the basis of the Namibian pelagic and demersal fishing industry, which operates out of Walvis Bay. The northern Benguela has experienced large fluctuations in fish stocks and sardine stocks in particular have decreased markedly from several million tons in the 1950s and 1960s to a few hundred thousand tons in recent years. Currently, the sardine industry relies heavily on the variable annual recruitment of sardine for its catches, making them susceptible to environmental impacts such as for example Benguela Niño events (Bartholomae & van der Plas 2007). The pelagic fishery is carried out by a fleet of entirely Namibian-owned steel and wooden-hulled purse-seining vessels, which operate out of Walvis Bay. The fishing extends from mid-February till the end of August, depending on when the

quotas are filled. The fleet consists of ~30 registered vessels ranging in length from 21 - 49 m, although in recent years less than half have been operational due to the severely depleted pelagic resource. Principally sardine is targeted for canning, whilst by-catch species are used for fish-meal and oil production. The fishery is entirely industrialised, with the smaller vessels concentrating on catches for fish meal, whilst the larger vessels concentrate on sardines for canning. The catches are brought back from as far afield as the Kunene River for processing at the reduction plants and canneries in Walvis Bay. Although primarily working further offshore, the purse-seiners may operate inshore to depths of 10 m (Hampton 2003). The Namibian demersal fishery is concentrated on the edge of the continental shelf, as a minimum trawl depth of 200 m has been set in order to protect the juvenile stocks. Interaction with the project is thus not expected.

5.4.3 Ecotourism

The old West Coast Recreation Area, now part of the newly proclaimed Dorob National Park, is renowned for its excellent angling, and is visited annually by thousands of fishermen. Popular beach angling spots along the coast have been identified and named to indicate their distance from Swakopmund (e.g. Mile 14, Mile 72 and Mile 108, at which campsites are located). Next to fishing from the shore, several skiboat operators from Swakopmund and Walvis Bay also offer guided angling tours. Specifically shark angling tours targeting bronze whalers, have become increasingly popular over the last decade and have become an established part of the local coastal tourist industry (Holtzhausen & Camarada 2007).

Swakopmund itself is described as the coastal playground of Namibia, and is increasingly attracting international tourism. Although its environment is its greatest economic asset (Skov *et al.* 2008), the area has now also become world-famous for adventure seekers who visit the area for quad-biking, sand boarding, tandem skydiving, camel and horse trails, paragliding, hot air ballooning etc.

In recent years Walvis Bay has also begun successfully marketing its natural marine and desert attractions - the Bay itself and the Lagoon, the Kuiseb Delta and the Namib Desert, and the Dunebelt north of it. Specifically marine ecotourism has become increasingly important, with ten whale-watching operators currently offering general nature trips that include sightings of dolphins and whales, as well as other marine life (e.g. fur seals, turtles and sunfish) out of both Walvis Bay and Swakopmund. Over the last years, whale watching tourist numbers have increased dramatically at an average annual growth rate of 20% (O'Connor *et al.* 2009).

Various operators in Walvis Bay also offer 4x4 excursions to the Sandwich Harbour area, which include the Walvis Bay Lagoon, the Saltpans, the Kuiseb River Delta, and - if weather and tides allow for it - the Sandwich Harbour Lagoon (Figure 24, left).



Figure 24: Ecotourism in the study area includes 4x4 excursions to Sandwich Harbour (left) and kayak tours to Pelican Point (right) (Photos: www.sandwich-harbour.com).

Other recreational activities in the study area are leisure boating, kayaking, windsurfing and various beach activities (Figure 24, right). In addition, the Walvis Bay and Swakop lagoons provide ample opportunity for excellent bird watching.

5.4.4 Industrial Uses

Desalination Plants

A Reverse Osmosis (RO) desalination plant 30 km north of Wlotzkasbaken provides water for the Areva Resources Southern Africa mine, a Uranium mine 65 km north-east of Swakopmund. The annual net production of treated water from this desalination plant is 20 million m³/annum with an associated sea water abstraction rate of ~48 million m³/annum and a brine discharge volume of ~31 million m³/annum.

A further RO desalination plant is planned by NamWater at Mile 6 just north of Swakopmund, also to provide water for the fast growing Uranium Mining Industry. The plant would have an output capacity of 25 million m³/annum. Associated with the plant would be a seawater intake structure designed for a maximum of 110 million m³/annum, with an installed abstraction capacity of 63 million m³/annum and a brine disposal system through which 38 million m³/annum of brine could be discharged back into the sea. The outcome, timelines and commercial aspects to the NamWater project however remain uncertain. These uncertainties have been a significant motivator for the proposed Rössing Uranium desalination plant.

The proposed Rössing Uranium desalination plant, although of a much smaller capacity, would therefore potentially constitute the third desalination plant in the area.

Saltworks

Namibia is the largest salt producer in sub-saharan Africa. Salt is the most important non-metallic minerals mined in Namibia, with the bulk of the salt output coming from the seawater evaporation pans at Walvis Bay and Swakopmund. The Swakopmund-based Salt Company (Pty) Ltd produces around 120,000 tons of edible salt annually (<http://www.saltco.com/index.html>). The saltworks are situated about 7 km (4 miles) north of Swakopmund. Production of the concentrated brine at the saltpan, known as Panther Beacon, began in 1933, but by 1952 the salt source was exhausted. Seawater has since been pumped into open evaporation and concentration ponds from which crystallized salt is removed with mechanical scrapers. The pans are shallow and of varying salinity. Apart from a few halophytes, the saltworks are devoid of vegetation.

Guano

Guano is rich in Nitrogen (14-16%), Phosphorus (9%) and Potassium (3%), and consequently is valuable as an agricultural fertilizer. The man-made guano platforms at Walvis Bay (**Error! Reference source not found.**, left), Swakopmund and Cape Cross are unique in the world, and guano has been scraped annually from them since the 1930s. The wooden platform between Swakopmund and Walvis Bay is located ~200 m offshore and has an area of 1.7 ha. A further two platforms of 4 ha each have been erected at the salt pans north of Swakopmund (see above) and at Cape Cross. The platforms currently produce about 2,500 tons of guano per season. Due to the absence of sand on the platforms, the guano is of a very high quality, fetching about N\$ 700 per ton. It is reaped every 12-18 month after the end of the main summer breeding season (<http://www.traveltonamibia.com/guano.htm>).

5.4.5 Existing Environmental Impacts

Traffic on Beaches

The West Coast Recreation Area between Swakopmund and Henties Bay is used extensively by recreational rock- and surf anglers, who visit the coast during the summer holiday period (December/January). Between January and April the majority of visitors are from South Africa and

visit the area only to fish. These anglers travel both on the coastal salt road as well as on the beach. Rock Bay (Mile 17), about 4.5 km south of Wlotzkasbaken is also popular with fishermen for launching small inflatable boats and skiboats to target fishing sites beyond the surf zone.

Various studies have been undertaken that show the effects of human-induced disturbances in the form of vehicle traffic on physical characteristics of the beaches, and on the faunal populations inhabiting them (Defeo & Alava 1995; Alonso *et al.* 2002; Borges *et al.* 2002; Gomez-Pina *et al.* 2002). Shearing and compression as a result of vehicle passage in soft sand can extend to depths of 20 cm. As most of the macrofaunal species occur in the top 10-cm layer of sand, compaction of these sediments due to vehicular traffic on beaches can result in significant reduction in macrofaunal abundance. While sensitivity to compaction varies between species, results have indicated that at upper levels of the beach, as few as 17 passes can damage up to 10% of the supralittoral fauna, with some species suffering mortalities of up to 98% (van der Merwe & van der Merwe 1991). The tendency for drivers to follow the same tracks accentuates this. Tyre ruts are also capable of trapping certain species (e.g. giant isopods, ghost crabs, and the chicks of certain shorebirds that nest on the upper beach), which are then subsequently run over by following traffic (van der Merwe & van der Merwe 1991; Brown 2000; amongst others). Although birds can get used to vehicles passing very close by, disturbed birds can be flushed from their nests resulting in decreased breeding success. Beach traffic can also be a constant disturbance to resting flocks at high tide, or to flocks feeding in the swash zone at low tide (van der Merwe & van der Merwe 1991).

Rock- and Surf-Angling

The central Namibian coast between Swakopmund and Henties Bay was once legendary for the large catches regularly made by recreational anglers visiting the area specifically to fish. Over the past two decades, however, average catches have decreased considerably (Holtzhausen & Kirchner 2001), and species specifically targeted by anglers (e.g. silver kob) are now considered to be over-exploited (Holtzhausen *et al.* 2001; Kirchner 2001).

Areva Reverse Osmosis Desalination Plant near Wlotzkasbaken

During the operational life time of the plant, a high-salinity brine is being more or less continuously discharged at a volume of ~85 000 m³/day. This might increase with an expansion of the plant. The reject brine is discharged back into the sea at 6 m water depth through a 600-m long single discharge pipeline with 27 diffusers, 150 mm in diameter and spaced 3 m apart. Chlorine is used to reduce biofouling of the pipelines, which is also pigged periodically to clear any macro-fouling of the lines. The chlorine is neutralized with sodium metabisulphite (SMBS). The maximum expected volume of cleaning chemicals is 53 m³, to be discharged into a volume of 8 x 3,773 m³ providing a 190 times dilution.

5.4.7 Potentially Threatened Habitats

Taking into account the waste-water characteristics of the proposed discharge from the desalination plant, potential impacts are most likely to target important marine ecosystems and beneficial uses that rely on the health of marine organisms and plants, such as recreational angling and marine aquaculture activities. Certain areas of special interest potentially to be impacted by discharges from the desalination plant into the marine environment were identified. These specific areas include:

- The natural intertidal and shallow subtidal environments adjacent to the discharge site;
- Recreational rock- and surf-angling; and
- Salt-producing and current and/or future mariculture activities.

6. IDENTIFICATION OF KEY ISSUES AND SOURCES OF POTENTIAL ENVIRONMENTAL IMPACT

During the course of the environmental scoping process for the proposed Rössing Uranium desalination plant, key issues were identified relating to potential impacts on the marine environment. These are identified below in terms of the construction, commissioning, operation and decommissioning phase.

6.1 Identification of Key Issues

6.1.1 Construction Phase

The potential impacts associated with the construction of feedwater intake and brine discharge structures into the marine environment are related to:

- Onshore construction (human activity, air, noise and vibration pollution, dust, blasting and piling driving, disturbance of coastal flora and fauna) (to be dealt with by others); and
- Construction and installation of pipeline intakes and discharge (construction site, pipe lay-down areas, and trenching in the marine environment, vehicular traffic on the beach and consequent disturbance of intertidal and subtidal biota).

The desalination plant will be constructed a set-back distance from the existing shoreline. Consequently, issues associated with the location of the plant, and the associated pipelines leading to and from these constructions are not deemed to be of relevance to the marine environment, and will be dealt with by other specialist studies. However, infrastructure extending into the sea will potentially impact on intertidal and shallow subtidal biota during the construction phase in the following ways:

- Temporary loss of benthic habitat and associated communities due to preparation of seabed for buried pipeline laying and associated activities (e.g. jetties);
- Temporary loss of supratidal habitat as a result of vehicular traffic and earth moving equipment on the shore, and associated spoils dumping, backfilling and stockpiling activities;
- Possible temporary short-term impacts on habitat health due to turbidity generated during construction;
- Temporary disturbance of marine biota, particularly marine mammals and turtles, due to construction activities (blasting and piling driving, breakwater construction);
- Interruption of longshore sediment movement by sheet piling and jetty structure resulting in increased erosion and/or accretion around the construction site (refer to the Coastal Dynamics Specialist Study);
- Possible impacts to marine water quality and sediments through hydrocarbon pollution by marine construction infrastructure and plant; and
- Potential contamination of marine waters and sediments by inappropriate disposal of spoil and/or surplus rock from construction activities or backfilling, used lubricating oils from marine machinery maintenance and human wastes, which could in turn lead to impacts upon marine flora, fauna and habitat.

6.1.2 Commissioning Phase

Once construction has been completed, it will take about 3 months to commission the new desalination plant. During the commissioning phase, seawater will be pumped into the plant at up to peak production rates. However, any fresh water produced will be combined with the brine and discharged. As the discharge will have a salinity equivalent to that of normal seawater, it will not have an environmental impact during the commissioning phase.

It may be necessary to discard the membrane storage solution and rinse the membranes before plant start-up. If the storage solution contains a biocide or other chemicals which may be harmful to marine life and this solution is discharged to the sea, local biota and water quality may be affected.

6.1.3 Operational Phase

The key issues and major potential impacts are mostly associated with the operational phase. The key issues related to the presence of pipeline infrastructure and brine discharges into the marine environment are:

- Altered flows at the intake and discharge resulting in ecological impacts (e.g. entrainment and impingement of biota at the intake, flow distortion/changes at the discharge, and affects on natural sediment dynamics);
- Potential for habitat health impacts/losses resulting from elevated salinity in the vicinity of the brine discharge;
- The effect of the discharged effluent potentially having a higher temperature than the receiving environment;
- Biocidal action of residual chlorine in the effluent;
- The effects of co-discharged constituents in the waste-water;
- The abstraction of large volumes of feedwater resulting in the removal of particulate matter from the water column where it is a significant food source, as well as changes in phytoplankton production due to changes in nutrients, reduction in light, water column structure and mixing processes; and
- Direct changes in dissolved oxygen content due to the difference between the ambient dissolved oxygen concentrations and those in the discharged effluent, and indirect changes in dissolved oxygen content of the water column and sediments due to changes in phytoplankton production as a result of nutrient input.

Additional engineering design considerations, not strictly constituting issues to be considered within this marine specialist study, include the following:

- Structural integrity of the intake and outfall pipelines (e.g. related to shoreline movement);
- Potential changes in shoreline dynamics due to the presence of intake structures and discharge pipelines (see WSP 2014a).
- Potential re-circulation of brine effluent;
- Pipeline maintenance and replacement requirements;
- Suitable disposal of solid waste, i.e. filter backwash, and sludge from pre-treatment processes; and
- Water quality of feed-waters that should include consideration of possible deteriorating water quality (particularly algal blooms, sediments that may be stirred up during storms, or large-scale hypoxia or sulphur eruptions in bottom waters), that may require specific mitigation measures or planned flexibility in the operations of the desalination plant.

6.1.4 Decommissioning Phase

The minimum anticipated life of the desalination plant is approximately 10 years. The individual RO modules will be replaced as and when required during this period. No decommissioning procedures or restoration plans have been compiled at this stage. Being a modular plant, decommissioning should not involve extensive demolition of the plant area. In the case of decommissioning the pipeline will most likely be left in place. The potential impacts during the decommissioning phase are thus expected to be minimal in comparison to those occurring during the construction and operational phase, and no key issues related to the marine environment are identified at this stage, since cessation in the operation of the plant will result in an immediate discontinuation of the majority of the identified marine impacts.

7. ASSESSMENT OF ENVIRONMENTAL IMPACTS

The sources of potential impacts and key issues identified in terms of the construction, commissioning, and operational phases of the proposed Rössing Uranium desalination plant relating to potential risks to the marine environment are discussed in detail and assessed below.

7.1 Construction of Intake and Discharge Structures

7.1.1 Disturbance of the Coastal Zone

The use of intake structures and discharge pipelines in the engineering designs for the desalination plant is unavoidable, but will involve considerable disturbance of the high-shore, intertidal and shallow subtidal habitats during the construction and installation process. The intake and outfall points of the pipelines will be located below the low water mark, in the surf zone.

The seawater intake will comprise a jetty situated within the intertidal zone with vertical turbine pumps to abstract the seawater and convey it to the jetty's elevated land side section from where it will discharge into gravity flow pipelines and flow towards the head of an existing overland channel. The channel will lead to a new inland seawater pond system located near the desalination plant. The fairly small flow requirement, together with the use of a pond system for redundancy, storage and improvement of water quality, allows for a shallow intake within the intertidal zone thereby avoiding the need for excessive trenching and blasting. The present intake for the Swakopmund Salt works operates under the same principles and has been proven to be feasible and efficient. The jetty itself would consist of two lines of steel piles, with a steel framework supported from the piles. Construction of the jetty may thus require pile driving and the construction of concrete foundations in the shallow subtidal and intertidal zones.

The brine outfall will consist of a pumpstation, and a 400 mm diameter HDPE pipeline 2.25 km in length. The pipeline will be trenched on land and across the beach and concrete encased from the high water mark to the discharge location in the surf zone to provide stability on the seabed and adequately protect it where it crosses the surf zone. The outfall will terminate in a brine diffuser.

Individual pipeline sections will be fabricated by the supplier and transported to site. This will require a sufficiently large and relatively flat onshore area (immediately inland of the final pipeline position) where the pipes can be stockpiled and prepared. Coastal vegetation and associated fauna at the jetty and pipeline construction sites will almost certainly be severely disturbed or removed. The pipe sections will subsequently be butt-welded together into long strings, and placed either on the jetty or in the excavated trench. Once trenched, the discharge pipeline will be covered with concrete and rock. Obviously, the physical removal of sediments or bedrock in the discharge pipeline trench, and disposal thereof into the surf zone will result in the total destruction of the associated benthic biota within the immediate area. Mobile organisms such as fish, shore birds and marine mammals, on the other hand, are capable of avoiding the construction area and although severely disturbed for the duration of construction activities, should not be significantly affected by the excavations.

Despite this unavoidable disturbance of the intertidal and shallow subtidal habitats, the activities would remain localised and impacts would generally become less intrusive with increasing distance from the construction disturbance and should not extend beyond a hundred metres of the construction site. Provided the construction activities are all conducted concurrently, the duration of the disturbance should also only be limited to a period of not more than 18 months. Active rehabilitation of intertidal communities is not possible, but rapid natural recovery of disturbed habitats in the turbulent intertidal and surf zone areas can be expected. Furthermore, the exposed pipeline, concrete foundations and concrete casing will serve as a new 'hard-bottom' substrate for colonisation by marine benthic communities. The ecological recovery of marine habitats is generally defined as the establishment of a successional community of species, which progresses towards a community that is similar in species composition, population density and biomass to that previously present (Ellis 1996). In general, communities of short-lived species and/or species with a high reproduction rate (opportunists) may recover more rapidly than communities of slow growing, long-lived species. Opportunists are usually small, mobile, highly reproductive and fast growing

species and are the early colonisers. Habitats in the nearshore wave-base regime, which are subjected to frequent disturbances, are typically inhabited by these opportunistic species (Newell *et al.* 1998). Recolonisation will start rapidly after cessation of trenching, and species numbers may recover within short periods (weeks) whereas biomass often remains reduced for several years (Kenny & Rees 1994, 1996).

Studies on the disturbance of beach macrofauna and rocky shore communities on the southern African West Coast by beach mining activities and shore-based diamond diving operations have ascertained that, provided physical changes to beach morphology and rocky intertidal zones are kept to a minimum, biological 'recovery' of disturbed areas will occur within 2-5 years (Nel *et al.* 2003; Pulfrich *et al.* 2003; Pulfrich *et al.* 2004). Disturbed subtidal communities within the wave base (<40 m water depth) might recover even faster (Newell *et al.* 1998; Pulfrich & Penney 2001).

Disturbance of the intertidal and subtidal rocky shore and/or beach during installation of the intake and discharge pipelines is consequently deemed of high magnitude within the immediate vicinity of the construction sites, with impacts persisting over the short- to medium-term and is considered to be of **MEDIUM** significance without mitigation. This rating is applicable to the construction of both the intake and the discharge pipelines, regardless of location. With the implementation of mitigation measures, the duration of the impacts may reduce to short-term thus reducing the significance to **LOW**.

Disturbance and destruction of marine biota through alteration and disruption of the coastal zone during construction	
CRITERIA	
Extent	Site Specific; the construction impacts will occur within 500 m of the source
Magnitude	High; biota in the construction footprint will be severely disturbed or completely eliminated
Duration	Short- to Medium-term; disturbed marine communities likely to recover within 2-5 years
Significance	MEDIUM
Probability	Definite
Confidence	Certain
Reversibility	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> • Restrict disturbance of the intertidal and subtidal areas to the smallest area possible. • Lay pipeline in such a way that required rock blasting is kept to a minimum. • Restrict traffic on upper shore to minimum required. • Restrict traffic to clearly demarcated access routes and construction areas only. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> • Have good house-keeping practices in place during construction.
Significance with Mitigation	LOW

7.1.2 Pollution and Accidental Spills

Construction activities in the intertidal and shallow subtidal zones will involve extensive traffic on the shore by heavy vehicles and machinery, as well as the potential for accidental spillage or leakage of fuel, chemicals or lubricants. Any release of liquid hydrocarbons has the potential for direct, indirect and cumulative effects on the marine environment through contamination of the water and/or sediments. These effects include physical oiling and toxicity impacts to marine fauna and flora, localised mortality of plankton, pelagic eggs and fish larvae, and habitat loss or contamination (CSIR 1998; Perry 2005). Many of the compounds in petroleum products have been known to smother organisms, lower fertility and cause disease in aquatic organisms. Hydrocarbons

are incorporated into sediments through attachment to fine dust particles, sinking and deposition in low turbulence areas. Due to differential uptake and elimination rates filter-feeders particularly mussels can bioaccumulate organic (hydrocarbons) contaminants (Birkeland *et al.* 1976). Concrete work will be required in the intertidal and shallow subtidal zones during construction and installation of the pipelines. As cement is highly alkaline, wet cement is strongly caustic, with the setting process being exothermic. Excessive spillage of cement in the intertidal area may thus potentially increase the alkalinity of the water column with potential sublethal or lethal effects on marine organisms.

During construction (and also during operation), litter can enter the marine environment. Inputs can be either direct by discarding garbage into the sea, or indirectly from the land when litter is blown into the water by wind. Marine litter is a cosmopolitan problem, with significant implications for the environment and human activity all over the world. Marine litter travels over long distances with ocean currents and winds. It originates from many sources and has a wide spectrum of environmental, economic, safety, health and cultural impacts. It is not only unsightly, but can cause serious harm to marine organisms, such as turtles, birds, fish and marine mammals. Considering the very slow rate of decomposition of most marine litter, a continuous input of large quantities will result in a gradual increase in litter in coastal and marine environment. Suitable waste management practices should thus be in place to ensure that littering is avoided.

Potential hydrocarbon spills and pollution in the intertidal zone during installation of the intake and discharge pipelines is thus deemed of medium intensity within the immediate vicinity of the construction sites, with impacts persisting over the short- to medium-term. The impact is therefore assessed to be of **MEDIUM** significance without mitigation. This rating is applicable to the construction of both the intake and the discharge pipelines, regardless of location. With the implementation of mitigation measures, impacts would become **LOW**.

Detrimental effects on marine biota through accidental hydrocarbon spills, concrete works and litter in the coastal zone during construction	
CRITERIA	
Extent	Site Specific; the construction impacts will occur within 500 m of the source
Magnitude	Medium; physiological functioning of biota may be reduced or they may die
Duration	Short- to Medium-term; affected communities likely to recover within 2-5 years
Significance	MEDIUM
Probability	Probable
Confidence	Certain
Reversibility	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> • Conduct a comprehensive environmental awareness programme amongst contracted construction personnel. • Only equipment and vehicles actively involved in construction should be permitted on the beach and associated works areas. When not in use, and overnight, all equipment and plant must be withdrawn to higher ground; • Refuelling of equipment from a bowser should take place on higher ground away from the beach and wet works areas; • For equipment maintained in the field, oils and lubricants to be contained and correctly disposed of off-site. • Maintain vehicles and equipment to ensure that no oils, diesel, fuel or hydraulic fluids are spilled. • Vehicles should have a spill kit (peatsorb/ drip trays) onboard in the event of a spill. • No mixing of concrete in the intertidal zone and care taken to dispose of concrete washwater in a responsible manner that will not leach back to the ocean. • Regularly clean up concrete spilled during construction. • No dumping of excess concrete or mortar on the sea bed. • Ensure regular collection and removal of refuse and litter from intertidal areas. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> • Have good house-keeping practices in place during construction.
Significance with Mitigation	LOW

7.1.3 Increased Turbidity

Excavations, disturbance and turnover of sediments and boulders in the intertidal and/or surf zone will result in increased suspended sediments in the water column and physical smothering of biota by the discarded sediments. The effects of elevated levels of particulate inorganic matter and deposition thereof have been well studied, and are known to have marked, but relatively predictable effects in determining the composition and ecology of intertidal and shallow subtidal benthic communities (e.g. Engledow & Bolton 1994, Iglesias *et al.* 1996, Slattery & Bockus 1997). Increased suspended sediments in the surf zone and nearshore can potentially affect light penetration and thus phytoplankton productivity and algal growth, load the water with inorganic suspended particles, which may affect the feeding and absorption efficiency of filter-feeders, and can cause scouring of biota (e.g. shells, kelp stipes).

Rapid deposition of material from the water column will have a smothering effect. Some mobile benthic animals inhabiting soft-sediments are capable of migrating vertically through more than 30 cm of deposited sediment (Newell *et al.* 1998). Sand inundation of reef habitats was found to directly affect species diversity whereby community structure and species richness appears to be controlled by the frequency, nature and scale of disturbance of the system by sedimentation (Seapy & Littler 1982, Littler *et al.* 1983, Schiel & Foster 1986, McQuaid & Dower 1990, Santos 1993, Airolidi

& Cinelli 1997 amongst others). For example, frequent sand inundation may lead to the removal of grazers thereby resulting in the proliferation of algae (Hawkins & Hartnoll 1983; Littler *et al.* 1983; Marshall & McQuaid 1989; Pulfrich *et al.* 2003a, 2003b).

Construction activities required for the installation of the intake and discharge pipelines for the Rössing Uranium desalination plant will be highly localised. The impact of the resulting sediment plumes is likewise expected to be localised and of short duration (only for a couple of hours to a few days after cessation of excavation activities). As the biota of sandy and rocky intertidal and subtidal habitats in the wave-dominated nearshore areas of southern Africa are well adapted to high suspended sediment concentrations, periodic sand deposition and resuspension, impacts are expected to occur at a sublethal level only.

Elevated suspended sediment concentrations in nearshore waters due to construction activities is thus deemed of low magnitude within the immediate vicinity of the construction sites, with impacts persisting over the short-term only. The impact is therefore assessed to be of **VERY LOW** significance both without and with mitigation. This rating is applicable to the construction of both the intake and the discharge pipelines, regardless of location. As elevated suspended sediment concentrations are an unavoidable consequence of construction activities in the intertidal zone, no direct mitigation measures, other than the “no go” alternative, are possible. Impacts can however be kept to a minimum through responsible construction practices.

Reduced physiological functioning of marine organisms due to increased turbidity of nearshore waters during excavations	
CRITERIA	
Extent	Site Specific; turbidity impacts will occur within 500 m of the source
Magnitude	Very Low; biota are well adapted to high suspended sediment concentrations
Duration	Short-term; suspended sediment plumes will persist for hours or days only
Significance	VERY LOW
Probability	Definite
Confidence	Certain
Reversibility	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> No dumping of construction materials in the intertidal and subtidal zones. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Have good house-keeping practices in place during construction.
Significance with Mitigation	VERY LOW

Smothering of benthos through redeposition of suspended sediments	
CRITERIA	
Extent	Local; sediments may be carried some distance from the construction site by surf zone currents before being deposited
Magnitude	Very Low; biota are well adapted to sediment deposition and re-suspension
Duration	Short-term; deposited sediments are likely to be constantly resuspended in the turbulent surf zone environment
Significance	VERY LOW
Probability	Definite
Confidence	Certain
Reversibility	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> No dumping of construction materials in the intertidal and subtidal zones. Implement design measures to reduce the loss of materials from temporary earthen berms, construction platforms or access roads during the construction period. i.e. reduce the construction programme of wet works where possible, use geotextiles to reduce scouring of earthen berms, etc. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Have good house-keeping practices in place during construction.
Significance with Mitigation	VERY LOW

7.1.4 Construction Noise and Blasting

During jetty installation and pipeline trenching operations, noise and vibrations from excavation machinery and pile drivers may have an impact on surf zone biota, marine mammals and shore birds in the area. Noise levels during construction are generally at a frequency much lower than that used by marine mammals for communication (Findlay 1996), and these are therefore unlikely to be significantly affected. Additionally, the maximum radius over which the noise may influence is very small compared to the population distribution ranges of surf zone fish species, resident cetacean species and the Cape fur seal. Both fish and marine mammals are highly mobile and should move out of the noise-affected area (Findlay 1996). Similarly, shorebirds and terrestrial biota are typically highly mobile and would be able to move out of the noise-affected area.

Trenching of the discharge pipeline may require blasting to attain the required depths. As details of the probable blast levels, blasting practice and duration of the blasting required to ensure suitable burial of the pipeline have not yet been determined, the assessment that follows is generic only. Effects of underwater blasting and pile driving on marine organisms have received extensive coverage in the formal peer-reviewed scientific literature (see Lewis 1996 and Keevin & Hemen 1997 for references), as well as in various assessments for seismic surveys, underwater construction and weapons testing. The following impact description is based on two reviews on the subject provided in Lewis (1996) and Keevin & Hemen (1997).

Explosives generate chemical energy, which is released as physical, thermal, and gaseous products. The most important of these for marine organisms is the physical component which, as a shock wave, passes into the surrounding medium. Depending on the blasting practice, some of the energy may escape into the water column, and it is this shock wave that is the primary cause of damage to aquatic life at, or some distance from the shot point. Thermal energy dissipation, in contrast, is generally limited to the immediate vicinity (<10 m) of the exploding material, and in shallow water gaseous products produce minor shock wave amplitudes.

The nature of the shock wave generated by the blast depends on the type of explosive used. Relatively low energy explosives such as black powder are slow burning and produce a shock wave with a shallow rise height. Dynamite and other high explosives have a rapid detonation velocity and produce a more abrupt shock wave. Consequently, high explosives have more dramatic effects on marine organisms.

Two damage zones are associated with an underwater explosion:

- an immediate kill zone of relatively limited extent, but within which all animals are susceptible to damage through disruption of their body tissues by the pressure wave generated by the explosion, and
- a more extensive remote damage zone in which damage is caused by negative pressure pulses, generated when the compression wave is reflected from an air-water interface. The negative pulses act on gas bodies within the organism inducing injuries such as haemorrhaging and contusions of the gastro-intestinal tract (mammals and birds) or rupture of swimbladders in fish.

Keevin & Hempen (1997) and Lewis (1996) provide information on blast-effects on a variety of shallow water (<10 m) organisms. Appendix A.1 provides a summary of these effects focussing on the marine macrophytic algae, major invertebrate macrofaunal taxa, fish, turtles and marine mammals that may occur in the blast area off the desalination plant site.

From this summary, the following can be gleaned:

- Any effects on macrophytes through blasting would be limited to the immediate vicinity of the charges.
- Marine invertebrates appear to be relatively immune to blast effects in terms of obvious injury or mortalities, suggesting that any blast-effects are likely to remain confined to the immediate area of blasting.
- In fish, the swim bladder is the organ most frequently damaged through blasting, potentially leading to high mortality in the immediate area of blasting. In contrast, fish species that do not possess swim bladders seem to be largely immune to underwater explosions. Egg and fish larvae may also be affected by underwater explosions, but impact ranges seem to be restricted to the immediate vicinity of the blasting. Although injury or mortality of fish and/or their eggs and larvae in the immediate area of the blasting is likely to occur, the probability of the blasting programme having a measurable effect at the population level on fish in the study area is judged to be unlikely, as surf zone and nearshore species along the central Namibian coastline are widely distributed.
- The limited information available on blasting effects on swimming and diving birds suggests that mortality occurs primarily within the immediate vicinity (< 10 m) of the blast.
- Effects on sea turtles may occur up to a distance of 1 km from the underwater explosion. Although occurring in the study area, turtles are infrequent visitors in the shallow nearshore regions.
- Similar to fish, injuries to marine mammals generated by underwater explosions are primarily trauma of various levels to organs containing gas, and mortality can occur in the immediate area around the blasting. Given the generally low numbers of seals in the study area relative to the overall population size any population level mortality effects, or injuries that may be caused are judged to be insignificant. Seals and scavenging birds may, however, be attracted to the blasting area by stunned and dead fish following a blast. Although occurring in the study area, whales and dolphins are infrequent visitors in the shallow nearshore regions, being more common further offshore. However, Heaviside's Dolphin and the Common Bottlenose Dolphin occur in shallow waters (<50 m) and could be vulnerable to detonations.

It is recommended that the area around the blasting site be visually searched before blasting commences and blasting postponed should a marine mammal, sea turtle or flocks of swimming and diving birds be spotted within a 2-km radius around the blasting point. The blasting programme should also be scheduled to allow seals and other scavengers feeding on dead fish to have left the area before the next blasting event. The probability of the proposed blasting programme having a

measurable effect on sea turtles or marine mammals in the study area is unlikely if these recommendations are strictly adhered to.

Disturbance and injury to marine biota due to construction noise is thus deemed of low magnitude within the immediate vicinity of the construction sites, with impacts persisting over the short-term only. In the case of blasting, however, the impact would be of high magnitude, but also persist over the short-term only. Without mitigation, the impacts of construction noise and blasting are therefore assessed to be of **LOW** and **MEDIUM** significance, respectively. This rating is applicable to the construction of both intake and the discharge pipelines, regardless of location. The implementation of mitigation would reduce the magnitude of the impact of construction noise to low and thus the overall significance to **VERY LOW**. In the case of blasting, the magnitude would reduce to medium and the extent to site specific with the implementation of mitigation measures, thereby reducing the overall significance to **LOW**. As the noise associated with construction is unavoidable, no direct mitigation measures, other than the “no go” alternative, are possible. Impacts can however be kept to a minimum through responsible construction practices and an accelerated wet works construction program.

As details of the probable blast levels, blasting practice and duration of the blasting required have not yet been finalised, confidence in the assessment of this impact is rated as medium.

Disturbance of shore birds and marine biota through construction noise	
CRITERIA	
Extent	Site Specific; noise will be restricted to the immediate vicinity of the construction site
Magnitude	Medium; some biota will be disturbed
Duration	Short-term; for the duration of construction activities only
Significance	LOW
Probability	Probable
Confidence	Sure
Reversibility	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Restrict construction noise and vibration-generating activities to the absolute minimum required. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Have good house-keeping practices in place during construction.
Significance with Mitigation	VERY LOW

Disturbance of and injury to shore birds and marine biota through blasting	
CRITERIA	
Extent	Local; blasting noise could potentially extend beyond the immediate vicinity of the construction site
Magnitude	High ; some biota will be disturbed and may be injured
Duration	Short-term; for the duration of construction activities only
Significance	MEDIUM
Probability	Probable
Confidence	Sure
Reversibility	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> • Restrict blasting to the absolute minimum required (one blast per day). • Use blasting methods which minimise the environmental effects of shock waves through the use of smaller, quick succession blasts directed into the rock. • Avoid onshore blasting during the breeding season of shore-birds. • Undertake visual observation prior to blasting to ensure there are no marine mammals and turtles present in the immediate vicinity (approximately 2-km radius). <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> • Development of a responsible blasting schedule.
Significance with Mitigation	LOW

7.1.5 Installation of Structures

Installation of the jetty and discharge pipeline will effectively eliminate any (sandy or rocky) biota in the structural footprint, and reduce the area of seabed available for colonisation by marine benthic communities. Although the loss of substratum as a result of the jetty and discharge pipeline constitutes a negative impact, it will be temporary only, as the structures themselves will provide an alternative substratum for colonising communities. Assuming that the hydrographical conditions around the structures will not be significantly different to those on the seabed, a similar community to the one previously present can be expected to develop, thereby constituting a positive impact.

The composition of the fouling community on artificial structures depends on the age (length of time immersed in water) and the composition of the substratum, and usually differs from the communities of nearby natural rocky reefs (Connell & Glasby 1999; Connell 2001). Colonisation of hard substratum goes through successional stages (Connell & Slayter 1977). Early successional communities are characterised by opportunistic algae (e.g. *Ulva* sp., *Enteromorpha* sp.). These are eventually displaced by slower growing, long-lived species such as mussels, sponges and/or coralline algae, and mobile organisms, such as urchins and lobsters, which feed on the fouling community. With time, a consistent increase in biomass, cover and number of species can usually be observed (Bombace *et al.* 1994; Relini *et al.* 1994; Connell & Glasby 1999). Depending on the supply of larvae and the success of recruitment, the colonisation process can take up to several years. For example, a community colonising concrete blocks in the Mediterranean was found to still be changing after five years with large algae and sponges in particular increasing in abundance (Relini *et al.* 1994). Other artificial reef communities, on the other hand, were reported to reach similar numbers of species (but not densities and biomass) to those at nearby artificial reefs within eight months (Hueckel *et al.* 1989).

The elimination of marine benthic communities in the structural footprint is an unavoidable consequence of the installation of intake and discharge structures, and no direct mitigation

measures, other than the “no go” alternative, are possible. The initial negative impacts are, however, deemed of low magnitude within the immediate vicinity of the construction sites. Furthermore, the negative impacts persist over the short-term only as the new structures will offer a new settling ground for hard bottom species and will be rapidly colonised. The impact is therefore assessed to be of **VERY LOW** significance both without and with mitigation. This rating is applicable to the construction of both the intake jetty and the discharge pipeline, regardless of location.

Elimination of benthic communities through loss of substratum in structural footprint	
CRITERIA	
Extent	Site Specific; restricted to the structural footprints of the jetty and discharge pipeline
Magnitude	Low; some biota will be eliminated or disturbed
Duration	Short-term; recolonisation in the intertidal and shallow subtidal can be expected within 2 years
Significance	VERY LOW
Probability	Definite
Confidence	Certain
Reversibility	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Mimimising wet works construction footprint and duration could marginally reduce this impact; No other direct mitigation is possible other than pursuing the “no go” alternative. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Leave pipeline in place post closure to prevent unnecessary disturbance of the seabed and associated communities.
Significance with Mitigation	VERY LOW

7.2 Operational Phase

7.2.1 Impingement and Entrainment of organisms at intake

Intake of water directly from the ocean through a submerged intake structure located in the surf zone will result in loss of marine species as a result of impingement and entrainment. Impingement refers to injury or mortality of larger organisms (e.g. fish, jellyfish) that collide with and are trapped by intake screens, whereas entrainment refers to smaller organisms that slip through the screens and are taken into the desalination plant intake ponds with the feed water. Impingement mortality is typically due to suffocation, starvation, or exhaustion due to being pinned up against the intake screens or from the physical force of the rakes used to clear screens of debris. The significance of impingement is related primarily to the location of the intake structure and is a function of intake velocity. The reduction of the average intake velocity of the feedwater to ~0.1 - 0.15 m/s, which is comparable to background currents in the ocean, will allow mobile organisms to swim away from the intake under these flow conditions (UNEP 2008). The intake of large quantities of seawater may also affect water circulation, especially in areas such as gullies and rockpools that are characterised by weak natural currents and waves.

While using screens reduces impingement, entrainment effects are likely to remain, as most of the entrained organisms are too small to be screened out without significantly reducing the intake water volume. Entrained material includes holoplanktic organisms (permanent members of the plankton, such as copepods, diatoms and bacteria) and meroplanktic organisms (temporary members of the plankton, such as juvenile shrimps and the planktonic eggs and larvae of

invertebrates and fish). Mortality rates of organisms entering desalination plants in the feedwater are likely to be 100% since the seawater is forced, at high pressure, through filters or membranes to remove particles, including the small organisms that are taken in with the feed-water. Furthermore, the feed-water will be treated with a biocide specifically designed to eliminate and kill entrained biota.

Although the mortality caused by entrainment may affect the productivity of coastal ecosystems, the effects are difficult to quantify (UNEP 2008; WHO 2007). Planktic organisms show temporal and spatial variations in species abundance, diversity and productivity, but it can be assumed that species common in the Benguela region will be prevalent in the surface waters of the project area. Furthermore, plankton species have rapid reproductive cycles. Due to these circumstances it seems unlikely that the operation of a single desalination facility of the capacity proposed at the Swakopmund Saltworks will have a substantial negative effect on the ability of plankton organisms to sustain their populations. The entrainment of eggs and larvae from common invertebrate and fish species will also unlikely adversely affect the ability of these species to reproduce successfully. The reproduction strategy of these species is to produce a large number of eggs and larvae, of which only a small percentage reaches maturity due to natural mortality (such as starvation of larvae or failure to settle in a suitable location). For example, an entrainment study for a RO Pilot Plant in San Francisco Bay showed that the estimated effects of fish larvae entrainment were minimal and indicated little potential for population-level effects (Tenera Environmental 2007). The significance of entrainment is related both to the location of the intake, as well as the overall volume of feed-water required. As the feed-water volumes required for the Rössing Uranium desalination plant are comparatively small, impingement and entrainment impacts are unlikely to be of significance.

A further issue of potential concern is the removal of particulate matter from the water column, where it is a significant source of food for surf zone and nearshore communities. For the comparatively small feed-water volumes required for the Rössing Uranium desalination plant this is unlikely to be of significance, as the surf zone in the study area is particularly productive, and particulate organic matter frequently accumulates on the shore as foam and scum.

Algal blooms, which typically develop during periods of unusually calm wind conditions when sea surface temperatures are high (February to April), can negatively impact source water quality and may result in elevated organics in the source water and accelerated biofouling of RO installations. Red tides may result in the release of algal toxins of small molecular weight, which may impact product water quality. These are, however, typically effectively removed during the reverse osmosis process. Abstraction of the feed-water at depth and a reduced intake velocity can minimise the entrance of algal material in open water intakes (UNEP 2008). For the current project, the feed-water will be abstracted from the surf zone. As the coastline of the study area is characterised by high wave energy, algal wrack often accumulates in large quantities in intertidal gullies and may thus similarly accumulate around the feed-water intake. This algal material could clog the screens at the intake and negatively impact source water quality through elevated organics. Transport of the water along the overland channel and interim storage in an inland seawater pond system will, however, allow much of this material to settle out prior to the feedwater entering the plant.

Considering the comparatively low feed-water volumes required for this project and the fact that feed-water will be abstracted from the surf zone, the loss of marine species through impingement and entrainment is deemed of low magnitude, but with impacts persisting over the operational life time of the plant. The impact is therefore assessed to be of **LOW** significance without mitigation and reducing the **VERY LOW** with the implementation of mitigation measures.

Loss of marine species through impingement and entrainment	
CRITERIA	
Extent	Site Specific; restricted to the area around the intake structure
Magnitude	Low; some impingement and entrainment of biota will occur
Duration	Long-term; over the operational life time of the plant
Significance	LOW
Probability	Definite
Confidence	Certain
Reversibility	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Adjust peak intake velocities to <0.15 m/s. Ensure installation of screens on the end of the intake pipes, or the use of a screen box or shroud. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Although an entrainment and impingement study is typically recommended for large desalination plants, the comparatively low volumes of feed-water to be extracted from the surf zone for this project would not justify such a study.
Significance with Mitigation	VERY LOW

7.2.2 Flow Distortion

The potential of scouring of sediment around the discharge outlet is a serious design issue for an effluent system discharging high volumes into a shallow receiving water body (Carter & van Ballegooyen 1998). For the current project, however, the comparatively low brine volumes (174 litres/second) and their discharge into the highly turbulent surf zone are such that the potential impacts on the limited bottom sediments present in the area are expected to be limited, and will unlikely be detectable above those resulting from natural wave action, or seasonal inshore-offshore movement of sand. Should any impacts associated with flow distortion be detectable, they would be of low magnitude within the immediate vicinity of the discharge. Despite persisting over the operational life time of the plant, the impact is deemed to be of **VERY LOW** significance when seen in context with the highly dynamic natural sediment movements typical of the coastline. This rating is applicable regardless of the location of the discharge pipeline.

Potential flow distortion around the discharge outlet	
CRITERIA	
Extent	Site Specific; restricted to the area around the discharge point
Magnitude	Very Low; flow distortion is likely to be negligible
Duration	Long-term; over the operational life time of the plant
Significance	VERY LOW
Probability	Unlikely
Confidence	Sure
Reversibility	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> No mitigation measures deemed necessary. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Design outlet velocities so as to minimise the potential for flow distortion.
Significance with Mitigation	VERY LOW

7.2.3 Desalination Plant Effluents

During operation, the desalination plant will discharge a high-salinity brine into the surf zone through a single outfall pipeline. Due to its increased salinity (at 66 psu:~1.7 times that of seawater), the brine is denser (heavier) than the surrounding seawater and would sink towards the seabed and flow away from the discharge point in the near-bottom layers of the water column, flowing down-slope (*i.e.* offshore) into deeper water. For the proposed discharge, the jet stream from the pipe end would be utilised to accelerate the brine directly into the oncoming rolling waves, thereby ensuring rapid mixing with the surrounding seawater. Depending on the discharge velocity, the volumes of brine being discharged and the local environmental conditions, thorough mixing throughout the water column is expected, but depending on the degree of mixing, the diluted brine may again sink towards the seabed and continue to dilute due to natural mixing processes. The region where the brine settles to the seafloor is termed the “near field” or “sacrificial mixing zone” as it represents an area in which large changes in water quality, sediments or biota can be expected. In other words, contaminant concentrations will be such that they will result in changes beyond natural variation in the natural diversity of species and biological communities, rates of ecosystem processes and abundance/biomass of marine life. Although the surf zone carries a significant amount of turbulent energy, it has a limited capacity to transport the brine to the open ocean. If the mass of the saline discharge exceeds the threshold of the surf zone’s salinity load transport capacity, the excess salinity would begin to accumulate in the surf zone and could ultimately result in a long-term salinity increment in this zone beyond the level of tolerance of the aquatic life (WHO 2007). This salinity threshold mixing/transport capacity of the surf zone was determined using hydrodynamic modelling.

Under the design specifications for the Rössing Uranium desalination plant project, the feed-waters will be drawn from a seawater intake located below the mean low water mark in the surf zone and is expected to be well mixed (*i.e.* no thermocline expected). Although no specific heating of the intake water will be done, transport and storage of water prior to it entering the desalination plant may potentially result in an elevation in temperature. This potential increase is assumed to be 2-4°C above ambient water temperature. On discharge, the slightly heated, dense effluent would sink towards the seabed where the receiving water masses may potentially have lower temperatures than the brine. However, discharge into the oncoming waves will ensure rapid dispersal throughout the water column, and no changes in absolute or mean temperatures of the receiving water are expected. Only under conditions of extreme calm, when the receiving waters may be stratified, would a thermal footprint be expected. Depending on the RO technology ultimately implemented, the brine may also contain traces of chemical residuals from RO membrane cleaning processes.

Table 1-1 (pg 8) lists the expected composition of the brine effluent as known at the time of compilation of this Specialist Study.

Salinity

All marine organisms have a range of tolerance to salinity, which is related to their ability to regulate the osmotic balance of their individual cells and organs to maintain positive turgor pressure. Aquatic organisms are commonly classified in relation to their range of tolerance as stenohaline (able to adapt to only a narrow range of salinities) or euryhaline (able to adapt to a wide salinity range), with most organisms being stenohaline.

Salinity changes may affect aquatic organisms in two ways:

- direct toxicity through physiological changes (particularly osmoregulation), and
- indirectly by modifying the species distribution.

Salinity changes can also cause changes to water column structure (e.g. stratification) and water chemistry (e.g. dissolved oxygen saturation and turbidity). For example, fluctuation in the salinity regime has the potential to influence dissolved oxygen concentrations, and changes in the stratification could result in changes in the distribution of organisms in the water column and sediments. Behavioural responses to changes in salinity regime can include avoidance by mobile animals, such as fish and macro-crustaceans, by moving away from adverse salinity and avoidance by sessile animals by reducing contact with the water by closing shells or by retreating deeper into sediments.

However, in marine ecosystems adverse effects or changes in species distribution are anticipated more from a reduction rather than an increase in salinity (ANZECC 2000), and most studies undertaken to date have investigated effects of a decline in salinity due to an influx of freshwater, or salinity fluctuations in estuarine environments, where most of the fauna can be expected to be of the euryhaline type. As large-scale desalination plants have only been in operation for a short period of time, very little information exists on the long-term effects of hypersaline brine on organisms in coastal marine systems (Al-Agha & Mortaja 2005). However, from the limited studies that have been published, it has been observed that salinity has a toxic effect on numerous organisms dependant on specific sensitivities (Mabrook 1994; Eniev *et al.* 2002), and by upsetting the osmotic balance, can lead to the dehydration of cells (Kirst 1989; Ruso *et al.* 2007).

Sub-lethal effects of changed salinity regimes (or salinity stress) can include modification of metabolic rate, change in activity patterns, slowing of development and alteration of growth rates (McLusky 1981; Moullac *et al.* 1998), lowering of immune function (Matozzo *et al.* 2007) and increased mortality rates (Fagundez & Robaina 1992). The limited data available include a reported tolerance of adults of the mussel *Mytilus edulis* of up to 60 psu (Barnabe 1989), and successful fertilization (Clarke 1992) and development (Bayne 1965) of its larvae at a salinity of up to 40 psu. The alga *Gracilaria verrucosa* can tolerate salinity ranges from 9-45 psu (Engledow & Bolton 1992). The shrimp *Penaeus indicus* was capable of tolerating a salinity range of 1 to 75 psu if allowed an acclimation time of around 48 hours (McClurg 1974), the oyster *Crassostrea gigas* tolerated salinities as high as 44 psu (King 1977), and the shrimp *Penaeus monodon* survived in 40 psu saline water (Kungvankij *et al.* 1986a, b, cited in DWAF 1995). Chen *et al.* (1992) reported a higher moulting frequency in juveniles of the prawn *Penaeus chinensis* at a salinity of 40 psu. Lethal effects were reported for seagrass species: for example, salinities of 50 psu caused 100% mortality of the Mediterranean seagrass *Posidonia oceanica*, 50% mortality at 45 psu, and 27% at 40 psu. Salinity concentrations above 40 psu also stunted plant growth and no-growth occurred at levels exceeding 48 psu (Latorre 2005). The high saline concentration can also lead to an increase of water turbidity, which is likely to reduce light penetration, an effect that might disrupt photosynthetic processes (Miri & Chouikhi 2005). The increased salt concentration can reduce the production of plankton, particularly of invertebrate and fish larvae (Miri & Chouikhi 2005). One of the main factors of a change in salinity is its influence on osmoregulation, which in turn affects uptake rates of chemical or toxins by marine organisms. In a review on the effects of multiple stressors on aquatic organisms, Heugens *et al.* (2001) summarise that in general metal toxicity increases with decreasing salinity, while the toxicity of organophosphate insecticides increases with increasing salinity. For other chemicals no clear relationship between toxicity and salinity was observed. Some evidence,

however, also exists for an increase in uptake of certain trace metals with an increase in salinity (Roast *et al.* 2002; Rainbow & Black 2002).

Very few ecological studies have been undertaken to examine the effects of high salinity discharges from desalination plants on the receiving communities. One example is a study on the macrobenthic community inhabiting the sandy substratum off the coast of Blanes in Spain (Raventos *et al.* 2006). The brine discharge from this plant was approximately 33,700 m³/d, more than double the effluent volume considered for the Rössing Uranium desalination plant. Visual census of the macrobenthic communities were carried out at two control points (away from the discharge outlet) and one impacted (at the discharge outlet) location several times before and after the plant began operating. No significant variations attributable to the brine discharges from the desalination plant were found. This was partly attributed to the high natural variability that is a characteristic feature of seabeds of this type, and also to the rapid dilution of the hypersaline brine upon leaving the discharge pipe. Other studies, however, indicated that brine discharges have led to reductions in fish populations, and to die-offs of plankton and coral in the Red Sea (Mabrook 1994), and to mortalities in mangrove and marine angiosperms in the Ras Hanjurah lagoon in the United Arab Emirates (Vries *et al.* 1997). Salinity increases near the outfall of a RO plant on Cyprus were reported to be responsible for a decline of macroalgae forests, and echinoderm species vanished from the discharge site (Argyrou 1999 cited in UNEP 2008).

Research conducted on abalone (*Haliotis diversicolor supertexta*) has shown that they experience significant mortality at salinities greater than 38 psu (Cheng & Chen 2000). Cheng *et al.* (2004) demonstrated that salinity stress affects the immune system of abalone, making them more vulnerable to bacterial infection. The immune capabilities in bivalve molluscs (e.g. the clam *Chamelea gallina*, Matozzo *et al.* 2007) and crustaceans (e.g. the prawn *Allacrobachium rosenbergii*, Chen & Chen 2000) have also been shown to be compromised by changes in salinity. The Indian spider lobster *Panulirus homarus*, suffered from a depressed immune system when exposed to salinities over 45 psu, subsequently resulting in 100% mortality (Verghese *et al.* 2007). Desalination plants therefore have the potential to impact on the viability of fishing industries, if the brine accumulates beyond the optimal range for commercially important species.

The South African Water Quality guidelines (DWAf 2005) set an upper target value for salinity of 36 psu. This is 1.8 psu above the median ambient salinity (34.2 psu) for the area (WSP 2014b). The paucity of information on the effects of increased salinity on marine organisms makes an assessment of the high salinity plume difficult. However, this guideline seems sufficiently conservative to suggest that no adverse effects should occur for salinity <36 psu. At levels exceeding 40 psu, however, significant effects are expected, including possible disruptions to molluscan bivalves (e.g. mussels/oysters/clams) and crustacean (and possibly fish) recruitment as salinities >40 psu may affect larval survival (e.g. Bayne 1965; Clarke 1992). This applies particularly to the larval stages of fishes and benthic organisms in the area, which are likely to be damaged or suffer mortality due to osmotic effects, particularly if the encounter with the discharge effluent is sudden.

In the case of the proposed Rössing Uranium desalination plant, the brine, which will have a salinity of ~66 psu, will be discharged through a single port diffuser into the turbulent surf zone where the effluent would be expected to be rapidly diluted. The southern site pipe discharges 70 m into the surf zone (measured from the high water mark) while the northern site will discharge at approximately 90 m into the surf zone. Toxic effects of elevated salinities are likely to be experienced only by a very limited range of sensitive species, which may consequently be excluded from the sacrificial zone and/or the discharge gully. Most intertidal and shallow subtidal species are likely to experience sub-lethal effects only, if at all, and these would be restricted to within the immediate vicinity (i.e. within the discharge gully) of the outfall. As benthic communities within this region are largely ubiquitous and naturally highly variable at temporal and spatial scales, the loss or exclusion of sensitive species within the highly localised area around the outfall can be considered insignificant in both a local and regional context.

The results from the near-field dilution modelling study (WSP 2014b) are summarised below. Assuming a discharge with an exit velocity of 6 m/s through a single port located directly above the seabed in water depths of 0.9 - 2.3 m and directed horizontally offshore, the model identified that the 18 x dilution required to meet the water quality guidelines would not be achieved within an area of 64 - 66 m² around the discharge point for a wave induced longshore current of 0.25 m/s

(energetic condition) and 0.08 m/s (calm condition) perpendicular to the jet discharge direction, respectively. These dilution predictions can be considered conservative, however, as the interaction of jet discharges with surface waves was not taken into account in the model. Under the typically rough conditions along the coastline of the study area it can be assumed that required dilutions would be achieved well within this area and the observed effluent footprints would be considerably reduced or undetectable for most of the year. Therefore, only under under 'worst-case' conditions during a very calm period for a very short time (1% of the time), would the required dilutions not be achieved. Under such calm conditions, the brine would not be sufficiently mixed and would remain close to the seabed due to its greater density. The plume may thus extend through the narrow surf zone, potentially pooling in seabed depressions, and thereby resulting in a more extensive footprint. Frequent strong wind or storm events that are typical for this coastline are, however, likely to prevent any long term cumulative build up of high-density saline pools at the seafloor. Any detrimental effects on marine organisms would thus be sub-lethal and transient, and unlikely to be detectable above natural environmental perturbations.

When oscillating tidal currents and local surf zone processes were considered by way of an intermediate dilution model, it was identified that at the southern discharge site (Option 5), the maximum area influenced by the brine was 25-30 m from the outfall diffuser in a cross-shore direction, and 35-45 m in the alongshore direction (Figure 25, top). For the northern discharge site (Option 1), the area of influence amounted to 30-40 m from the outfall diffuser in a cross-shore direction, and 35-50 m in the alongshore direction (Figure 25, bottom). Within 15 m of the discharge point, the achievable dilutions are thus achieved for most of the time, with only isolated periods of <0.5 days, when dilutions were not achieved.

The effects of elevated salinities on the physiological functioning of marine organisms is considered to be of medium magnitude and dispersion modelling results indicate that should they occur, effects will remain localised (salinities return to ambient within a maximum radius of 22 m from the diffuser port under transient, 'worst-case' conditions). Impacts will, however, persist over the operational life time of the plant. The impact is therefore assessed to be of **MEDIUM** significance without mitigation. Mitigation in the form of suitable engineering designs to ensure adequate dispersion and dilution of the brine in the receiving surf zone environment would reduce the significance to **LOW**.



Figure 25: The diluted intermediate brine influence area for the southern (top) discharge location (Option 5) and the northern (bottom) discharge location (Option 1) (from WSP 2014b).

Reduced physiological functioning of marine organisms due to elevated salinity		
CRITERIA	Outfall 1 (Northern Option)	Outfall 5 (Southern Option)
Extent	Site Specific; restricted to an area of ~5 m radius around the discharge point	Site Specific; restricted to an area ~20 m offshore and 85 m alongshore around the discharge point
Magnitude	Medium; some biota will be negatively affected	Medium; some biota will be negatively affected
Duration	Long-term; over the operational life time of the plant	Long-term; over the operational life time of the plant
Significance	MEDIUM	MEDIUM
Probability	Probable	Probable
Confidence	Sure	Sure
Reversibility	Reversible	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Ensure engineering designs at the seaward end of the discharge pipe achieve the highest required dilution of brine (18x), thereby limiting increased salinities to the minimum achievable mixing zone only. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to validate the predictions of the hydrodynamic modelling study and monitor constituents of the effluent to ensure compliance with water quality guidelines. 	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Ensure engineering designs at the seaward end of the discharge pipe achieve the highest required dilution of brine (18x), thereby limiting increased salinities to the minimum achievable mixing zone only. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to validate the predictions of the hydrodynamic modelling study and monitor constituents of the effluent to ensure compliance with water quality guidelines.
Significance with Mitigation	LOW	LOW

Avoidance of the brine footprint by marine organisms is considered to be of low magnitude and would remain confined to the mixing zone. Impacts will, however, persist over the operational life time of the plant. The impact is therefore assessed to be of **LOW** significance without mitigation, reducing to **VERY LOW** significance with the implementation of mitigation.

Avoidance behaviour by invertebrates, fish and marine mammals of the discharge area		
CRITERIA	Outfall 1 (Northern Option)	Outfall 5 (Southern Option)
Extent	Site Specific; restricted to an area of ~60 m radius around the discharge point	Site Specific; restricted to an area ~20 m offshore and 85 m alongshore around the discharge point
Magnitude	Low; some biota may avoid the mixing zone	Low; some biota may avoid the mixing zone
Duration	Long-term; over the operational life time of the plant	Long-term; over the operational life time of the plant
Significance	LOW	LOW
Probability	Probable	Probable
Confidence	Certain	Certain
Reversibility	Reversible	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Ensure engineering designs at the seaward end of the discharge pipe achieve the highest required dilution of brine (18x), thereby limiting increased salinities to the minimum achievable mixing zone only. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to validate the predictions of the hydrodynamic modelling study and monitor constituents of the effluent to ensure compliance with water quality guidelines. 	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Ensure engineering designs at the seaward end of the discharge pipe achieve the highest required dilution of brine (18x), thereby limiting increased salinities to the minimum achievable mixing zone only. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to validate the predictions of the hydrodynamic modelling study and monitor constituents of the effluent to ensure compliance with water quality guidelines.
Significance with Mitigation	LOW	LOW

Temperature

Generally, there is no heating process of the intake water in RO desalination plants. However, the temperature of the feed water may increase slightly during its passage along the onland channel and in the seawater holding pond. Such an increase is not expected to exceed 3 °C.

Bamber (1995) defined four categories for direct effects of thermal discharges on marine organisms:

- Increases in mean temperature;
- Increases in absolute temperature;
- High short term fluctuations in temperature; and
- Thermal barriers.

Increased mean temperature

Changes in water temperature can have a substantial impact on aquatic organisms and ecosystems, with the effects being separated into two groups:

- influences on the physiology of the biota (e.g. growth and metabolism, reproduction timing and success, mobility and migration patterns, and production); and
- influences on ecosystem functioning (e.g. through altered oxygen solubility).

The impacts of increased temperature have been reviewed in a number of studies along the West Coast of South Africa (e.g. Luger *et al.* 1997; van Ballegooyen & Luger 1999; van Ballegooyen *et al.* 2004, 2005). A synthesis of these findings is given below.

Most reports on adverse effects of changes in seawater temperature on southern African West Coast species are for intertidal (e.g. the white mussel *Donax serra*) or rocky bottom species (e.g. abalone *Haliotis midae*, kelp *Laminaria pallida*, mytilid mussels, Cape rock lobster *Jasus lalandii*). Cook (1978) specifically studied the effect of thermal pollution on the commercially important rock lobster *Jasus lalandii*, and found that adult rock lobster appeared reasonably tolerant of increased temperature of +6°C and even showed an increase in growth rate. The effect on the reproductive cycle of the adult lobster female was, however, more serious as the egg incubation period shortened and considerably fewer larvae survived through the various developmental stages at +6°C above ambient temperature. Zoutendyk (1989) also reported a reduction in respiration rate of adult *J. lalandii* at elevated temperatures.

Other reported effects include an increase in biomass of shallow water hake *Merluccius capensis* and West Coast sole *Austroglossus microlepis* at 18°C (MacPherson & Gordo 1992) but no influence of temperatures of <17.5°C on chub-mackerel *Scomber japonicus* (Villacastin-Herrero *et al.* 1992). In contrast, 18°C is the lower lethal limit reported for larvae and eggs of galjoen *Distichius capensis* (Van der Lingen 1994).

Internationally, a large number of studies have investigated the effects of heated effluent from coastal power stations on the open coast. These concluded that at elevated temperatures of <5°C above ambient seawater temperature, little or no effects on species abundances and distribution patterns were discernable (van Ballegooyen *et al.* 2005). On a physiological level, however, some adverse effects were observed, mainly in the development of eggs and larvae (e.g. Cook, 1978, Sandstrom *et al.* 1997; Luksiene *et al.* 2000).

The South African Water Quality Guidelines recommend that the maximum acceptable variation in ambient temperature should not exceed 1°C (DWAF 2005), which is an extremely conservative value in view of the negligible effects of thermal plumes on benthic assemblages reported elsewhere for a ΔT of +5°C or less.

All benthic species have preferred temperature ranges and it is reasonable to expect that those closest to their upper limits (*i.e.* boreal as opposed to temperate) would be negatively affected by an increase in mean temperature. The sessile biota in the Benguela region are, however, naturally exposed to wide temperature ranges due to surface heating and rapid vertical mixing of the water column and intrusions of cold bottom shelf water into the system. It can thus be assumed that the biota in these waters are relatively robust and well-adapted to substantial natural variations in temperature.

The application of the ANZECC (2000) water quality guideline (that requires that the median temperature in the environment with an operational discharge should not lie outside the 20 and 80 percentile temperature values for a reference location or ambient temperatures observed prior to the construction and operation of the proposed discharge), may be more appropriate to the high temperature variability conditions in the study area. Conditions in the surf zone are, however, expected to be well mixed and thermoclines would not be expected.

Although not modelled for the current study, no discernible temperature footprint would be expected as temperature differences between the brine and receiving waters are expected to be <3°C. Although this would not be compliant with either the South African Water Quality Guidelines (DWAF 2005) or the ANZECC (2000) guidelines, discharge into the turbulent surf zone would ensure rapid mixing of the thermal footprint with the receiving water. Furthermore, as seawater temperatures in the area vary between 10°C and 23°C (see Figure 8), the biota are well adapted to temperature fluctuations and a localised increase in temperature of <3°C is not expected to have significant effects.

Increased absolute temperature

The maximum observed sea surface temperature in the region typically is <18°C. Strong wind events and wave action in the surf zone are likely to mix the water column to such an extent that the bottom waters will have similar water temperatures to the surface waters. The discharged brine will not be heated above this naturally occurring maximum temperature and therefore an increase in absolute temperature is not expected and is not further assessed here.

Short term fluctuations in temperature and thermal barriers

Temperature fluctuations are typically caused by variability in flow or circulation driven by frequently reversing winds or tidal streams. For example, Bamber (1995) described faunal impoverishment in a tidal canal receiving hot water effluent where the temperature variability was ~12°C over each tidal cycle. As noted above, although likely well mixed by surf zone turbulence, the receiving waters in the area may vary rapidly in temperature, and the ecological effects of potential brine-induced changes of <3°C in temperature are therefore not further assessed.

For thermal barriers to be effective in limiting or altering marine organism migration paths they need to be persistent over time and cover a large cross-sectional area of the water body. The predictions for the brine plume distributions indicate that neither condition will be met in the study area. Although the migration pathways of various fish species (e.g. snoek, silver kob, dusky kob, white steenbras, Wes Coast steenbras) potentially pass through the impact area, the salinity footprint does not typically extend more than 100 m offshore and 100 m alongshore, and effects of the plume on the migratory behaviour of these species is thus considered highly unlikely.

The effects of elevated temperature on marine communities is considered to be of low magnitude. Impacts will, however, persist over the operational life time of the plant. The impact is therefore assessed to be of **VERY LOW** significance without mitigation. Mitigation in the form of suitable engineering designs to ensure adequate dispersion and mixing of the effluent in the receiving surf zone environment would reduce the probability of the impact occurring but maintain the significance at **VERY LOW**.

Reduced physiological functioning of marine organisms due to elevated temperature		
CRITERIA	Outfall 1 (Northern Option)	Outfall 5 (Southern Option)
Extent	Site Specific; restricted to an area of ~60 m radius around the discharge point	Site Specific; restricted to an area ~20 m offshore and 85 m alongshore around the discharge point
Magnitude	Very Low; some biota may avoid the mixing zone	Very Low; some biota may avoid the mixing zone
Duration	Long-term; over the operational life time of the plant	Long-term; over the operational life time of the plant
Significance	VERY LOW	VERY LOW
Probability	Probable	Probable
Confidence	Certain	Certain
Reversibility	Reversible	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Ensure engineering designs at the seaward end of the discharge pipe achieve the highest required dilution of brine (18x), thereby limiting temperature elevations to the minimum achievable mixing zone only. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to validate the predictions of the hydrodynamic modelling study and monitor constituents of the effluent to ensure compliance with water quality guidelines. 	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Ensure engineering designs at the seaward end of the discharge pipe achieve the highest required dilution of brine (18x), thereby limiting temperature elevation to the minimum achievable mixing zone only. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to validate the predictions of the hydrodynamic modelling study and monitor constituents of the effluent to ensure compliance with water quality guidelines.
Significance with Mitigation	VERY LOW	VERY LOW

Dissolved Oxygen

Dissolved oxygen (DO) is an essential requirement for most heterotrophic marine life. Its natural levels in seawater are largely governed by local temperature and salinity regimes, as well as organic content. Coastal upwelling regions are frequently exposed to hypoxic conditions owing to extremely high primary production and subsequent oxidative degeneration of organic matter. Along the southern African west coast, low-oxygen waters are a feature of the Benguela system.

Hypoxic water (<2 ml O₂/ℓ) has the potential to cause mass mortalities of benthos and fish (Diaz & Rosenberg 1995). Marine organisms respond to hypoxia by first attempting to maintain oxygen delivery (e.g. increases in respiration rate, number of red blood cells, or oxygen binding capacity of haemoglobin), then by conserving energy (e.g. metabolic depression, down regulation of protein synthesis and down regulation/modification of certain regulatory enzymes), and upon exposure to prolonged hypoxia, organisms eventually resort to anaerobic respiration (Wu 2002). Hypoxia reduces growth and feeding, which may eventually affect individual fitness. The effects of hypoxia on reproduction and development of marine animals remains almost unknown. Many fish and marine organisms can detect, and actively avoid hypoxia (e.g. rock lobster “walk-outs”). Some macrobenthos may leave their burrows and move to the sediment surface during hypoxic conditions, rendering them more vulnerable to predation. Hypoxia may eliminate sensitive species, thereby causing changes in species composition of benthic, fish and phytoplankton communities. Decreases in species diversity and species richness are well documented, and changes in trophodynamics and functional groups have also been reported. Under hypoxic conditions, there is a general tendency for suspension feeders to be replaced by deposit feeders, demersal fish by pelagic fish and macrobenthos by meiobenthos (see Wu 2002 for references). Further anaerobic degradation of organic matter by sulphate-reducing bacteria may additionally result in the production of hydrogen sulphide, which is detrimental to marine organisms (Brüchert *et al.* 2003).

Because oxygen is a gas, its solubility in seawater is dependent on salinity and temperature, whereby temperature is the more significant factor. Increases in temperature and/or salinity result in a decline of dissolved oxygen levels. The temperature of the effluent is not significantly elevated in relation to the intake water temperature, and a reduction in dissolved oxygen is thus only expected as a result of the elevated salinity of the brine. For example, saturation levels of dissolved oxygen in seawater decrease with rising salinity from 5.69 ml/ℓ at 15°C and 35 psu, to 4.54 ml/ℓ at for example 67.5 psu (DWAF 1995), not taking into account any biological use of oxygen due to respiration, oxidation and degradation. In summer months the surface water may reach temperatures of 23°C, and the saturation level of DO in the brine at this temperature would decline from 4.91 ml/ℓ at 35 psu to 3.97 ml/ℓ at 67.5 psu. These approximate calculations for example brine of 67.5 psu translate into a 19-20% reduction of DO in the brine. The South African Water Quality Guidelines for Coastal Marine Waters (DWAF 2005) state that for the west coast, the dissolved oxygen should not fall below 10% of the established natural variation. A potential difference in DO concentration of 20% is within the natural variability range of the waters in the Benguela, and the potential for a reduction in dissolved oxygen levels will also drastically reduce within a few meters of the outlet as the receiving water body is very shallow and therefore likely to be well mixed.

Near-bottom waters on the southern African West Coast are often characterised by hypoxic conditions as a result of decomposition of organic matter and low-oxygen water generation processes. A decrease in DO levels in the discharged brine is thus not of great concern. Cumulative effects may occur though during such low oxygen events but compared to the potentially large footprint of the natural hypoxic water masses, the footprint of the effluent itself will be minimal.

As discussed above, the expected changes in dissolved oxygen are associated with both direct changes in dissolved oxygen content due to the difference between the ambient dissolved oxygen concentrations and those in the effluent being discharged. However, indirect changes in dissolved oxygen content of the water column and sediments due to changes in hydrodynamic and ecosystem functioning in the area are also possible. For example, oxygen concentrations may change (particularly in the bottom waters and in the sediments) due to changes in phytoplankton production as a result of changes in nutrient dynamics (both in terms of changes in nutrient inflows and vertical mixing of nutrients) and subsequent deposition of organic matter. Several of the scale

control additives typically used in desalination plant operations have the potential to act as nutrients for plants (e.g. sodium tripolyphosphate and trisodium phosphate). In principle the phosphate can act as a plant nutrient and thus increase algal growth (Lattemann & Höpner 2003), however, phosphate generally is not limiting in marine environments, unless there are significant inputs of nitrogen (nitrates, ammonia), which is the limiting nutrient in such systems.

A critical factor that also needs to be taken into account is that oxygen depletion in the brine might occur through the addition of sodium metabisulfite (SMBS), which is commonly used as a neutralizing agent for chlorine (Lattemann & Höpner 2003) (see below). SMBS is an oxygen scavenger and if not properly dosed, can severely deplete the dissolved oxygen in the discharged water. In such cases, aeration of the effluent is recommended prior to discharge, in which case, the brine may in fact have a higher DO concentration than the receiving water body during natural low oxygen events.

The effects of reduced dissolved oxygen concentrations on marine communities are considered to be of medium magnitude and effects will likely remain localised. Impacts will, however, persist in the very short-term only, as 1) plankton blooms (should they occur) in response to elevated nutrients would be ephemeral only, and 2) accidental overdosing of SMBS would occur intermittently only, despite dechlorination being practiced over the life time of the plant. The impact is therefore assessed to be **INSIGNIFICANT**.

Reduced physiological functioning of marine organisms due to reduced dissolved oxygen concentrations		
CRITERIA	Outfall 1 (Northern Option)	Outfall 5 (Southern Option)
Extent	Site Specific; restricted to an area of ~60 m radius around the discharge point	Site Specific; restricted to an area ~20 m offshore and 85 m alongshore around the discharge point
Magnitude	Low; some biota may be affected by hypoxic conditions	Low; some biota may be affected by hypoxic conditions
Duration	Short-term; effects would be ephemeral and intermittent	Short-term; effects would be ephemeral and intermittent
Significance	VERY LOW	VERY LOW
Probability	Probable	Probable
Confidence	Certain	Certain
Reversibility	Reversible	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Avoid overdosing with SMBS or aerate effluent prior to discharge. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. 	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Avoid overdosing with SMBS or aerate effluent prior to discharge. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines.
Significance with Mitigation	VERY LOW	VERY LOW

Pretreatment of Intake Waters

Pretreatment of the intake water and periodical cleaning of the RO membranes is essential in the effective operation of desalination plants. Pretreatment and cleaning include treatment against biofouling, suspended solids and scale deposits. The type of pretreatment system used is determined primarily by the intake type (e.g. pretreatment for open water intake is generally more complex and comprehensive than that for sub-surface intakes) and the feed-water quality. Standard desalination technology typically involves chemical pretreatment as well as chemical

membrane cleaning. More recently, innovative environmentally friendly desalination technology has been developed, which involves effective filtration at the pre-treatment phase thereby minimising biofouling and eliminating the need for pretreatment biocides and membrane cleaning chemicals (i.e. no coagulants or disinfectants required).

As both technologies are being considered for the Rössing Uranium desalination plant, for the sake of completeness, an assessment of potential pretreatment chemicals and membrane cleaning additives is provided below.

The main components of a chemical pretreatment system for the desalination plant are:

- Control of biofouling by addition of an oxidising (chlorine-based) or non-oxidising (e.g. Dibromonitripropionamide (DBNPA)) biocide, and dechlorination with sodium metabisulfite (in the case of chlorine-based products),
- Removal of suspended material by flocculation (possibly bioflocculation),
- Control of scaling by acid addition (lowering the pH of the incoming seawater) and/or dosing of special 'antiscalant' chemicals,
- Cartridge filters as a final protection barrier against suspended particles and microorganisms before the RO units.

Transport of the feedwater along the overland channel and interim storage in an inland seawater pond system will facilitate the settling out of any organic material abstracted with the seawater prior to the feedwater entering the plant, thereby reducing the need for excessive biocides and/or chemicals co-discharged with the brine.

Biocides

Chlorination of the intake water is undertaken to ensure that the pumping systems (e.g. intake pipe and membranes) are maintained free of biofouling organisms. For example, larvae of sessile organisms (e.g. mussels, barnacles) can grow in the intake pipe, and impede the intake flow of the feed-water. Biofouling of the membranes by algae, fungi and bacteria can rapidly lead to the formation and accumulation of slimes and biofilms, which can increase pumping costs and reduce the lifespan of the membranes.

There are two main groups of biocides: the oxidising biocides and the non-oxidising biocides. The classification is based on the mode of biocidal action against biological material. Oxidising biocides include chlorine and bromine-based compounds and are non selective with respect to the organisms they kill. Non-oxidising biocides are more selective, in that they may be more effective against one type of micro-organisms than another. A large variety of active ingredients are used as non-oxidising biocides, including quaternary ammonium compounds, isothiazolones, halogenated bisphenols, thiocarbamates as well as others. In desalination plants, the non-oxidising Dibromonitripropionamide (DBNPA) is frequently used as an alternative to an oxidising biocide. DBNPA has extremely fast antimicrobial action and rapid degradation to relatively non-toxic end products. A summary of its environmental fate is included in Appendix A.3.

Should a biocide be required for the Rössing Uranium desalination plant, it is proposed that either sodium hypochlorite (NaOCl) or chlorine gas be used. The chlorine-based biocide should be added intermittently at the plant's intake structure as shock dosages of 10 minute duration every 4 hours. This would likely only be required in the case of a long pipeline running from the intake all the way to the plant. If the plant uses the channel system then treatment with a biocide at the seawater intake can not be permitted and dosing would occur at the plant intake. In this event the short seawater intake pipes will need to be mechanically cleaned by pigging.

Before the feed-water enters the RO units, residual chlorine needs to be neutralised with sodium metabisulfite (SMBS) to avoid membrane damage, as RO membranes are typically made from polyamide materials which are sensitive to oxidising chemicals such as chlorine. As a consequence, chlorine concentration will be very low to non-detectable in the brine effluent of the plant and is thus assumed to be below the 3 µg/ℓ limit as permitted by ANZECC (2000), which provides the most conservative guideline value (Table 4-2).

Compliance with the guidelines is thus expected, but for the sake of completeness a summary of chlorine chemistry and its potential effects on the receiving environment is provided in Appendix A.2. This serves to highlight the importance of assuring that chlorine is at all times sufficiently neutralised before discharge of the brine.

The effects of residual chlorine on marine communities are considered to be of high magnitude, but effects will likely remain localised. Impacts will persist over the medium-term as impacted marine communities will recover within 2-5 years. The impact is therefore assessed to be of **MEDIUM** significance without mitigation, but would reduce to **VERY LOW** with mitigation.

Detrimental effects on marine organisms due to residual chlorine levels in the mixing zone		
CRITERIA	Outfall 1 (Northern Option)	Outfall 5 (Southern Option)
Extent	Site Specific; restricted to an area of ~60 m radius around the discharge point	Site Specific; restricted to an area ~20 m offshore and 85 m alongshore around the discharge point
Magnitude	High; chlorine is highly toxic to marine biota	High; chlorine is highly toxic to marine biota
Duration	Short- to medium-term; impacted marine communities will recover within 2-5 years	Short- to medium-term; impacted marine communities will recover within 2-5 years
Significance	MEDIUM	MEDIUM
Probability	Definite	Definite
Confidence	Certain	Certain
Reversibility	Reversible	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Implement shock dosing of biocide in preference to continual dosing. Dechlorinate effluent prior to discharge with sodium metabisulphite (SMBS). Undertake 'pigging' of intake and discharge pipelines to reduce the need for and costs of biocides. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Use a non-oxidising biocide (DBNPA) in preference to chlorine. Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. Give serious consideration to implementing the chemical -free ProGreen technology. 	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Implement shock dosing of biocide in preference to continual dosing. Dechlorinate effluent prior to discharge with sodium metabisulphite (SMBS). Undertake 'pigging' of intake and discharge pipelines to reduce the need for and costs of biocides. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Use a non-oxidising biocide (DBNPA) in preference to chlorine. Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. Give serious consideration to implementing the chemical -free ProGreen technology.
Significance with Mitigation	VERY LOW	VERY LOW

A major disadvantage of chlorination is the formation of organohalogen compounds (e.g. trihalomethanes, see Appendix A.2). However, as only a few percent of the total added chlorine is recovered as halogenated by-products, and as by-product diversity is high, the environmental concentration of each substance can be expected to be relatively low. Dechlorination will further considerably reduce the potential for by-product formation. Nonetheless, there is some evidence that chlorinated-dechlorinated seawater increased mortality of test species and chronic effects of

dechlorinated seawater were observed, which were assumed to be due to the presence of halogenated organics formed during chlorination (see UNEP 2008 for references).

The effects of halogenated by-products on marine communities are considered to be of medium magnitude, but effects will be chronic and endure over the long-term. However, as only a very small percentage of the chlorine will transform into toxic by-products that cannot be eliminated by dechlorination, and the likelihood of halogenated by-products reaching lethal concentrations is very low the impact would reduce to be of **LOW** significance both without and with mitigation.

Chronic effects on marine organisms due to formation of halogenated by-products		
CRITERIA	Outfall 1 (Northern Option)	Outfall 5 (Southern Option)
Extent	Site Specific; restricted to an area of ~60 m radius around the discharge point	Site Specific; restricted to an area ~20 m offshore and 85 m alongshore around the discharge point
Magnitude	Medium; halogenated byproducts are toxic to marine biota, but only a small percentage of chlorine will be transformed	Medium; halogenated byproducts are toxic to marine biota, but only a small percentage of chlorine will be transformed
Duration	Long-term; over the operational life time of the plant	Long-term; over the operational life time of the plant
Significance	LOW	LOW
Probability	Unlikely	Unlikely
Confidence	Certain	Certain
Reversibility	Reversible	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> No direct mitigation is possible as chlorine chemistry is complex and type and concentrations of by-product formation cannot be predicted. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. Give serious consideration to implementing the chemical -free ProGreen technology. 	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> No direct mitigation is possible as chlorine chemistry is complex and type and concentrations of by-product formation cannot be predicted. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. Give serious consideration to implementing the chemical -free ProGreen technology.
Significance with Mitigation	LOW	LOW

De-chlorination

SMBS is a powerful reducing agent that reduces hypobromous acid (HOBr) to hydrobromic acid (HBr) and is in turn oxidised to sulfate. Although the reaction products are non-hazardous, SMBS may cause oxygen depletion if dosing is not adjusted properly. However, SMBS rapidly reacts with free chlorine but has a much slower reaction with naturally occurring dissolved oxygen. The reaction chemistry involved also means that SMBS can remove less oxygen from the seawater than the quantity of chlorine they are capable of removing. In case of overdosing with SMBS and resultant low oxygen levels, aeration of the effluent, prior to discharge may be necessary.

As marine communities in the Benguela system are adapted to naturally occurring hypoxia, the effect is considered to be of low magnitude, of localised extent and persisting over the short-term

only. The impact is therefore assessed to be of **VERY LOW** significance without mitigation, and with mitigation.

Reduction in dissolved oxygen concentrations as a result of dechlorination		
CRITERIA	Outfall 1 (Northern Option)	Outfall 5 (Southern Option)
Extent	Site Specific; restricted to an area of ~60 m radius around the discharge point	Site Specific; restricted to an area ~20 m offshore and 85 m alongshore around the discharge point
Magnitude	Low; some biota may be affected by hypoxic conditions	Low; some biota may be affected by hypoxic conditions
Duration	Short-term; effects would be ephemeral and intermittent	Short-term; effects would be ephemeral and intermittent
Significance	VERY LOW	VERY LOW
Probability	Probable	Probable
Confidence	Certain	Certain
Reversibility	Reversible	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> • Implement shock dosing of biocide in preference to continual dosing. • Avoid over-dosing of SMBS. • Aerate the effluent prior to discharge. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> • Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. • Give serious consideration to implementing the chemical -free ProGreen technology. 	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> • Implement shock dosing of biocide in preference to continual dosing. • Avoid over-dosing of SMBS. • Aerate the effluent prior to discharge. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> • Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. • Give serious consideration to implementing the chemical -free ProGreen technology.
Significance with Mitigation	VERY LOW	VERY LOW

Bacterial re-growth

Excessive bacterial re-growth in the brine after chlorination is a further concern. For example, this was reported for a RO desalination plant in Egypt (Diab 2002), where bacterial counts in the brine were 7-10 times higher than those in the feed-water thereby posing potential health risks to marine biota as well as users of the marine environment (e.g. swimmers, surfers, divers). Besides inadequate maintenance of the plant and an ineffective cleaning in place (CIP) process, excessive bacterial aftergrowth has also been attributed to the use of continuous chlorination. The reason for this ineffectiveness is that chlorination results in the breakdown of high molecular dissolved organics into nutrients, thus forming assimilable organic carbon (AOC). In addition, microorganisms subject to low levels of biocides often exude extracellular polysaccharides as a protective biofilm that increases their survival rate. Both, the availability of surplus nutrients and the survival of some microorganisms can cause a heavy re-growth in desalination systems following chlorination (UNEP 2008). For most large RO facilities, continuous chlorination has proven ineffective and has been replaced by intermittent shock chlorination. Shock dosing is also proposed for this project. In severe cases of biogrowth, additional shock treatment may become necessary to re-establish low bacterial numbers from time to time. Sodium metabisulfite is most commonly used for this purpose, with a typical application of 500-1 000 mg/ℓ for 30 minutes (Redondo & Lomax 1997). It has to be noted though that SMBS reduces bacterial numbers by oxygen depletion and is therefore

only effective against aerobic microorganisms, while some other bacteria might survive in anaerobic conditions.

The health risks associated with excessive bacterial re-growth following chlorination are considered to be of low magnitude, will likely remain localised, but may persist over the life time of the plant. The impact is therefore assessed to be of **VERY LOW** significance without mitigation. The implementation of mitigation measures would ensure that should bacterial regrowth occur, this would only persist in the short-term, and significance would remain **VERY LOW**.

Excessive bacterial re-growth in the brine after chlorination		
CRITERIA	Outfall 1 (Northern Option)	Outfall 5 (Southern Option)
Extent	Site Specific; restricted to an area of ~60 m radius around the discharge point	Site Specific; restricted to an area of ~60 m radius around the discharge point
Magnitude	Very Low; bacterial regrowth is likely to be limited	Very Low; bacterial regrowth is likely to be limited
Duration	Long-term; over the operational life time of the plant	Long-term; over the operational life time of the plant
Significance	VERY LOW	VERY LOW
Probability	Probable	Probable
Confidence	Certain	Certain
Reversibility	Reversible	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Use intermittent shock dosing with a biocide to avoid bacterial resistance to the biocide. Monitor the brine for excessive bacterial re-growth and if necessary use SMBS shock dosing to reduce bacterial numbers (note that the brine will be oxygen depleted after this treatment and needs to be aerated before discharge). <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Ensure efficient CIP process and adequate maintenance of plant. Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. Give serious consideration to implementing the chemical -free ProGreen technology. 	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Use intermittent shock dosing with a biocide to avoid bacterial resistance to the biocide. Monitor the brine for excessive bacterial re-growth and if necessary use SMBS shock dosing to reduce bacterial numbers (note that the brine will be oxygen depleted after this treatment and needs to be aerated before discharge). <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Ensure efficient CIP process and adequate maintenance of plant. Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. Give serious consideration to implementing the chemical -free ProGreen technology.
Significance with Mitigation	VERY LOW	VERY LOW

Co-discharged Waste-water Constituents

In addition to the biocide dosing, the pretreatment of the feed-water includes the removal of suspended solids, the control of scaling, and the periodical cleaning in place of the RO membranes. Specifications and volumes of cleaning chemicals that may be used in the pretreatment and CIP process and may be co-discharged with the brine effluent are listed in Table 1-1. As different chemicals are suited for different types of membranes, exact specifications for the additives will

only be known once the desalination plant operator has been appointed and the membrane type decided on. Manufacturers of RO membranes will provide relevant information in product manuals and are likely to offer consultation with regard to pretreatment and CIP chemicals. This section thus describes the use and effects of cleaning chemicals that are used conventionally in desalination plants with an open water intake.

Flocculants

Ferric chloride (FeCl_3) will be used as primary coagulant or flocculant in the pretreatment system. When added to water, a hydrolysis reaction produces an insoluble ferric hydroxide precipitate that binds non-reactive molecules and colloidal solids into larger aggregations that can then be more easily settled / floated or filtered from the water before it passes through to the RO membranes. Dosing of sulfuric acid to establish slightly acidic pH values and addition of coagulant aids such as polyelectrolytes can enhance the coagulation process. Polyelectrolytes are organic substances with high molecular masses (like polyacrylamide) that help to bridge particles together. The dosage of coagulants and coagulant aids is normally correlated with the amount of suspended material in the intake water. It can range between < 1 and 30 mg/ℓ for coagulants and between 0.2 and 4 mg/ℓ for polyelectrolytes. The resulting ferric hydroxide floc is retained when the seawater passes through the filter beds. The filters are backwashed on a periodic basis (few times every day), using filtered seawater or permeate water, to clean the particulate material off the filters. This produces a sludge that contains mainly sediments and organic matter, and filter coagulant chemicals. If co-discharged to the sea, ferric chloride may cause discoloration of the receiving water, and the sludge discharge may lead to increases in turbidity and suspended matter and has blanketing effects (Sotero-Santos *et al.* 2007, Lattemann & Höpner 2003). As a pro-active measure in favour of direct discharge to the sea, the sludge (mixed with other cleaning waste solutions) should be treated in an on-site sludge handling facility where it can be neutralised, and the solids removed and recycled or transported to a landfill site. The remaining waste-water could then be blended into and co-discharged with the brine effluent. Residual ferric hydroxide in the brine would thus be minimal to non-detectable.

After passing through the filter beds, the feed-water is put through a Dissolved Air Flotation (DAF) tank. DAF is a water treatment process that clarifies waters by the removal of suspended matter such as oil or solids. The removal is achieved by dissolving air in the water under pressure and then releasing the air at atmospheric pressure in a flotation tank or basin. The released air forms tiny bubbles which adhere to the suspended matter causing the suspended matter to float to the surface of the water where it may then be removed by a skimming device. Best practice involves keeping the supernatant water together with other cleaning waste-water, and treating this in the sludge handling facilities prior to drip-feeding the neutralised waste flows into the brine effluent.

It can be expected that the footprints for typical dilutions and dispersion of co-pollutants in the brine will be similar for a dilution factor of 18, to those obtained for salinity. The effects on marine communities of discharging co-pollutants with the brine are considered to be of low magnitude, will remain localised (within a maximum of 22 m under transient, 'worst-case' conditions), but would persist over the life time of the plant. The impact is therefore assessed to be of **LOW** significance without mitigation, but would reduce to **VERY LOW** with mitigation.

Detrimental effects on marine organisms through discharge of co-pollutants in backwash waters		
CRITERIA	Outfall 1 (Northern Option)	Outfall 5 (Southern Option)
Extent	Site Specific; restricted to an area of ~60 m radius around the discharge point	Site Specific; restricted to an area ~20 m offshore and 85 m alongshore around the discharge point
Magnitude	Low; some biota may be affected by the co-pollutants	Low; some biota may be affected by the co-pollutants
Duration	Long-term; over the operational life time of the plant	Long-term; over the operational life time of the plant
Significance	LOW	LOW
Probability	Probable	Probable
Confidence	Certain	Certain
Reversibility	Reversible	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Treat backwash waste-water in the sludge handling facility, neutralize, and remove solids for alternative disposal to a licenced facility on land. Use low-toxicity chemicals as far as practicable. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. Give serious consideration to implementing the chemical -free ProGreen technology. 	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Treat backwash waste-water in the sludge handling facility, neutralize, and remove solids for alternative disposal to a licenced facility on land. Use low-toxicity chemicals as far as practicable. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. Give serious consideration to implementing the chemical -free ProGreen technology.
Significance with Mitigation	VERY LOW	VERY LOW

Antiscalants

Scaling on the inside of tubes or on RO membranes impairs plant performance. Antiscalants are commonly added to the feed-water in desalination plants to prevent scale formation. The main representatives of antiscalants are organic, carboxylic-rich polymers such as polyacrylic acid and polymaleic acid. Acids and polyphosphates are still in use to a limited degree but are generally on the retreat as they can cause eutrophication (see for example Shams *et al.* 1994). Polyphosphate antiscalants are easily hydrolysed to orthophosphate, which is an essential nutrient for primary producers. Their use may cause a nutrient surplus and an increase in primary production at the discharge site, through formation of algal blooms and increased growth of macroalgae (DWAf 2007). When the organic material decays, this in turn can lead to oxygen depletion.

In contrast, phosphonate and organic polymer antiscalants have a low toxicity to aquatic invertebrate and fish species, but some substances exhibit an increased toxicity to algae (see UNEP 2008 for reference). The typical antiscalant dosing rate in desalination plants (1-2 mg/ℓ), however, is a factor of 10 lower than the level at which a chronic effect was observed (20 mg/ℓ), and it is 10 to 5,000 times lower than the concentrations at which acutely toxic effects were observed. It is recommended that phosphonate be used as the antiscalant for the Rössing Uranium desalination plant, with antiscalant concentration in the brine of 4-5 mg/ ℓ, which would be far below chronic effects level. Due to the antiscalants capability of binding nutrients they may, however, interfere with the natural processes of dissolved metals in seawater following discharge (see UNEP 2008 for reference). Some of these metals may be relevant micronutrients for marine algae.

The effects on marine communities of discharging antiscalants with the brine are considered to be of low magnitude, will remain localised (within a maximum of 22 m under transient, 'worst-case' conditions), but would persist over the life time of the plant. The impact is therefore assessed to be of **LOW** significance without mitigation, but would reduce to **VERY LOW** with mitigation.

Detrimental effects on marine organisms through discharge of antiscalants in backwash waters		
CRITERIA	Outfall 1 (Northern Option)	Outfall 5 (Southern Option)
Extent	Site Specific; restricted to an area of ~60 m radius around the discharge point	Site Specific; restricted to an area ~20 m offshore and 85 m alongshore around the discharge point
Magnitude	Low; some biota may be affected by the antiscalants	Low; some biota may be affected by the antiscalants
Duration	Long-term; over the operational life time of the plant	Long-term; over the operational life time of the plant
Significance	LOW	LOW
Probability	Probable	Probable
Confidence	Certain	Certain
Reversibility	Reversible	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> • Limit the use of scale-control additives to minimum practicable quantities. • Avoid antiscalants that increase nutrient levels (e.g. polyphosphate antiscalants). • Select an antiscalant that has relevant eco-toxicological testing. • Conduct Whole Effluent Toxicity (WET) testing of the brine effluent. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> • Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. • Give serious consideration to implementing the chemical -free ProGreen technology. 	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> • Limit the use of scale-control additives to minimum practicable quantities. • Avoid antiscalants that increase nutrient levels (e.g. polyphosphate antiscalants). • Select an antiscalant that has relevant eco-toxicological testing. • Conduct Whole Effluent Toxicity (WET) testing of the brine effluent. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> • Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. • Give serious consideration to implementing the chemical -free ProGreen technology.
Significance with Mitigation	VERY LOW	VERY LOW

Cleaning in Place Chemicals

Despite feed-water pretreatment, membranes may become fouled by biofilms, accumulation of suspended matter and scale deposits, necessitating periodic cleaning. In standard desalination technology plants, the cleaning intervals (CIP) of RO membranes are typically three to six months depending on the quality of the plant's feed-water (Einav *et al.* 2002). The cleaning interval suggested for the proposed desalination plant is four times per year. The chemicals used are mainly weak acids and detergents. Alkaline cleaning solutions (pH 11-12) are used for removal of silt deposits and biofilms, whereas acidified solutions (pH 2-3) remove metal oxides and scales. Further chemicals such as detergents, oxidants, complexing agents and/or non-oxidising biocides for membrane disinfection, are often added to improve the cleaning process. These additional chemicals are usually generic types or special brands recommended by the membrane manufacturers. Common cleaning chemicals include Sulphuric acid, Ethylenediaminetetra-acetic

acid (EDTA), Sodium tripolyphosphate (STPP), and Trisodium phosphate (TSP), and Dibromonitripropionamide (DBNPA) as the non-oxidising biocide. Appendix A.3 provides a short summary of the environmental fates and effects of these chemicals.

After the cleaning process is complete and the cleaning agents have been circulated through the membranes, the membranes are rinsed with product water several times. For the Rössing Uranium desalination plant project, it is proposed that the residual membrane cleaning solution and rinse water will be blended with the other residual streams from the DAF and filtration systems, and drip-fed into the brine effluent. Generally, the toxicity of the various chemicals used in the pre-treatment and CIP process (aside from biocides) is relatively low (see Appendix A.3), and none of the products are listed as tainting substances (DWAf 2005).

The effects on marine communities of discharging CIP chemicals with the brine are considered to be of very low magnitude and will likely remain localised (within a maximum of 22 m under transient, 'worst-case' conditions). As discharge will be intermittent, effects are likely to persist over the short-term only. The impact is therefore assessed to be of **VERY LOW** significance without mitigation, and with mitigation.

Detrimental effects on marine organisms or ambient seawater pH through discharge of residual cleaning solutions used periodically for cleaning in place		
CRITERIA	Outfall 1 (Northern Option)	Outfall 5 (Southern Option)
Extent	Site Specific; restricted to an area of ~60 m radius around the discharge point	Site Specific; restricted to an area ~20 m offshore and 85 m alongshore around the discharge point
Magnitude	Very Low; some biota may be affected by CIP chemicals	Very Low; some biota may be affected by CIP chemicals
Duration	Short-term; effects will be intermittent	Short-term; effects will be intermittent
Significance	VERY LOW	VERY LOW
Probability	Probable	Probable
Confidence	Certain	Certain
Reversibility	Reversible	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Collect residual cleaning solutions and membrane filter washes and neutralize and remove solids before discharge. Use low-toxicity chemicals as far as practicable. Conduct Whole Effluent Toxicity (WET) testing of the brine effluent. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. Give serious consideration to implementing the chemical -free ProGreen technology. 	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Collect residual cleaning solutions and membrane filter washes and neutralize and remove solids before discharge. Use low-toxicity chemicals as far as practicable. Conduct Whole Effluent Toxicity (WET) testing of the brine effluent. <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to monitor constituents of the effluent to ensure compliance with water quality guidelines. Give serious consideration to implementing the chemical -free ProGreen technology.
Significance with Mitigation	VERY LOW	VERY LOW

For assessment purposes, the near-field model used dilution target values of 18-times dilution. These are merely nominal conservative required dilutions that provide indicative results for

potential co-discharges. The assumption here is that the respective water quality guidelines will be sufficiently stringent for required dilutions for co-discharges of at least 18 x to be necessary. The model outputs, however, could be re-processed assuming any specified thresholds deemed to be representative of the pollutant of concern. In that sense the modelling approach utilised was entirely generic and scalable.

The area around the discharge point where the required dilution is not achieved occurs only during intermittent and short periods of extreme calm. It is unlikely that in such short time a surplus of nutrients will lead to a significant increase in algal production, or in the case of antiscalants, to a noticeable reduction in micronutrients. Mitigating measures include discharge of the brine through a diffuser, and the avoidance of polyphosphate antiscalants. A Whole Effluent Toxicity test of the discharged brine is recommended to more reliably assess the impact of any co-discharged constituents and to calculate the required dilution rate.

Heavy Metals

The brine from a desalination plant often contains low amounts of heavy metals that pass into solution when the plant's interior surfaces corrode. In RO plants, non-metal equipment and stainless steels are typically used. The RO brine may therefore contain traces of iron, nickel, chromium and molybdenum, but contamination levels are generally low (Hashim & Hajjaj 2005; Lattemann & Höpner 2003). Heavy metals tend to enrich in suspended material and finally in sediments, so that areas of restricted water exchange and soft bottom habitats impacted by the discharge could be affected by heavy metal accumulation. Many benthic invertebrates feed on this suspended or deposited material, with the risk that metals are enriched in their bodies and passed on to higher trophic levels. At this stage, no assessment of the potential concentration of heavy metals can be provided, as it is an incidental by-product of desalination plant processes. It is therefore recommended that limits are established for heavy metal concentrations in the brine discharges (see **Error! Reference source not found.** for guideline values), and the brine regularly monitored to avoid exceedance of these limits.

The effects on marine communities of heavy metals in the brine from corrosion processes are considered to be of low magnitude, but will likely remain localised. As heavy metals can accumulate in the sediments, the effects would persist in the long-term. The impact is therefore assessed to be of **LOW** significance without mitigation, and would reduce to **VERY LOW** with mitigation.

Detrimental effects on marine organisms of heavy metals from corrosion processes		
CRITERIA	Outfall 1 (Northern Option)	Outfall 5 (Southern Option)
Extent	Site Specific; restricted to an area of ~60 m radius around the discharge point	Site Specific; restricted to an area ~20 m offshore and 85 m alongshore around the discharge point
Magnitude	Low; some biota may accumulate heavy metals	Low; some biota may accumulate heavy metals
Duration	Long-term; over the operational life time of the plant and beyond	Long-term; over the operational life time of the plant and beyond
Significance	LOW	LOW
Probability	Probable	Probable
Confidence	Certain	Certain
Reversibility	Reversible	Reversible
Mitigation Measures	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Design the plant to reduce corrosion to a minimum by ensuring that dead spots and threaded connections are eliminated. Corrosion resistance is considered good when the corrosion rate is <0.1 mm/a (UNEP 2008). <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to ensure compliance with water quality guidelines. 	<p>Essential mitigation measures:</p> <ul style="list-style-type: none"> Design the plant to reduce corrosion to a minimum by ensuring that dead spots and threaded connections are eliminated. Corrosion resistance is considered good when the corrosion rate is <0.1 mm/a (UNEP 2008). <p>Best practice mitigation measures:</p> <ul style="list-style-type: none"> Implement a water quality monitoring programme to ensure compliance with water quality guidelines.
Significance with Mitigation	VERY LOW	VERY LOW

7.2.4 “No go” Alternative

During the planning stages of the proposed Rössing Uranium desalination plant development, various alternatives were considered for supplying the feedwater volumes to the RO plant and for discharge of the brine. These were presented in a series of Trade-off Studies.

The “no go” alternative for the desalination plant implies that Rössing Uranium’s commercial sustainability will be detrimentally affected due to the continued high overhead costs of purchasing sufficient water to supply their mining operations, which may jeopardise feasibility and life-of-mine.

While the “no go” option will ensure no environmental damage as a result of the desalination plant and related infrastructure, all socio-economic benefits accruing from the mine and from the supply of water in the region will not be maximised.

7.2.5 Cumulative Impacts

Anthropogenic activities in the coastal zone can result in complex immediate and indirect effects on the natural environment. Effects from disparate activities can combine and interact with each other in time and space to cause incremental or cumulative effects. Cumulative effects can also be defined as the total impact that a series of developments, either present, past or future, will have on the environment within a specific region over a particular period of time (DEAT IEM Guideline 7, Cumulative effects assessment, 2004).

To define the level of cumulative impact in the intertidal and subtidal environment, it is therefore necessary to look beyond the environmental impacts of the current project and consider also the influence of other past or future developments in the area.

The coastline of the project area cannot be considered particularly “pristine” as it is already heavily impacted by regular vehicular traffic and seasonally high visitor numbers who utilize the area primarily for rock- and surf-angling and coastal recreation. The intake pipeline and supporting jetty and the northern discharge option would be located in close proximity to the current intake structure for the Salt Works and an old decommissioned, concrete encased intake pipeline, respectively. The southern discharge option would be situated immediately south of the current bitterns discharge location of the Salt Works. The bitterns are discharged intermittently onto the beach and do not contain co-pollutants. Cumulative effects of the proposed development with existing infrastructure and discharges from the Salt Works are thus anticipated. In contrast, potential cumulative effects of discharges from the Areva RO desalination plant, located some 23 km to the north of the Salt Works near Wlotskasbaken, are unlikely.

Although it is difficult to quantify the potential cumulative impacts of the proposed desalination plant with existing infrastructure and discharges, the selection of technologies and processes proposed for the plant are state-of-the-art and every effort has been made in the planning phase to select the least environmentally damaging option for feed-water treatment and cleaning of plant components, thereby reducing discharges of hazardous components into the environment. Cumulative impacts are thus expected to remain of low intensity at the local scale, but persisting over the operational life of the plant (long-term), and are therefore rated as being of LOW significance.

8. Recommendations and Conclusions

8.1 Comparison of Alternatives

From a marine ecological perspective, there are no noteworthy reasons for preferring the one discharge site alternative over the other. The hydrodynamic modelling results, however, indicate that the plume footprints are slightly smaller at the southern site than at the northern site.

The coastline is relatively uniform over the ~4 km stretch under consideration, and is already heavily impacted by regular vehicular traffic and seasonally high visitor numbers who utilize the area primarily for rock- and surf-angling and coastal recreation. Neither of the proposed discharge sites can therefore be considered particularly “pristine”. Macrofaunal communities inhabiting the beach are relatively species poor and on some beaches in the adjacent area show signs of moderate disturbance. No unique or new species were found on any of the beaches sampled in the vicinity of the study area (Pulfrich 2007), and the species assemblages were typical of high-energy, exposed southern African West Coast beaches. None of the species encountered are currently classified as rare or endangered.

8.2 Environmental Acceptability

The main marine impacts associated with the proposed Rössing Uranium desalination plant are related to the construction of the intake and outfall structures during the construction phase, and the intake of feed water from, and consequent discharge of a high-salinity brine back into the ocean during the operational phase.

Construction Phase

Construction activities as part of the proposed development will severely impact the rocky shore and nearshore habitats and their associated communities, but the impacts will be highly localised and confined to the immediate construction area. The installation of the intake and discharge structures will result in considerable disturbance of the high-shore, intertidal and shallow subtidal habitats at the construction site. The construction will involve substantial excavation activities on the intertidal beach, concreting of pipelines and installation of the jetty on the rocky intertidal and in the surf zone, as well as extensive traffic on the shore by heavy vehicles and machinery, and the potential for associated hydrocarbon spills. Although the activities in the intertidal zone will be localised and confined to within a hundred metres of the construction site, the boulders and sediments will be completely turned over in the process and the associated macrofauna will almost certainly be entirely eliminated. Rock blasting may be necessary to remove existing bedrock to the required depth and pile driving may be required during jetty installation, resulting in disturbance of coastal and marine biota. The physical removal of sediments or bedrock in the trench will result in the total destruction of the associated sessile benthic biota. Excavating operations will also result in increased suspended sediments in the water column and physical smothering of macrofauna by the discarded sediments. However, provided construction activities are not phased over an extended period, the shoreline is not repeatedly disturbed through persistent activities and suitable post-construction rehabilitation measures are adopted (e.g. track rehabilitation, Removal of foreign construction materials which may hamper recovery of biota, backfilling excavations above mean sea level with the excavated material as trenching progresses, so as to maintain the original shore profile as far as possible), the macrofaunal communities are likely to recover in the short-to medium-term. The benthic communities of these shores are highly variable, on both spatial and temporal scales, and subject to dramatic natural fluctuations, particularly as a result of episodic disturbances such as unusual storms, and low oxygen events. As a consequence, the benthos is considered to be relatively resilient, being well-adapted to the dynamic environment, and capable of keeping pace with rapid biophysical changes (McLachlan & De Ruyck 1993). The highly localised, yet significant impacts over the short term thus need to be weighed up against the long-term benefits of the desalination plant.

Operational Phase

The key potential impacts on the marine environment of the proposed Rössing Uranium desalination plant are mostly associated with the operational phase. The impacts involve impingement and entrainment of biota at the intake point, and impacts associated with water quality due to pre-treatment of feed-water and discharge of the brine effluent.

The seawater intake considered for this project will result in impingement and entrainment of biota. Careful designing of the intake with appropriate screens can reduce impingement substantially and should be implemented. The entrainment of biological matter and suspended matter, however, cannot be eliminated but through transfer of the water along the channel and interim storage in an onland pond much of the abstracted organic material should settle out and extensive chemical pre-treatment of the feed-water can thus be avoided. Furthermore, if biofiltration as part of the ProGreen pre-treatment techniques are implemented, this would have substantial positive consequences from both environmental and operational costs perspective.

Unless the ProGreen technology is implemented, the need for pre-treatment of the feed water will also result in the use of chlorination to prevent biofouling of the pipelines and screens, and the use of other cleaning materials, which will be co-discharged with the reject brine. Under this scenario, the impacts associated with the brine discharge would include:

- the effect of elevated salinities in the discharged effluent;
- the effect of the effluent having a higher temperature than the receiving environment;
- biocidal action of residual chlorine in the effluent (residual chlorine will be neutralized with sodium metabisulfite before the feed-water reaches the RO membranes);
- the effects of co-discharged constituents in the brine;
- the removal of particulate matter from the water column where it is a significant food source, as well as changes in phytoplankton production due to changes in nutrients, water column structure and mixing processes; and
- direct changes in dissolved oxygen content due to the difference between the ambient dissolved oxygen concentrations and those in the discharged effluent (especially if sodium bisulfate is used to neutralize residual chlorine), and indirect changes in dissolved oxygen content of the water column and sediments due to changes in phytoplankton production as a result of nutrient input.

It is particularly important that the development of a coherent density flow of brine along the seabed is avoided by ensuring complete mixing in the surf zone at the point of discharge. Consequently, the effluent must be discharged in an area of relatively high wave energy where regular mixing of the water column can be expected as a result of the exposed nature of the coastline. Careful consideration of available technologies and processes in the plant design for the proposed desalination plant is thus the key issue that will allow the selection of the least environmentally damaging option for feedwater treatment, cleaning of plant components and brine disposal, thereby reducing discharges of hazardous components into the environment and ensuring adequate and rapid dilution of the effluent in the receiving water.

The near-field modelling results indicate that under average sea conditions, the predicted plume footprint for the southern discharge site is limited in spatial extent to a maximum area of 25-30 m from the outfall diffuser in a cross-shore direction, and 35-45 m in the alongshore direction. For the northern discharge site, the area of influence amounted to 30-40 m from the outfall diffuser in a cross-shore direction, and 35-50 m in the alongshore direction. Salinity would thus return to ambient levels (34.2 psu) within this area, and co-pollutants in the brine would be sufficiently diluted to no longer pose a hazard to marine biota. The maximum predicted plume footprints would be transient only and are predicted to occur approximately 1% of the time under extremely calm conditions.

8.3 General Recommendations and Mitigation Measures

8.3.1 General Recommendations relevant to Desalination Plants

The experience of existing operational desalination plants and the considerable research in the field of desalination techniques has shown that careful planning and design of the plant is vital for successful long-term plant operation (Campbell & Jones 2005; WHO 2007; UNEP 2008). From a systems engineering perspective, three criteria drive the design of a seawater reverse osmosis plant, namely, quality of feedwater, reverse osmosis membrane specification, and water quality of permeate (Jones 2008). From a marine environmental perspective, feedwater quality is the most critical issue as the quality is largely determined by the type of intake system used. As the first

step in the pre-treatment process, the type of intake structure used is a key component, as it will affect a range of source water quality parameters and will ultimately impact the performance of downstream treatment facilities. Ultimately, the choice of intake technology determines the amount of biological and suspended material pumped into the plant, and the level of chemical pollution co-discharged with the brine. Many recent studies and reviews have advocated the use of sub-surface intakes (Indirect abstraction) where possible (Lattemann & Höpner 2003; WHO 2007; Peters *et al.* 2007; Peters & Pintó 2008; National Water Commission 2008; Pankratz 2008), as these have the advantage that the overlaying sediment acts as a natural filter, thereby significantly reducing impingement and entrainment effects. This in turn protects downstream equipment, enhances process performance and reduces the capital and operating costs of the pre-treatment system. The implementation of sub-surface intakes, however, depends on the seawater demand of the plant (above certain volumes this becomes impractical) and local geological conditions. From the geophysical surveys conducted as part of the NamWater study (CSIR 2008) and the diver survey undertaken as part of this study, it is clear that there are insufficient unconsolidated sediments in the proposed Mile 6 area to facilitate the implementation of sub-surface intakes using the sediments as a natural filter.

8.3.2 Mitigation Measures

The essential mitigation measures are listed below for both the construction and operational phases of the desalination plant.

Construction Impacts

Heavy vehicle traffic associated with construction and pipeline installation must be kept to a minimum, and be restricted to clearly demarcated access routes and construction areas only. All construction activities in the coastal zone must be managed according to a strictly enforced Environmental Management Plan. Good house-keeping must form an integral part of any construction operations on the beach from start-up, including, but not limited to:

- drip trays under all vehicles parked on the beach;
- no vehicle maintenance or refuelling on beach;
- oil spill contingency plan for accidental oil spills;
- accidental diesel and hydrocarbon spills to be cleaned up accordingly; and
- no concrete mixing on the shore.

Should they be required, all blasting activities must be conducted in accordance with recognised standards and safety requirements. The area around the blasting site should be visually searched before blasting commences, and the blasting postponed should a marine mammal, sea turtle and/or flocks of swimming and diving birds be spotted within a 2-km radius around the blasting point. Following a previous blast, stunned or dead fish may attract seals and scavenging birds. The blasting programme should be scheduled to allow seals to have left the area before the next blasting event. The number of blasts should be restricted to the absolute minimum required, and should consist of smaller, quick succession blasts directed into the rock using a time-delay detonation.

Operational Impacts

Seawater Intake

In the case of pipeline intakes, there are several alternative design or mitigation measures that can completely avoid or reduce the impact of impingement. Intake velocities should be kept below ~0.15 m/s to ensure that fish and other organisms can escape the intake current. This can be achieved through a combination of pumping rates and intake design as is the case for the proposed Rössing Uranium desalination plant. Further mitigation options involve screens, which prevent the intake of fish and wrack while still allowing adequate water flow.

Furthermore, manual cleaning of the screen box and intake and seawater delivery pipelines will be necessary as marine growth, scaling and sediment settlement will occur. Most marine pipelines

employ a pigging system for regular maintenance cleaning, in which a 'pig' (bullet-shaped device with bristles) is introduced into the pipeline to mechanically clean out the structure. The pigging device is introduced at the intake structure and allowed to travel to the pump station, from where it is retrieved. For the discharge pipeline, it is introduced in the desalination plant, and is removed again on the seaward side.

Should chlorination of the intake water be necessary, this should be undertaken intermittently to ensure that the intake pipeline and feed-water pumping systems remain free of biofouling organisms, and to prevent bacterial re-growth in the brine. However, as the RO membranes are sensitive to oxidizing chemicals, neutralisation of residual chlorine, with sodium metabisulfite (SMBS), is necessary if membrane damage is to be avoided. Residual chlorine in the brine discharge must be below 3 µg/ℓ to comply with international water quality guidelines.

Scaling of the plant pipelines and RO membranes is controlled by the addition either of acid or specific antiscalant chemicals. Acids and polyphosphates cause eutrophication through formation of algal blooms and macroalgae, and should therefore be avoided. The preferred alternative would be to use phosphonate and organic polymer antiscalants, which have a low toxicity to aquatic invertebrate and fish species. These are proposed for the Rössing Uranium desalination plant. Depending on the membrane type, the antiscalant product should preferably be one for which relevant eco-toxicological testing has already been undertaken.

The recommendations provided above are in line with best practice for desalination plants of the capacity proposed by Rössing Uranium. Essential mitigation measures would comprise the use of low toxicity phosphonate and organic polymer antiscalants.

Discharges

The discharge pipe should be fitted with a suitable diffuser system at its seaward end to ensure rapid and efficient dilution of the effluent with the receiving water, thereby reducing plume footprints near the seabed and minimising impacts on marine ecology. The design of the diffuser and discharge rates would meet the requirements of the South African Marine Water Quality Guidelines and the Operational Policy for the Disposal of Land-derived Water containing Waste to the Marine Environment insofar as they are applicable to this type of installation.

During commissioning of the desalination plant, it may be necessary to discard the membrane storage solution and rinse the membranes before plant start-up. If the membrane storage solution contains a biocide or other chemicals these must either be neutralised before being discharged to sea, or the storage solution disposed of at an appropriate waste disposal facility.

Traces of residual chlorine in the brine discharge must be below 3 µg/ℓ (ANZECC (2000) guideline levels) by neutralising with sodium metabisulfite (SMBS). As marine organisms are extremely sensitive to residual chlorine, it is vital to ensure that the residual chlorine concentration in the discharged brine is at all times reduced to a level below that which may have lethal or sublethal effects on the biota, particularly the larval stages. Should the exceedance of the recommended guideline (<3 µg/ℓ) be a more persistent or recurrent event, there could be serious implications for marine biota in the discharge gully and the plant would need to be closed down until the problem has been rectified.

The use of SMBS during dechlorination is, however, associated with oxygen depletion in the effluent if overdosing occurs, as the substance is an oxygen scavenger. Shock dosing with SMBS is also an effective way of eliminating re-growth of aerobic bacteria in the discharge pipelines. Aeration of the effluent prior to discharge is therefore recommended, preferably with a permanent aeration system. Alternatively, if a permanent *in situ* effluent monitoring system is in place, aeration can be undertaken intermittently when monitoring results detect unacceptably low dissolved oxygen levels in the effluent.

If DBNPA were to be used as alternative to chlorine, mitigation measures to ensure low residuals of DBNPA in any discharge to the marine environment include appropriate design of the brine basin so as to ensure greater and sufficient dilution of the DBNPA residuals in the effluent stream and higher

degradation rate before discharge. A better option would be carefully monitored dosing to ensure minimal DBNPA concentrations in the discharge.

The solids generated by the filtration, backwash and CIP processes will be mixed with the DAF sludge. It is essential that the solids are removed in a sludge handling facility and subsequently disposed of in an accredited landfill site, and not discharge back to the sea in the brine. The remaining waste-water can then be blended into and co-discharged to sea with the brine effluent.

8.4 Monitoring Recommendations

Monitoring plays a key role in ensuring that plant operations function as intended and achieve the provision of water with minimal environmental impacts. It includes validation, operational monitoring, verification and surveillance. Validation is the process of obtaining evidence that control measures are capable of operating as required, in other words it should confirm that specific pieces of equipment achieve accepted performance standards. Operational monitoring is the planned series of observations or measurements undertaken to assess the ongoing performance of individual control measures in preventing, eliminating or reducing hazards. Operational monitoring will normally be based on simple and rapid procedures such as measurement of turbidity and chlorine residuals or inspection of the distribution system integrity. Verification provides assurance that a system as a whole is providing safe water while surveillance reviews compliance with identified guidelines standards and regulations.

8.4.1 Recommendations for Validation

International guidelines (WHO 2007; UNEP 2008) recommend that, prior to the design and construction of the desalination plant, a study be conducted on the chemical and physical properties of the raw water. A thorough raw water characterisation at the proposed intake site should include an evaluation of physical, microbial and chemical characteristics, meteorological and oceanographic data, and aquatic biology. Seasonal variations should also be taken into account. The study should consider all constituents that may impact plant operation and process performance including water temperature, total dissolved solids (TDS), total suspended solids (TSS), membrane scaling compounds (calcium, silica, magnesium, barium, etc.) and total organic carbon (TOC).

Once the desalination plant is in full operation, a monitoring program should be implemented to ensure that the required level of dilution (as predicted by the numerical modelling) is in fact achieved. Typical brine and thermal footprints need to be confirmed by sampling with a conductivity-temperature-depth (CTD) probe after an initial period of operation of the discharge both to confirm the performance of the discharge system and the numerical model predictions. This should be done for a suitably representative range of “conservative” environmental conditions, *i.e.* conditions for which dispersion of the effluent is likely to be the most limited. It is envisaged that two to three field surveys of one to two days duration would be adequate to confirm the performance of the discharge system and the accuracy of model predictions. If field observations and monitoring fail to mirror predicted results, the forecasted impacts will need to be re-assessed.

To ensure complete confidence in the potential effects of the antiscalant to be used in the desalination plant and that the co-discharged waste-water constituents are being managed to concentrations that will not have significant environmental impacts, it will be necessary to undertake toxicity testing of the discharge for a full range of operational scenarios (*i.e.* shock-dosing, etc). Such sampling and Whole Effluent Toxicity (WET) testing need only be undertaken for the duration and extent necessary to determine an effluent profile under all operational scenarios.

8.4.2 Recommendations for Operational Monitoring

To quantify the full impact of the brine discharge on the marine environment, all affected habitats and/or communities should be monitored before and during the discharge. However, prior research has indicated that this is impractical, impossible or simply unnecessary. Monitoring should rather focus on what are likely to be the most sensitive, significantly affected and/or representative species, communities or resources. The proposed discharge area includes two principal kinds of

habitat - subtidal unconsolidated sediments and reefs, and intertidal rocky shores. In both cases a suite of standard, and widely accepted techniques have been developed for the monitoring of invertebrate communities associated with these habitats. The nature of the marine environment in the Mile 6 area, however, makes the design and successful implementation of a subtidal monitoring program for this project particularly difficult. An initial diving survey has established that the habitat in the vicinity of the outlet consists of a mixture of sandy and rocky habitats. Typical tools for sampling of sandy biota (e.g. suction sampler, van Veen grab) are therefore not suitable, and a quantitative, visual census of the benthic communities by suitably qualified scientific divers is the only feasible alternative. The exposed nature of the coastline in combination with the naturally high sediment load in nearshore waters, however, results in extremely poor visibility at the seabed. Experience has shown that such conditions persist even during periods of calm weather and low sea swell. For example, despite careful preparation and close observation of the weather and sea conditions, it was not possible to establish a well informed and quantitative baseline in the Mile 6 area as part of the NamWater Study due to near-zero visibility at the study sites. Under such persistent conditions, the results of visual census diving surveys will lack the resolution required to identify any changes in the biota that might be associated with the brine discharge. Considering the high costs associated with implementing diving surveys, and the uncertainty of obtaining sufficient quantitative information a monitoring program reliant on diving surveys is thus not recommended.

A monitoring program of intertidal rocky shore communities is, however, feasible and was recommended for the Areva RO plant near Wlotzkasbaken and a baseline survey was conducted in September 2007. No subsequent impact monitoring surveys have, however, been undertaken. It is suggested that Rössing Uranium consider implementing a structured before-after/control-impact monitoring program (Underwood 1992, 1993, 1994), which would commence prior to the start of construction and continue for at least 5 years following the commencement of brine discharge. The results of such a monitoring program will not only inform/ verify the extent and magnitude of the construction impacts for the desalination plant, but also the cumulative effects of the brine discharge.

Although it is predicted that residual chlorine levels in the discharge will be below guideline levels, continuous monitoring of the effluent for residual chlorine and dissolved oxygen levels is essential. Should residual chlorine be detected in the brine, SMBS dosing should immediately be increased. This may in turn lead to reduced oxygen levels in the effluent requiring aeration of the brine before discharge. Furthermore, bacterial re-growth should be periodically assessed (every 6 months) and if high bacterial numbers are encountered in the brine, shock-dosing with SMBS should be undertaken. Continuous monitoring of oxygen levels would then indicate whether aeration of the effluent is necessary.

The waste brine often contains low amounts of heavy metals from corrosive processes, which tend to enrich in suspended material and ultimately marine sediments. It is recommended that the effluent be monitored regularly (every 6-12 months) for heavy metals until a profile of the discharge in terms of heavy metal concentrations is determined. These heavy metal concentrations in the brine effluent would then need to be assessed based on existing guidelines (DWA 2005; ANZECC 2000). A summary of these guidelines is provided in Table 4-2. An inspection program at similar intervals (6-12 months) to check corrosion levels of plant constituent parts and the physical integrity of the intake and outlet pipes and diffuser should be implemented and components replaced or modified if excessive corrosion is identified or specific maintenance is required.

8.4.3 Recommendations for Surveillance Reviews

A monitoring program should be developed to study the effects of the discharged brine on the receiving water body, and/or intertidal biological communities surrounding the discharge location, particularly as monitoring of the affected subtidal benthic communities is in this case not feasible. This recommendation is reinforced by the *Guidelines for Environmental Evaluation for Seawater Desalination* published by the South African Department of Water Affairs and Forestry (DWA 2007), in which it is stated that it is essential that the effects of the discharge of brine into any water body be monitored according to a monitoring program performed at 6-monthly intervals over a period of approximately 4 years. This monitoring program would validate numerical modelling results and/or ecological assessments based on these (see above). Depending on initial results, reduced monitoring (*i.e.* annually) may be acceptable. This monitoring will include measurement of

the main water quality parameters such as temperature, salinity and dissolved oxygen as a minimum. It is further recommended that every effort be made to publish the results in a peer-reviewed journal.

Monitoring of the intertidal benthic communities at the impact site, and comparison with communities at a further two reference sites could be used to assess the ecological impacts of the hypersaline plume.

Information from the water quality and benthic community monitoring programs should be used to develop a contingency plan that examines the risk of contamination, and considers procedures that must be implemented to mitigate any unanticipated impacts (e.g. mixing zone larger than expected under certain conditions).

8.5 Conclusions and Impact Statement

The impact assessment (Section 7 above) identified that the marine environment will be impacted to some degree during both the construction and operational phases of the proposed Rössing Uranium desalination plant. In summary:

Three negative impacts of medium significance (before mitigation) associated with the construction phase were identified:

- Disturbance and destruction of marine biota through alteration and disruption of the coastal zone during construction;
- Detrimental effects on marine biota through accidental hydrocarbon spills, concrete works and litter in the coastal zone during construction;
- Disturbance of and injury to shore birds and marine biota through blasting.

Two negative impacts of medium significance (before mitigation) associated with the operational phase were identified:

- Reduced physiological functioning of marine organisms due to elevated salinity;
- Detrimental effects on marine organisms due to residual chlorine levels in the mixing zone.

With few exceptions, recommended management actions and mitigation measures will reduce the negative impacts of medium significance to low or very low.

If all environmental guidelines, and appropriate mitigation measures advanced in this report, and the SEMP for the proposed project as a whole, are implemented, there is no reason why the proposed development of the Rössing Uranium desalination plant should not proceed. The impacts of operational discharges of brine (and potential co-pollutants) on marine water quality remains highly localised and confined to a <100 m² area around the discharge. Furthermore, as the brine is discharged into the surf zone, rapid dilution and mixing with the receiving water body is expected thereby ensuring that detectable effects on marine communities are unlikely to occur

9. REFERENCES

- Alonso, J., Alcantara-Carrio, Cabrera, J., 2002. Tourist resorts and their impact on beach erosion at Sotavento Beaches, Fuerteventura, Spain. *Journal of Coastal Research* 36, 1-7.
- Bailey G.W., Chapman, P., 1991. Chemical and physical oceanography. In: Short-term variability during an Anchor Station Study in the southern Benguela Upwelling system. *Progress in Oceanography* 28, 9-37.
- Bailey, G.W., Beyers, C.J., De B., Lipschitz, S.R., 1985. Seasonal variation of oxygen deficiency in waters off southern South West Africa in 1975 and 1976 and its relation to catchability and distribution of the Cape rock-lobster *Jasus lalandii*. *South African Journal of Marine Science* 3, 197-214.
- Bally, R., McQuaid, C.D., Brown, A.C., 1984. Shores of mixed sand and rock: an unexplored marine ecosystem. *South African Journal of Science* 80, 500-503.
- Barendse, J., Best, P.B., Thornton, M., Elwen, S.H., Rosenbaum, H.C., Carvalho, I., Pomilla, T.J.Q., Meyer, M., Leeney, R.H., 2011. Transit station or destination? Attendance patterns, movements, and abundance estimate of humpback whales off west South Africa from photographic and genotypic matching. *African Journal of Marine Science* 33: 353-373.
- Barnard, P. (ed.), 1998. Biological diversity in Namibia - a country study. Namibian National Biodiversity Task Force, Windhoek.
- Barnes, J.I., Alberts, M., 2008. Sustainable natural resource use on the coast of Namibia, DEA Research Discussion Paper. Number 78. October 2008
- Barnes, J.I., Zeybrandt, F., Kirchner, C.H., Sakko, A.L., Macgregor, J., 2004. Economic valuation of the recreational shore fishery: a comparison of techniques. In: Sumaila, U.R., Steinshamn, S.I., Skogen, M.D., Boyer, D., (Eds). *Ecological, Economic and Social Aspects of Namibian Fisheries*. Delft; Eburon Academic Publishers, 215-230.
- Bartholomae, C.H., Hagen, E., 2007. Short-term variability in alongshore winds and temperature off Swakopmund, Namibia, during a non-upwelling event in 1998-1999. *African Journal of Marine Science* 29, 141-145.
- Bartholomae, C.H., van der Plas, A.K., 2007. Towards the development of environmental indices for the Namibian shelf, with particular reference to fisheries management. *African Journal of Marine Science* 29, 25-35.
- Batty, M., Tjipute, M., Sowman., M., Cardoso, P., Fielding, P., 2005. Overview and analysis of social, economic and fisheries information to promote artisanal fisheries in the BCLME Region - Namibia, BCLME Project LMR/AFSE/03/01/B, 40pp.
- Berg, J.A., Newell, R.I.E., 1986. Temporal and spatial variations in the composition of seston available to the suspension-feeder *Crassostrea virginica*. *Estuarine & Coastal Shelf Science* 23, 375-386.
- Best, P.B., 2007. *Whales & Dolphins of the Southern African Subregion*. Cape Town: Cambridge University Press. pp 338.
- Beyers C.J. De B., 1979. Stock assessment and some morphometric and biological characteristics of the rock-lobster *Jasus lalandii* on Marshall Rocks, its main commercial fishing area off South West Africa, 1971-1974. Investigational Report of the Sea Fisheries Branch, South Africa 117, 1-26.
- Beyers, C.J. De B., Wilke, C.G., Goosen, P.C., 1994. The effects of oxygen deficiency on growth, intermoult period, mortality and ingestion rates of aquarium-held juvenile rock lobster *Jasus lalandii*. *South African Journal of Marine Science* 14, 79-88.
- Bianchi, G., Carpenter, K.E., Roux, J.-P., Molloy, F.J., Boyer, D., Boyer, H.J., 1999. FAO species identification guide for fishery purposes. Field guide to the Living Marine Resources of Namibia, 256 pp.

- Blaber, S.J.M., Blaber, T.G., 1980. Factors affecting the distribution of juvenile estuarine and inshore fish. *Journal of Fish Biology* 17, 143-162.
- Borges, P. Andrade, C., Freitas, M.C., 2002. Dune, bluff and beach erosion due to exhaustive sand mining - the case of Santa Barbara Beach, Sao Miguel (Azores, Portugal). *Journal of Coastal Research* 36, 89-95.
- Boyd, A.J., Oberholster, G.P.J., 1994. Currents off the west and south coasts of South Africa. *S. Afr. Shipping News and Fish. Ind. Rev.* 49, 26-28.
- Branch, G.M., 1981. The biology of limpets: physical factors, energy flow and ecological interactions. *Oceanography & Marine Biology: An Annual Review* 19, 235-380.
- Branch, G.M., Griffiths, C.L., 1988. The Benguela ecosystem part V: the coastal zone. *Oceanography & Marine Biology: An Annual Review* 26, 395-486.
- Brandão, A., Best, P.B., Butterworth, D.S., 2011. Monitoring the recovery of the southern right whale in South African waters. Paper SC/S11/RW18 submitted to IWC Southern Right Whale Assessment Workshop, Buenos Aires 13-16 Sept. 2011.
- Bricelj, V.M., Malouf, R.E., 1984. Influence of algal and suspended sediment concentrations on the feeding physiology of the hard clam *Mercenaria mercenaria*. *Marine Biology* 84, 155-165.
- Brown, A., 2000. Is the sandy beach isopod *Tylos granulatus* an endangered species? *South African Journal of Science* 96, 466.
- Brüchert, V., Barker Jørgensen, B., Neumann, K., Riechmann, D., Schlösser M., Schulz, H., 2003. Regulation of bacterial sulfate reduction and hydrogen sulfide fluxes in the central Namibian coastal upwelling zone. *Geochim. Cosmochim. Acta* 67(23), 4505-4518.
- Carr, A., 1987. Impact of non-degradable marine debris on the ecology and survival outlook of sea turtles. *Marine Pollution Bulletin*. 18: 352-356.
- Carter, R.A., 2008. Walvis Bay coal fired power station environmental feasibility assessment. Marine ecology specialist study on cooling water discharges, Lwandle Technologies (Pty) Ltd for Ninham Shand (Pty) Ltd
- Cetus Projects CC, 2008. Specialist Report - Potential Environmental Impacts of Proposed 2D and 3D Seismic Surveys, Namibia. Document prepared for CCA Environmental (Pty) Ltd., 35 Roeland Square, 30 Drury Lane, Cape Town, 8001.
- Chapman, P., Shannon, L.V., 1985. The Benguela Ecosystem. Part II. Chemistry and related processes. *Oceanography & Marine Biology: An Annual Review* 23, 183-251.
- Christie, N.D., 1974. Distribution patterns of the benthic fauna along a transect across the continental shelf off Lamberts Bay, South Africa. Ph.D. Thesis, University of Cape Town, 110 pp & Appendices.
- Christie, N.D., 1976. A numerical analysis of the distribution of a shallow sublittoral sand macrofauna along a transect at Lambert's Bay, South Africa. *Transactions of the Royal Society of South Africa* 42: 149-172.
- Clark, B.M., 1997a. Variation in surf zone fish community structure across a wave exposure gradient. *Estuarine & Coastal Shelf Science* 44, 659-674.
- Clark, B.M., 1997b. Dynamics and Utilisation of Surf Zone Habitats by Fish in the South-Western Cape, South Africa. Unpublished PhD Thesis, University of Cape Town.
- Clark, B.M., Atkinson, L.J., Pulfrich, A., 2005. Sandy Beach and Rocky Intertidal Monitoring Studies in the Bogenfels Mining Licence Area, Namibia. Monitoring Report 2005. Report to NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia.
- Clark, B.M., Atkinson, L.J., Steffani, N., Pulfrich, A., 2004. Sandy Beach and Rocky Intertidal Baseline Monitoring Studies in the Bogenfels Mining Licence Area, Namibia. Monitoring Report 2004. Report to NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia.
- Clark, B.M., Bennett, B.A., Lamberth, S.J., 1994. A comparison of the ichthyofauna of two estuaries and their adjacent surf zones, with an assessment of the effects of beach-seining

- on the nursery function of estuaries for fish. South African Journal of Marine Science, 14, 121-131.
- Clark, B.M., Nel, R., 2002. Baseline survey of sandy beaches in Mining Area 1. Unpublished Report, NAMDEB Diamond Corporation.
- Clark, B.M., Pulfrich, A., Atkinson, L.J., 2006. Sandy Beach and Rocky Intertidal Monitoring Studies in the Bogenfels Mining Licence Area, Namibia. Monitoring Report 2006. Report to NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia. 133pp.
- Cockroft, A.C., Schoeman, D.S., Pitcher, G.C., Bailey, G.W., Van Zyl, D.L.. 2000. A mass stranding or “walkout” of West Coast rock lobster *Jasus lalandii* in Elands Bay, South Africa: causes, results and implications. In: Von Kaupel Klein, J.C. & F.R. Schram (Eds), The Biodiversity Crises and Crustacea. Crustacean Issues 11, 673-688.
- Collins, T., Cerchio, S., Pomilla, C., Loo J., Carvalho, I., Nguouessono, S., Rosenbaum, H.C., 2008. Revised estimates of abundance for humpback whale breeding stock B1: Gabon. Paper SC60/SH28 submitted to the 60th Meeting of the Scientific Committee of the International Whaling Commission.
- COWI, 2003a. Walvis Bay Local Agenda 21 Project. Coastal area study. Intensive monitoring campaign. Municipality of Walvis Bay, Danida and COWI Consulting Engineers. April 2003.
- CSIR, 1989. A Study of Some of the Physical and Biotic Processes Affecting Dredging within the Walvis Bay Lagoon. Stellenbosch. CSIR Report EMA-C 891009.
- CSIR, 1992. Sedimentation History and Present State of the Walvis Bay Lagoon. Stellenbosch. CSIR Report EMAS-C 92040 (Unpublished). 92pp.
- CSIR, 2002. A decision support tool: Operational time for the Namdeb inshore mining project. CSIR Report ENV-S-C 2002-120.
- CSIR, 2005. Sedimentation of the Small-Craft Harbour at Swakopmund. CSIR Report ENV-S-C 2005-004, submitted to Desert Child and Windhoek Consulting Engineers, April 2005.
- CSIR, 2008a. Hydrographical and Geophysical Survey for Desalination Plant on Central West Coast of Namibia. Prepared for NamWater, prepared by Built Environment, CSIR. November 2008. pp. 8.
- CSIR, 2009. Environmental Impact Assessment for the Proposed Desalination Project at Mile 6 Near Swakopmund, Namibia, Draft EIR Report. Stellenbosch: CSIR.
- Currie, H., Grobler, K., Kemper, J., 2008. Concept note, background document and management proposal for the declaration of Marine Protected Areas on and around the Namibian offshore islands and adjacent coastal areas. Draft 4., Prepared by Feike Natural Resource Management advisors for the Ministry of Fisheries and Marine Resources, NACOMA - Namibian Coast Conservation & Management Project and World Wide Fund For Nature. pp. 119 + Appendices.
- Currie, H., Grobler, K., Kemper, J., (eds) 2009. *Namibian Islands' Marine Protected Area*. Ministry of Fisheries and Marine Resources, Namibia. http://www.nacoma.org.na/key_Activities/Marine Protected Areas.htm.
- Daly, M.A., Mathieson, A.C., 1977. The effects of sand movements on intertidal seaweeds and selected invertebrates at Bound Rock, New Hampshire. Marine Biology 43, 45-55.
- David, J.L., Benavente, E., Bres-Patry, C., Dusautoir, J.C., Echaide, M., 2004. Are neopolyploids a likely route for a transgene walk to the wild? The *Aegilops ovata* × *Triticum turgidum durum* case. *Biol. J. Linn. Soc.*, 82: 503-510.
- De Decker, A.H., 1970. Notes on an oxygen-depleted subsurface current off the west coast of South Africa. Investigational Report: Division of Sea Fisheries, South Africa, 84, 24 pp.
- Defeo, O., De Alava, A., 1995. Effects of human activities on long-term trends in sandy beach populations: wedge clam *Donax hanleyanus* in Uruguay. Marine Ecology Progress Series 123, 73-82.
- Defeo, O., McLachlan, A., 2005. Patterns, processes and regulatory mechanisms in sandy beach macrofauna: a multi-scale analysis. Marine Ecology Progress Series 295, 1-20.

- Desprez, M., 2000. Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short-and long-term post-dredging restoration. *ICES Journal of Marine Science* 57: 1428-1438.
- Dethier, M.N., 1984. Disturbance and recovery in intertidal pools: maintenance of mosaic patterns. *Ecological Monographs* 54, 99-118
- Donn, T.E., 1986. Intertidal Macrofauna. In: McLachlan, A. (ed.). *Ecological surveys of sandy beaches on the Namib coast*. Unpublished ICR Report no. 13.
- Donn, T.E., Cockcroft, A.C., 1989. Macrofaunal community structure and zonation of two sandy beaches on the central Namib coast, South West Africa/Namibia. *Madoqua* 16, 129-135.
- Drake, D.E., Cacchione, D.A., Karl, H.A., 1985. Bottom currents and sediment transport on San Pedro Shelf, California. *Journal of Sedimentary Petrology* 55, 15-28.
- Dwyer, K.L., Ryder, C.E., Prescott R., 2003. Anthropogenic mortality of leatherback sea turtles in Massachusetts waters. p. 260. In: J.A. Seminoff (compiler). *Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation*. NOAA Technical Memorandum NMFS-SEFSC-503.
- Ellingsen, K.E., 2002. Soft-sediment benthic biodiversity on the continental shelf in relation to environmental variability. *Marine Ecology Progress Series* 232, 15-27.
- Elwen, S.H., Findlay, K.P., Kiszka, J., Weir, C.R., 2011a. Cetacean research in the southern African subregion: a review of previous studies and current knowledge. *African Journal of Marine Science* 33 (3): 469-493.
- Elwen, S.H., Leeney, R.H., 2008. Report of the Namibian Dolphin Project 2008: Ecology & conservation of coastal dolphins in Namibia. Submitted to the Ministry of Fisheries & Marine Resources, 28 Oct 2008. 24 pp. & 4 appendices.
- Elwen, S.H., Leeney, R.H., 2009. Report of the Namibian Dolphin Project 2009: Ecology & conservation of coastal dolphins in Namibia. Submitted to the Ministry of Fisheries & Marine Resources, 20 Nov 2009. 25 pp. & 1 appendix.
- Elwen, S.H., Leeney, R.H., 2010. Injury and Subsequent Healing of a Propeller Strike Injury to a Heaviside's dolphin (*Cephalorhynchus heavisidii*). *Aquatic Mammals* 36 (4): 382-387.
- Elwen, S.H., Leeney, R.H., 2011. Interactions between leatherback turtles and killer whales in Namibian waters, including possible predation. *South African Journal of Wildlife Research*
- Elwen, S.H., Snyman, A., Leeney, R.H., 2011b. Report of the Namibian Dolphin Project 2010: Ecology & conservation of coastal dolphins in Namibia. Submitted to the Ministry of Fisheries & Marine Resources, 27 Feb 2011. 36 pp. & 5 appendices.
- Elwen, S.H., Best, P.B., Thornton, M., Reeb, D., 2010. Near-shore distribution of Heaviside's (*Cephalorhynchus heavisidii*) and dusky dolphins (*Lagenorhynchus obscurus*) at the southern limit of their range in South Africa. *African Zoology* 45(1).
- Elwen, S.H., Leeney, R.H., 2011. Interactions between leatherback turtles and killer whales in Namibian waters, including possible predation. *South African Journal of Wildlife Research*
- Emanuel, B.P., Bustamante, R.H., Branch, G.M., Eekhout, S., Odendaal, F.J., 1992. A zoogeographic and functional approach to the selection of marine reserves on the west coast of South Africa. *South African Journal of Marine Science* 12, 341-354.
- Emeis, K.-C., Brüchert, V., Currie, B., Endler, R., Ferdelman, T., Kiessling, A., Leipe, T., Noli-Peard, K., Struck, U., Vogt, T., 2004. Shallow gas in shelf sediments of the Namibian coastal upwelling ecosystem. *Continental Shelf Research* 24, 627-642.
- Fegley, S.R., Macdonald, B.A., Jacobsen, T.R., 1992. Short-term variation in the quantity and quality of seston available to benthic suspension feeders. *Estuarine & Coastal Shelf Science* 34, 393-412.
- Field, J.G., Griffiths, C.L., 1991. Littoral and sublittoral ecosystems of southern Africa. In: A.C. Mathieson & P.H. Nienhuis (eds). *Ecosystems of the World 24. Intertidal and Littoral Ecosystems*, Elsevier Science Publishers, Amsterdam.

- Findlay, K.P., 1996. The impact of diamond mining noise on marine mammal fauna off southern Namibia. Specialist Study #10. In: Environmental Impact Report. Environmental Evaluation Unit (ed.) Impacts of deep sea diamond mining, in the Atlantic 1 Mining Licence Area in Namibia, on the natural systems of the marine environment. No. 11-96-158, University of Cape Town. Report to De Beers Marine (Pty) Ltd. pp. 370
- Findlay, K.P., Best, P.B., Ross, G.J.B., Cockcroft, V.G., 1992. The distribution of small odontocete cetaceans off the coasts of South Africa and Namibia. *South African Journal of Marine Science* 12: 237-270.
- Finley, K.J., Miller, G.W., Davis, R.A., Greene, C.R., 1990. Reactions of belugas, (*Delphinapterus leucas*) and narwhals (*Monodon monoceros*) to ice-breaking ships in the Canadian high arctic. *Can. Bull. Fish. Aquat. Sci.* 224: 97-117.
- Flach, E., Thomsen, L., 1998. Do physical and chemical factors structure the macrobenthic community at a continental slope in the NE Atlantic? *Hydrobiologia* 375/376, 265-285.
- Glassom, D., Branch, G.M., 1997. Impact of predation by greater flamingos *Phoenicopterus ruber* on the macrofauna of two southern African Lagoons. *Mar. Ecol. Prog. Ser.* 149: 1-12.
- Gomez-Pina, G. Munoz-Perez, J. Ramirez, J.L., Ley, C., 2002. Sand dune management problems and techniques, Spain. *Journal of Coastal Research* 36, 325-332.
- Goosen, A.J.J., Gibbons, M.J., McMillan, I.K., Dale, D.C., Wickens, P.A., 2000. Benthic biological study of the Marshall Fork and Elephant Basin areas off Lüderitz. Prepared by De Beers Marine (Pty) Ltd. for Diamond Fields Namibia, January 2000. 62 pp.
- Griffin, M., 1998. The species diversity, distribution and conservation of Namibian mammals. *Biodiversity and Conservation* 7: 483-494.
- Griffin, M., Coetzee, C.G., 2005. Annotated checklist and provisional national conservation status of Namibian mammals. Technical Reports of Scientific Services No. 4. Directorate of Scientific Services, Ministry of Environment and Tourism, Windhoek, Namibia. 207 pp.
- Griffiths, M.H., Heemstra, P.C., 1995. A contribution to the taxonomy of the marine fish genus *Argyrosomus* (Perciformes: Sciaenidae), with descriptions of two new species from southern Africa. *Ichthyol. Bull., J.L.B. Smith Inst. Ichthyol.* No. 65, 40 p
- Hampton, I., 2003. Harvesting the Sea. In: Molloy, F. & T. Reinikainen (Eds), 2003. Namibia's Marine Environment. Directorate of Environmental Affairs, Ministry of Environment & Tourism, Namibia, 31-69.
- Hansen. F.C., Cloete. R.R., Verheye, H.M., 2005. Seasonal and spatial variability of dominant copepods along a transect off Walvis Bay (23°S), Namibia. *African Journal of Marine Science* 27, 55-63.
- Hays, G.C., Houghton, J.D.R., Isaacs, C., King, R.S., Lloyd, C., Lovell, P., 2004. First records of oceanic dive profiles for leatherback turtles, *Dermochelys coriacea*, indicate behavioural plasticity associated with long-distance migration. *Animal Behaviour* 67: 733-743.
- Holtzhausen, J.A., 2000. Population dynamics and life history of westcoast steenbras *Lithognathus aureti* (Sparidae), and management options for the sustainable exploitation of the steenbras resource in Namibian waters. PhD Thesis, University of Port Elizabeth: 233 pages.
- Holtzhausen, J.A., Camarada, T.G., 2007. Migratory Behaviour and Assessment of the Bronze Whaler (*Carcharhinus brachyurus*), Final Report: BCLME Project Number LMR/CF/03/16.
- Holtzhausen, J.A., Mann, B.Q., 2000. Westcoast steenbras (*Lithognathus aureti*). In Mann, B. Q. (Ed), Southern African Marine Linefish Status Reports. *Special Publication Oceanographic Research Institute* 7:152-154.
- Holtzhausen, J.A., Kirchner, C.H. 1998. Kob fishery in Namibia. *Namibia Brief* 20, 130-133.
- Holtzhausen, J.A., Kirchner, C.H., 2001. An assessment of the current status and potential yield of Namibia's northern West Coast steenbras *Lithognathus aureti* population. *South African Journal of Marine Science* 23, 157-168.

- Holtzhausen, J.A., Kirchner, C.H., Voges, S.F., 2001. Observations on the linefish resources of Namibia, 1999-2000, with special reference to West Coast steenbras and silver kob. *South African Journal of Marine Science* 23, 135-144.
- James, M.C., Ottensmeyer, C.A., Myers, R.A., 2005. Identification of high-use habitat and threats to leatherback sea turtles in northern waters: new directions for conservation. *Ecology Letters* 8: 195-201.
- Jaramillo, E., McLachlan A., Dugan, J., 1995. Total sample area and estimates of species richness in exposed sandy beaches. *Marine Ecology Progress Series* 119, 311-314.
- Kendall, M.A., Widdicombe, S., 1999. Small scale patterns in the structure of macrofaunal assemblages of shallow soft sediments. *Journal of Experimental Marine Biology & Ecology* 237:127-140.
- Kenny, A.J., Rees, H.L., Greening, J., Campbell, S., 1998. The effects of marine gravel extraction on the macrobenthos at an experimental dredge site off north Norfolk, U.K. (Results 3 years post-dredging). *ICES CM 1998/V:14*, pp. 1-8.
- Kensley, B., 1978, Interaction between coastal processes and lagoonal fauna between Walvis Bay and Lüderitzbucht, South-West Africa. *Madoqua* 11(1):55-60.
- Kensley, B.F., Penrith, M.J., 1977. Biological survey of Sandvis 1, introduction and faunal list. *Madoqua* 10(3), 181-190.
- Kensley, B.F., Penrith, M.J., 1980. The constitution of the fauna of rocky intertidal shores of South West Africa. Part III: The north coast from False Cape Frio to the Kunene River. *Cimbebasia Series A* 5(4), 201-214.
- Kinoshita, I., Fujita, S., 1988. Larvae and juveniles of blue drum, *Nibea mitsukurii*, occurring in the surf zone of Tosa Bay, Japan. *Japanese Journal of Ichthyology* 35, 25-30.
- Kirchner, C.H., 2001. Fisheries regulations based on yield per recruit analysis for the linefish silver kob *Argyrosomus inodorus* in Namibian waters. *Fisheries Research* 52, 155-167.
- Kirchner, C.H., Beyer, J.E., 1999. Estimation of total catch of silver kob *Argyrosomus inodorus* by recreational shore-anglers in Namibia using a roving-roving creel survey. *South African Journal of Marine Science* 21, 191-199.
- Kirchner, C.H., Holtzhausen, J.A., Voges, S.F., 2001. Introducing size limits as a management tool for the recreational linefishery of silver kob, *Argyrosomus inodorus* (Griffiths and Heemstra), in Namibian waters *Fisheries Management and Ecology*, 8: 227-237.
- Kirchner, C.H., Sakko, A., Barnes, J.I., 2000. Estimation of the economic value of the Namibian recreational rock-and-surf fishery. *South African Journal of Marine Science* 22, 17-25.
- Kirkman, S.P., Oosthuizen, W.H., Meyer, M.A., Kotze, P.G.H., Roux, J-P. Underhill L.G., 2007, Making sense of census and dealing with missing data: trends in pup counts of Cape fur seal *Arctocephalus pusillus pusillus* for the period 1972-2004, *African Journal of Marine Science*, 29(2),
- Lamberth, S.J., van Niekerk, L., Hutchings, K., 2008. Comparison of, and the effects of altered freshwater inflow on, fish assemblages of two contrasting South African estuaries: the cool-temperate Olifants and the warm-temperate Breede. *African Journal of Marine Science* 30: 311-336.
- Lamberth, S.J., Branch, G.M. Clark, B.M., 2010. Estuarine refugia and fish responses to a large anoxic, hydrogen sulphide, "black tide" event in the adjacent marine environment. *Estuarine, Coastal and Shelf Science*, 86: 203-215.
- Lane, S.B., Carter, R.A., 1999. Generic Environmental Management Programme for Marine Diamond Mining off the West Coast of South Africa. Marine Diamond Mines Association, Cape Town, South Africa. 6 Volumes.
- Lasiak, T.A., 1981. Nursery grounds of juvenile teleosts: evidence from surf zone of King's beach, Port Elizabeth. *South African Journal of Science* 77, 388-390.

- Leeney, R.H., Carslake, D., Elwen, S.H., 2011. Using Static Acoustic Monitoring to Describe Echolocation Behaviour of Heaviside's Dolphins in Namibia. *Aquatic Mammals* 37 (2): 151-160.
- Leeney, R.H., Post, K., Best, P.B., Hazevoet, C.J., Elwen, S.H., (in rev.) Pygmy right whale records from Namibia.
- Louw, D.C., 2008. Internal memorandum. Phytoplankton Bloom along the Namibian Coast. Ministry of Fisheries & Marine Resources: Directorate of Resource Management. Date: 10/03/2008.
- Lutcavage, M.E., Plotkin, P., Witherington, Lutz, P.L., 1997. Human impacts on sea turtle survival. *In: The Biology of Sea Turtles*, Vol. 1. P.L. Lutz and J. Musick (eds.) CRC Press, Boca Raton, Fla. pp. 387-410.
- Lutz, P.L., Alfaro-Shulman, A.A., 1991. The effects of chronic plastic ingestion on green sea turtles. Report NOAA/SB21-WCH06134, U.S. Dept. of Commerce, Miami, FL.
- Lynam, C.P., Gibbons, M.J., Axelsen, B.A., Sparks, C.A.J., Coetzee, J., Heywood, B.G., Brierley, A.S., 2006. Jelly Fish overtake fish in a heavily fished ecosystem. *Curr. Biol.* 16, 492-493.
- Matthews, S.G., Pitcher, G.C., 1996. Worst recorded marine mortality on the South African coast. *In: Harmful and Toxic Algal Blooms*. Eds. Yasumoto, T, Oshima, Y., Fukuyo, Y. Intergovernmental Oceanographic Commission of UNESCO, pp 89-92.
- Maurihungirire, M., Griffin, K., 1998. Mariculture- a Namibian Perspective. *Namibia Brief* 20, 134-135.
- McLachlan, A., 1980. The definition of sandy beaches in relation to exposure: a simple rating system. *South African Journal of Science* 76, 137-139.
- McLachlan, A., 1985. The ecology of two sandy beaches near Walvis Bay, *Madoqua* 14(2), 155-163.
- McLachlan, A., 1986. Ecological surveys of sandy beaches on the Namib coast. Report No. 13, Institute for Coastal Research, University of Port Elizabeth, Port Elizabeth, 135 pp.
- McLachlan, A., De Ruyck, A., 1993. Survey of sandy beaches in Diamond Area 1. A report to Consolidated Diamond Mines, Oranjemund, Namibia.
- McLachlan, A., Jaramillo, E., Donn, T.E., Wessels, F., 1993. Sandy beach macrofauna communities and their control by the physical environment: a geographical comparison. *Journal of coastal Research*, Special Issue 15, 27-38.
- McQuaid, C.D., Branch, G.M., 1984. Influence of sea temperature, substratum and wave exposure on rocky intertidal communities: an analysis of faunal and floral biomass. *Marine Ecology Progress Series* 19, 145-151.
- McQuaid, C.D., Branch, G.M., Crowe, A.A., 1985. Biotic and abiotic influences on rocky intertidal biomass and richness in the southern Benguela region. *South African Journal of Zoology* 20, 115-122.
- Mecenero, S., Roux, J-P., Underhill, L.G., Bester, M.N., 2006. Diet of Cape fur seals *Arctocephalus pusillus pusillus* at three mainland breeding colonies in Namibia. 1. Spatial variation. *African Journal of Marine Science* 28: 57-71.
- Meyer, W.F., Ewart-Smith, C., Nel, R., Clark, B.M., 1998. Ecological impact of beach diamond mining on beach, rocky intertidal and surf zone biological communities in the Sperrgebiet, Namibia. Phase 1 - baseline surveys. AEC Unpublished Report. 77 pp.
- MFMR, 2001. Towards Responsible Development of Aquaculture. Namibia's Aquaculture Policy. Ministry of Fisheries and Marine Resources (MFMR), Namibia, 22pp.
- MFMR, 2004. Namibia's Aquaculture Strategic Plan, Ministry of Fisheries and Marine Resources (MFMR), Namibia, 34pp.
- MFMR, 2008. Ministry of Fisheries & Marine Resources: Directorate of Resource Management. Monthly Marine Environmental Update. March 2008. No. 03/2008. Compiled by C.H. Bartholomae.

- Modde, T., 1980. Growth and residency of juvenile fishes within a surf zone habitat in the Gulf of Mexico. *Gulf Research Reports* 6, 377-385.
- Molloy, F., Reinikainen T., (Eds), 2003. Namibia`s Marine Environment. Directorate of Environmental Affairs, Ministry of Environment & Tourism, Namibia, 160 pp.
- Monteiro, P.M.S., Van Der Plas, A.K., 2006. Low Oxygen Water (LOW) variability in the Benguela System: Key processes and forcing scales relevant to forecasting. In: Shannon V, Hempel G, Malanotte-Rizzoli P, Moloney C & J Woods (eds). *Large Marine Ecosystems*, Vol. 15, pp 91-109.
- Nashima, F.P., 2013. Structure and composition of intertidal communities at exposed and sheltered habitat, Central Namibian Coast. *Journal of Agriculture and Biodiversity Research* 2(3): 67-72.
- Nel, R., Ewart-Smith, C., Meyer, W.F., Clark, B.M., 1997. Macrofauna baseline study of three pocket-beaches in the Sperrgebiet, Namibia. AEC Unpublished Report. 23pp.
- Nelson, G., 1989. Poleward motion in the Benguela area. In: Poleward Flows along Eastern Ocean Boundaries. Neshyba et al. (eds) New York; Springer, 110-130 (Coastal and Estuarine Studies 34).
- Nelson, G., Hutchings, L., 1983. The Benguela Upwelling area. *Prog. Oceanogr.* 12, 333-356.
- Newman, G.G., Pollock, D.E., 1971. Biology and migration of rock lobster *Jasus lalandii* and their effect on availability at Elands Bay, South Africa. Investigational Report: Division of Sea Fisheries, South Africa 94, 1-24.
- NRC (National Research Council). 1990. Decline of the sea turtles: causes and prevention. National Academy Press, Washington, D.C.
- O'Connor, S.R., Campbell, H., Cortez, Knowles, T., 2009. Whale Watching Worldwide: tourism numbers, expenditures and expanding economic benefits, a special report from the International Fund for Animal Welfare, Yarmouth MA, USA, prepared by Economists at Large.
- Oosthuizen, W.H., 1991. General movements of South African (Cape) fur seals *Arctocephalus pusillus pusillus* from analysis of recoveries of tagged animals. *S. Afr. J. Mar. Sci.* 11: 21-30.
- Parkins, C.A., Field, J.G., 1998. The effects of deep sea diamond mining on the benthic community structure of the Atlantic 1 Mining Licence Area. Annual Monitoring Report - 1997. Prepared for De Beers Marine (Pty) Ltd by Marine Biology Research Institute, Zoology Department, University of Cape Town. pp. 44.
- Parry, D.M., Kendall, M.A., Pilgrim, D.A., Jones, M.B., 2003. Identification of patch structure within marine benthic landscapes using a remotely operated vehicle. *J. Exp. Mar. Biol. Ecol.* 285- 286: 497-511.
- Penrith, M.-L., Kensley, B.F., 1970a. The constitution of the intertidal fauna of rocky shores of South West Africa. Part I. Lüderitzbucht. *Cimbebasia Series A* 1(9), 191-239.
- Penrith, M.-L., Kensley, B.F., 1970b. The constitution of the fauna of rocky intertidal shores of South West Africa. Part II. Rocky Point. *Cimbebasia Series A* 1(10), 243-268.
- Pitcher, G.C., 1998. Harmful algal blooms of the Benguela Current. IOC, World Bank and Sea Fisheries Research Institute Publication. 20 pp.
- Pollock, D.E., Shannon, L.V., 1987. Response of rock-lobster populations in the Benguela ecosystem to environmental change - a hypothesis. *South African Journal of Marine Science* 5, 887-899.
- Potter, I.C., Beckley, L.E., Whitfield, A.K., Lenanton, R.C.J., 1990. Comparisons between the roles played by estuaries in the life cycles of fishes in temperate Western Australia and southern Africa. *Environmental Biology of Fishes* 28, 143-178.
- Potts, W.M., Heriques, R., Sauer, WHH, Santos, C.V., Shaw, P., 2012. Are changing water temperatures in the Benguela about to alter the evolutionary history of our *Argyrosomus* fishes? Proceedings of the 4th South African Marine Linefish Symposium, in press.

- Potts, W.M., Sauer, W.H.H., Henriques, R., Sequesseque, S., Santos, C.V., Shaw, P.W., 2010. The biology, life history and management needs of a large sciaenid fish, *Argyrosomus coronus* in Angola. *African Journal of Marine Science* 32 (2): 247 – 258.
- Potts, W.M., Sauer, W.H.H., Henriques, R., Kirchner, K., Munnick, K., Santos, C.V., Shaw, P.W., in prep. Rapid climate driven distributional shifts, complicates coastal fisheries management and alters the evolutionary history of fishes.
- Pulfrich, A., 1998. Assessment of the impact of diver-operated nearshore diamond mining on marine benthic communities in the Kerbe Huk area, Namibia. Report to NAMDEB Diamond Corporation (Pty) Ltd, 29pp.
- Pulfrich, A., 2004a. Baseline Survey of Sandy Beach Macrofaunal Communities at Elizabeth Bay: Beach Monitoring Report - 2004. Prepared for NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia, on behalf of CSIR Environmentek, 53pp.
- Pulfrich, A., 2004b. Baseline survey of intertidal and subtidal rocky shore communities at Elizabeth Bay: Intertidal and subtidal monitoring report - 2004. Prepared for NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia, on behalf of CSIR Environmentek, 36pp.
- Pulfrich, A., 2005. Survey of intertidal and subtidal rocky shore communities at Elizabeth Bay: Intertidal and subtidal monitoring report - 2005. Report to NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia, 39pp.
- Pulfrich, A., 2006. Survey of intertidal and subtidal rocky shore communities at Elizabeth Bay: Intertidal and subtidal monitoring report - 2006. Report to NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia, 52pp.
- Pulfrich, A., 2007a. Survey of intertidal and subtidal rocky shore communities at Elizabeth Bay: Intertidal and subtidal monitoring report - 2007. Report to NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia, 64pp.
- Pulfrich, A., 2007b Intertidal sandy beach communities at Wlotzkasbaken, Namibia. Trekkopje desalination plant: baseline monitoring report. Prepared by Pisces Environmental Services for Turgis Consulting (Pty) Ltd, pp. 38 + Appendix.
- Pulfrich, A., Atkinson, L.J., 2007. Monitoring environmental effects of sediment discharges from the Marine Dredging Treatment Plant on sandy beach and rocky intertidal biota in Mining Area 1, Namibia. Prepared for NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia, 85pp.
- Pulfrich, A., Parkins, C.A., Branch, G.M., 2003a. The effects of shore-based diamond-diving on intertidal and subtidal biological communities and rock lobsters in southern Namibia. *Aquatic Conservation: Marine & Freshwater Ecosystems* 13, 257-278.
- Pulfrich, A., Parkins, C.A., Branch, G.M., Bustamante, R.H., Velásquez, C.R., 2003b. The effects of sediment deposits from Namibian diamond mines on intertidal and subtidal reefs and rock-lobe populations. *Aquatic Conservation: Marine & Freshwater Ecosystems* 13, 233-255.
- Pulfrich, A., Penney, A.J., 1998. Assessment of the impact of diver-operated nearshore diamond mining on marine benthic communities in the Zweispitz area, Namibia. Report to NAMDEB Diamond Corporation (Pty) Ltd, 33pp.
- Pulfrich, A., Penney, A.J. 1999. Assessment of the impact of diver-operated nearshore diamond mining on marine benthic communities near Lüderitz, Namibia. Final Report to NAMDEB Diamond Corporation (Pty) Ltd, 33pp.
- Pulfrich, A., Penney, A.J., 1999b. The effects of deep-sea diamond mining on the benthic community structure of the Atlantic 1 Mining Licence Area. Annual Monitoring Report - 1998. Prepared for De Beers Marine (Pty) Ltd by Marine Biology Research Institute, Zoology Department, University of Cape Town and Pisces Research and Management Consultants CC. pp 49.
- Pulfrich, A., Penney, A.J. 2001. Assessment of the impact of diver-operated nearshore diamond mining on marine benthic communities near Lüderitz, Namibia. Phase III. Report to NAMDEB Diamond Corporation (Pty) Ltd, 50pp.

- Pulfrich, A., Penney, A.J., Brandão, A., Butterworth, D.S., Noffke, M., 2006. Marine Dredging Project: FIMS Final Report. Monitoring of Rock Lobster Abundance, Recruitment and Migration on the Southern Namibian Coast. Prepared for De Beers Marine Namibia, July 2006. 149pp.
- Pulfrich, A., Steffani, N., 2007. Environmental impact assessment: Proposed Trekkopje Reverse Osmosis Plant. Marine Ecological Aspects. Prepared by Pisces Environmental Services for Turgis Consulting (Pty) Ltd., pp. 119 + Appendices.
- Pulfrich, A., Steffani, N., 2008. Report on Swakopmund Field Survey - Marine Baseline Assessment. In: CSIR 2009. Environmental Impact Assessment for the Proposed Desalination Project at Mile 6 near Swakopmund, Namibia, Draft EIA Report, CSIR Report No. CSIR/CAS/EMS/ER/2009/0015/A, Stellenbosch.
- Rogers, J., 1979. Dispersal of sediments from the Orange River along the Namibian desert coast. S. Afr. J. Sci. 75(12), p567.
- Rogers, J., Bremner, J.M., 1991. The Benguela Ecosystem. Part VII. Marine-geological aspects. Oceanography & Marine Biology: An Annual Review 29, 1-85.
- Rose, B., Payne, A.I., 1991. Occurrence and behavior of the southern right whale dolphin, *Lissodelphis peronii*, off Namibia. *Marine Mammal Science* 7(1): 25-34.
- Ross, G.J.B., 1984. The smaller cetaceans of the east coast of southern Africa. *Ann. Cape. Prov. Mus. (nat. Hist.)*, 15 (2).
- Roux, J-P., Best, P.B., Stander, P.E., 2001. Sightings of southern right whales (*Eubalaena australis*) in Namibian waters, 1971-1999. *Journal of Cetacean Research and Management* (Special Issue 2): 181-185.
- Roux, J-P., Braby, R., Best, P.B., 2010. Southern right whales off Namibia and their relationship with those off South Africa. Paper SC/S11/RW16 submitted to IWC Southern Right Whale Assessment Workshop, Buenos Aires 13-16 Sept. 2011.
- Royal HaskoningDSV & WSP, 2014a. Trade-off Study Report 1: Intake Site Selection. Prepared for Gecko Namibia (Pty) Ltd. September 2014. 26pp.
- Royal HaskoningDSV & WSP, 2014b. Trade-off Study Report 3: Brine Discharge Methods. Prepared for Gecko Namibia (Pty) Ltd. September 2014. 21pp.
- Royal HaskoningDSV & WSP, 2014c. Trade-off Study Report 2: Brine Outfall Locations. Prepared for Gecko Namibia (Pty) Ltd. September 2014. 23pp.
- Samuel, Y., Morreale, S.J., Clark, C.W., Greene, C.H., Richmond, M.E., 2005. Underwater, low frequency noise in a coastal sea turtle habitat. *Journal of the Acoustical Society of America* 117: 1465-1472.
- SAN (South African Navy), 2007. South African Tide Tables, Hydrographer, South African Navy, Private Bag X1, Tokai 7966, 260pp.
- Seiderer, L.J., Newell, R.C., 1999. Analysis of the relationship between sediment composition and benthic community structure in coastal deposits: Implications for marine aggregate dredging. *ICES Journal of Marine Science* 56: 757-765.
- Shannon L.V., Pillar, S., 1985. The Benguela Ecosystem III. Plankton. *Oceanography & Marine Biology: An Annual Review* 24, 65-170.
- Shannon, L.J., Moloney, C.L., Jarre, A., Field, J.G., 2003. Trophic flows in the southern Benguela during the 1980s and 1990s. *Journal of Marine Systems* 39, 83 - 116.
- Shannon, L.V., 1985. The Benguela Ecosystem. Part 1: Evolution of the Benguela physical features and processes. *Oceanography & Marine Biology: An Annual Review* 23, 105-182.
- Shannon, L.V., Anderson, F.P., 1982. Application of satellite ocean colour imagery in the study of the Benguela Current system. *South African Journal of Photogrammetry, Remote Sensing and Cartography* 13(3), 153-169.
- Shannon, L.V., Pillar, S., 1985. The Benguela Ecosystem, Part III. Plankton. *Oceanography & Marine Biology: An Annual Review*, 24: 65-170.

- Shannon, L.V., O'Toole, M.J., 1998. BCLME Thematic Report 2: Integrated overview of the oceanography and environmental variability of the Benguela Current region. Unpublished Report, 58pp.
- Shillington, F.A., Peterson, W.T., Hutchings, L., Probyn, T.A., Waldron, H.N., Agenbag, J.J., 1990. A cool upwelling filament off Namibia, South West Africa: Preliminary measurements of physical and biological properties. *Deep-Sea Res.* 37 (11A), 1753-1772.
- Simmons, R.E., Barnes, K.N., Jarvis, A.M., Robertson, A., 1999. Important Bird Areas in Namibia. Research Discussion Paper No. 31. DEA of MET, 68pp.
- Sink, K., Holness, S., Harris, L., Majiedt, P., Atkinson, L., Robinson, T., Kirkman, S., Hutchings, L., Leslie, R., Lamberth, S.J., Kerwath, S., Von Der Heyden, S., Lombard, A., Attwood, C., Branch, G., Fairweather, T., Taljaard, S., Weerts, S., Cowley, P., Awad, A., Halpern, B., Grantham, H., Wolf, T., 2012. National Biodiversity Assessment 2011: Marine and Coastal Component Report. Pretoria: South African National Biodiversity Institute
- Skov, H., Bloch, R., Stuer-Lauridsen, F., Uushona, D., 2008. Strategic Environmental Assessment (SEA) for the coastal areas of the Erongo and Kunene Regions, Prepared for Namibian Coast Conservation and Management Project (NACOMA) 172pp.
- Smith, G.G., Mocke, G.P., 2002. Interaction between breaking/broken waves and infragravity-scale phenomena to control sediment suspension and transport in the surf zone. *Marine Geology* 187, 320-345.
- Snelgrove, P.V.R., Butman, C.A., 1994. Animal-sediment relationships revisited: cause versus effect. *Oceanography & Marine Biology: An Annual Review* 32, 111-177.
- Stage, J., Kirchner, C.H., 2005. An economic comparison of the commercial and recreational linefisheries in Namibia. *African Journal of Marine Science* 27, 577-584.
- Steffani, N., 2007a. Biological Baseline Survey of the Benthic Macrofaunal Communities in the Atlantic 1 Mining Licence Area and the Inshore Area off Pomona for the Marine Dredging Project. Prepared for De Beers Marine Namibia (Pty) Ltd. pp. 42 + Appendices.
- Steffani, N., 2007b. Biological Monitoring Survey of the Macrofaunal Communities in the Atlantic 1 Mining Licence Area and the Inshore Area between Kerbehuk and Bogenfels. 2005 Survey. Prepared for De Beers Marine Namibia (Pty) Ltd. pp. 51 + Appendices.
- Steffani, N., 2009a. Biological Monitoring Surveys of the Benthic Macrofaunal Communities in the Atlantic 1 Mining Licence Area and the Inshore Area - 2006/2007. Prepared for De Beers Marine Namibia (Pty) Ltd. (Confidential Report) pp. 81 + Appendices.
- Steffani, N., 2009b. Assessment of Mining Impacts on Macrofaunal Benthic Communities in the Northern Inshore Area of the De Beers ML3 Mining Licence Area - 18 Months Post-mining. Prepared for De Beers Marine. pp. 47 + Appendices.
- Steffani, N., 2009c. Baseline Study on Benthic Macrofaunal Communities in the Inner Shelf Region and Assessment of Mining Impacts off Chameis. November 2009. Prepared for Namdeb. pp. 45 + Appendices.
- Steffani, N., 2010. Biological Monitoring Surveys of the Benthic Macrofaunal Communities in the Atlantic 1 Mining Licence Area - 2008. Prepared for De Beers Marine Namibia (Pty) Ltd. (Confidential).
- Steffani, C.N., Pulfrich, A., 2004a. Environmental Baseline Survey of the Macrofaunal Benthic Communities in the De Beers ML3/2003 Mining Licence Area. Prepared for De Beers Marine South Africa, April 2004., 34pp.
- Steffani, C.N., Pulfrich, A., 2004b. The potential impacts of marine dredging operations on benthic communities in unconsolidated sediments. Specialist Study 2. Specialist Study for the Environmental Impact Report for the Pre-feasibility Phase of the Marine Dredging Project in Namdeb's Atlantic 1 Mining Licence Area and in the nearshore areas off Chameis. Prepared for PISCES Environmental Services (Pty) Ltd, September 2004.
- Steffani, C.N., Pulfrich, A., 2007. Biological Survey of the Macrofaunal Communities in the Atlantic 1 Mining Licence Area and the Inshore Area between Kerbehuk and Lüderitz 2001 - 2004 Surveys. Prepared for De Beers Marine Namibia, March 2007, 288pp.

- Stuart, C.T., 1975. Marine fauna collected at Sandwich Harbour, Namibia Desert Park, South West Africa. *Madoqua Series II* 4, 101-102.
- Tarr, J.G., Griffiths, C.L., Bally, R., 1985. The ecology of three sandy beaches on the Skeleton Coast of South West Africa, *Madoqua* 14(3), 295-304.
- Timonin, A.G., Arashkevich, E.G., Drits, A.V., Semenova, T.N., 1992 Zooplankton dynamics in the northern Benguela ecosystem, with special reference to the copepod *Calanoides carinatus*. In *Benguela Trophic Functioning*. Payne, A. I. L., Brink, K. H., Mann, K. H. and R. Hilborn (Eds). *S. Afr. J. mar. Sci.* 12, 545-560.
- Tjipute, M., Shuuluka, D., 2006. Zoobenthos survey in the Walvis Bay Lagoon. University of Namibia, Windhoek. Report to the Municipality of Walvis Bay.
- Tomalin, B.J., 1995. Specialist study on impacts of diamond mining on rock-lobster populations and the fishery. In: CSIR 1996. Environmental Impact Assessment for marine diamond mining in the Namibian Islands concession, Council for Scientific and Industrial Research Report EMAS-C96023. 33pp.
- Van Dalssen, J.A., Essink, K., Toxvig Madsen, H., Birklund, J., Romero, J., Manzanera, M., 2000. Differential response of macrozoobenthos to marine sand extraction in the North Sea and the Western Mediterranean. *ICES J. Mar. Sci.* 57: 1439-1445.
- Van Der Merwe, D., Van Der Merwe, D., 1991. Effects of off-road vehicles on the macrofauna of a sandy beach. *South African Journal of Science* 87, 210-213.
- Van Tamelen, P.G., 1996. Algal zonation in tidepools: experimental evaluation of the roles of physical disturbance, herbivory and competition. *Journal of Experimental Marine Biology & Ecology* 201, 197-231.
- Visser, G.A., 1969. Analysis of Atlantic waters off the coast of southern Africa. Investigational Report Division of Sea Fisheries, South Africa, 75, 26pp.
- Waldron, F.W., 1901. On the appearance and disappearance of a mud island at Walfish Bay. *Transactions of the South African Philosophical Society* 11(1), 185-194.
- Ward, L.G., 1985. The influence of wind waves and tidal currents on sediment resuspension in Middle Chesapeake Bay. *Geo-Marine Letters* 5, 1-75.
- Warwick, R.M., Goss-Custard, J.D., Kirby, R., George, C.L., Pope, N.D., Rowden, A.A., 1991. Static and dynamic environmental factors determining the community structure of estuarine macrobenthos in SW Britain: why is the Severn estuary different? *J. Appl. Ecol.* 28: 329-345.
- Weeks, S.J., Currie, B., Bakun, A., Peard, K.R., 2004. Hydrogen sulphide eruptions in the Atlantic Ocean off southern Africa: implications of a new view based on SeaWiFS satellite imagery. *Deep-Sea Research I* 51,153-172.
- Wickens, P., 1995. Namibian sealing debacle, an environmental and managerial disaster. *Afr. Wildl.* 49: 6-11.
- Williams, A.J., Dyer, B.M., Randall, R.M., Komen, J., 1990. Killer whales *Orcinus orca* and seabirds: "Play", predation and association. *Marine Ornithology* 18: 37-41.
- Witherington, B.E., Martin R.E., 2000. Understanding, assessing and resolving light-pollution problems on sea turtle nesting beaches, 2nd ed. Rev. Florida Marine Research Institute Technical Report TR-2.73 pp.
- Witt, M.J., Bonguno, E.A., Broderick, A.C., Coyne, M.S., Formia, A., Gibudi, A., Mounquengui, G.A., Moussounda, C., Safou, M., Nougessono, S., Parnell, R.J., Sounguet, G.-P., Verhage, S., Godley, B.J., 2011. Tracking leatherback turtles from the world's largest rookery: assessing threats across the South Atlantic. *Proceedings of the Royal Society B*. doi:10.1098/rspb.2010.2467
- Witzell, W.N., 1999. Distribution and relative abundance of sea turtles caught incidentally by the U.S. pelagic longline fleet in the western North Atlantic Ocean, 1992-1995. *Fisheries Bulletin* 97: 200-211.

- WSP Group Africa (Pty) Ltd, 2014a. Beach and Shoreline Dynamics Specialist Study and Impact Assessment. Prepared for Aurecon South Africa and SLR Namibia, on behalf of Rio Tinto - Rossing Uranium. October 2014. 28pp.
- WSP Group Africa (Pty) Ltd, 2014b. WSP Input to EIA Scoping Report for Rössing Uranium Desalination Plant near Swakopmund: Brine Discharge Specialist Study. Prepared for Aurecon South Africa and SLR Namibia, on behalf of Rio Tinto - Rossing Uranium. October 2014. 40pp.
- Yates, M.G., Goss-Custard, J.D., McGrorty, S.M., Lakhani, Dit Durrell, S.E.A., Levit, Clarke, R.T., Rispin, W.E., Moy, I., Yates, T., Plant, R.A., Frost, A.J., 1993. Sediment characteristics, invertebrate densities and shorebird densities on the inner banks of the Wash. *J. Appl. Ecol.* **30**: 599- 614.
- Zajac RN, Lewis RS, Poppe LJ, Twichell DC, Vozarik J, Digiacommo-Cohen ML (2000) Relationships among sea-floor structure and benthic communities in Long Island Sound at regional and benthoscape scales. *J. Coast. Res.* **16**: 627- 640.
- Zeybrandt, F., Barnes, J.I., 2001. Economic characteristics of demand in Namibia's marine recreational shore fishery. *South African Journal of Marine Science* **23**, 145-156.
- Zoutendyk, P., 1992. Turbid water in the Elizabeth Bay region: A review of the relevant literature. CSIR Report EMAS-I 92004.
- Zoutendyk, P., 1995. Turbid water literature review: a supplement to the 1992 Elizabeth Bay Study.

Appendix A:

1. Potential Effects of Blasting
2. Seawater Chlorination Chemistry and Associated Potential Impacts
3. Summary of Chemistry and Environmental Fate of Certain Desalination Plant Cleaning Chemicals and Biocides

A.1 Potential Effects of Blasting

The laying of the intake and discharge pipeline will require blasting. Keevin & Hempen (1997) and Lewis (1996) provide information on blast-effects on a variety of shallow water (<10 m) organisms. Below follows a summary of these effects focussing on the marine macrophytic algae, major invertebrate macrofaunal taxa, fish, turtles and marine mammals that may occur in the blast area off the desalination plant site.

Macrophytes

Smith (1996) measured blast effects on three species of algae, and found that both physical and physiological damage can occur within 10.5 m of a 2 kg explosive charge. Mortality (=biomass loss) was limited to within 8.5 m whilst depressions in photosynthetic rates post-blast occurred at all distances observed: 2.5 m - 10.5 m from the blast. This indicates that any disruptions to algal beds through blasting would be limited to the immediate vicinity of the charges.

Invertebrates

Due to the lack of gas bodies, marine invertebrates appear to be relatively immune to blast effects in terms of obvious injury or mortalities. Keevin & Hempen (1997) reported that oysters (*Ostrea virginica*) exposed to a 136.1 kg charge of TNT (high explosive) in open water had 100% survival at distances ranging from 7.6 - 122 m from the blast. Crabs (*Callinectes sapidus*) also showed high survival rates when exposed to a 90.7 kg open water charge, with mortalities ranging from 28% at a distance of 15.2 m from the blast, to 11% at a distance of 75 m. At 110 m from the charge, crab mortalities were zero. In a study by CSIR (1997) in Saldanha Bay, mud prawns (*Upogebia capensis*) suspended in perforated, thin walled plastic bags at 0.5 m, 30 m, 70 m and 120 m from six short interval (millisecond) 22.5 kg high explosive blasts in stemmed shot holes, showed no mortalities, and were actively swimming immediately after the blasts. In contrast, Keevin & Hempen (1997) reported 55% mortality in crabs exposed within 38 m - 15 m to a 13.6 kg blast in open water. Sublethal injuries in crabs, including carapace rupture, have been observed within metres to similarly moderately sized blasts (Keevin & Hempen 1997). This suggests that the blast-effects on invertebrates are likely to remain confined to the construction area and minimal far-field effects are likely to occur. Consequently deleterious impacts of underwater blasting on the invertebrate macrofauna in the vicinity of the pipeline are considered to be insignificant should they occur.

Fish

The swim bladder in fish is the organ most frequently damaged by shock (pressure) waves generated by underwater explosions (Lewis 1996, and authors cited therein). Post-mortem examinations of fish killed by underwater explosions generally show traumatic rupture of swim bladders and associated damage to adjacent organs including kidney, liver and spleen (Keevin & Hempen 1997). Further evidence of the role of the swim bladder in blast trauma is offered by the different apparent sensitivities to underwater explosions of physoclistous and physostomus fish species. The former have their swim bladder attached to the circulatory system and it consequently responds slowly to pressure changes, whereas the latter have the swim bladder ducted to the oesophagus with a relatively rapid pressure equalization response. Consequently physoclistic fish species, such as white bass (*Morone chrysops*) appear to be more sensitive to blasts than physostomus species such as trout (*Salmo* sp). Further factors moderating susceptibility to mortality and injury due to blast effects include body shape and overall size. In general thick bodied cylindrical fish, e.g. *Sphyræna* spp. (barracuda), are less susceptible to injury than more laterally compressed species such as Sparidae (Fitch & Young 1948). Furthermore, Yelverton *et al.* (1975) found that higher shock wave intensity was required to kill larger than smaller fish of the same species.

Fish species that do not possess swim bladders (e.g. sharks and rays, some bony fish such as sea chub *Girella* spp, scorpion fish *Scorpaena* and *Scorpaenichthys* sp., and soles such as *Trinectes* sp.) appear to be largely immune to underwater explosions. For example, Goertner *et al.* (1994) found that *Trinectes* were not killed beyond a distance of 1 m from an open water charge of 4.5 kg of the high explosive pentolite.

Hill (1978) has developed equations predicting lethal ranges and safe distances for fish exposed to open water explosions. Input information for these includes:

- Typical size (weight) of the fish species likely to be exposed to the charges
- Depth of the target fish in the water column
- Depth of the detonation, and
- Weight of the charge.

Keevin & Hempen (1997) provide nomograms based on Hill's (1978) equations for estimating ranges from these variables. Following Hill's (1978) recommendations ranges calculated from the nomograms should be doubled to account for possible energy focusing effects of shallow water. Given the fact that surf zone and nearshore species along the Namaqualand coastline are widely distributed, the probability of the blasting programme having a measurable effect at the population level on fish in the study area is judged to be unlikely and therefore of low impact.

Based on exposures of anchovy eggs and larvae to a small charge size of 50 g TNT, Kostyuchenko (1973) concluded that fish eggs and pre-air bladder inflation fish larvae suffer pathological injury from underwater explosions, but effect ranges appear to be relatively small (< 20 m). The 'Guidelines for the use of explosives in Canadian Fisheries waters' (Wright, cited in Keevin & Hempen 1997) utilise a wider range of data and define a peak particle velocity of 13 mm/s as the critical threshold. These data allow the calculation of setback distances for fish spawning areas according to the regression equation:

$$\text{Setback distance (m)} = 1.806 (\text{charge wt in kg}) + 34.61$$

It is assumed that fish eggs and larvae will be widely distributed along the Namaqualand coastline. Given the small area in which effects would possibly be generated, the probability of the proposed blasting programme having a measurable effect on fish eggs and larvae on a population level in the study area is unlikely.

Birds

Information on the effects of underwater blasting on swimming and diving birds is limited to experiments on ducks (Lewis 1996). Mortality occurred primarily within the immediate vicinity (< 10 m) of the blast, as a result of extensive pulmonary haemorrhaging and ruptured livers, kidneys, airsacs and eardrums. Birds beyond 20 m from the blast were largely uninjured. Lewis (1996) presents underwater blast criteria for birds on and beneath the water surface, from which safe and lethal ranges can be estimated.

In the case of underwater explosions, shock waves above the water surface are considered highly unlikely (O'Keeffe & Young 1984), and impacts on shore-birds can therefore be expected to be insignificant. Blasting on the shoreline, however, are likely to result in flight responses in nesting birds (Wambach *et al.* 2001), and resting or feeding flocks on the shore.

Turtles

A number of studies have demonstrated that sea turtles are killed and injured by underwater explosions (Duronslet *et al.* 1986; Gitschlag 1990; Gitschlag & Herozeg 1994; Gitschlag & Renaud 1989; Klima *et al.* 1988; O'Keeffe & Young 1984). Experiments undertaken to document the effects of underwater explosions on sea turtles, found that animals placed at intervals between 200-900 m from an explosive removal of an oil platform suffered averted cloaca and vasodilation, and in extreme cases lost consciousness, and if left in the water may have drowned. Carapace fractures in Loggerhead turtles which surfaced within minutes of a detonation have also been reported, as have extensive internal damage, particularly to the lungs.

Young (1991) developed the following equation to estimate sea turtle safe ranges, but as there has been no study establishing the relationship between underwater explosive pressures and mortality, this should be used for preliminary planning purposes only.

$$R = 222 W^{1/3}$$

Where R = range in m and W = charge weight in kg.

There are no data on non-lethal damage from underwater explosions or delayed mortality, both of which may have a greater impact on sea turtle populations than immediate death from explosions.

Although occurring in the study area, turtles are infrequent visitors in the shallow nearshore regions. It is recommended that the area around the blasting area be searched before blasting commences, and to postpone blasting if a sea turtle is spotted. Given the small area in which effects would possibly be generated, the probability of the proposed blasting programme having a measurable effect on turtles in the study area is unlikely if the above recommendation is adhered to.

Marine Mammals

Similar to fish, injuries to mammals generated by underwater explosions are primarily trauma of various levels to organs containing gas, such as lungs, ears, and the intestinal tract. Empirical evidence on seals suggests that close proximity to charges can result in mortality, with observations of seals being killed by an 11.4 kg dynamite charge exploded 23 m away (Hanson 1954, cited in Keevin & Hempen 1997). Empirical observations on blast effects on other mammals have allowed the formulation of quantitative relationships between explosive charge size and safe distances. Keevin & Hempen (1997) provide such relationships derived from Young (1991) and Hill (1978). Using three input variables, namely depth of the target animal, depth of detonation and weight of the charge, the safe distances from the predicted maximum charges can be estimated in terms of seal mortality and sub-lethal injury. Note that seals outside of the lethal range but within zero effect range limit may suffer blast injuries such as lung haemorrhaging or ear drum rupture (Hill 1978). However, animals are expected to recover unaided; *i.e.* no human intervention should be required.

Given the relatively small lethal range and the generally low numbers of seals in the study area relative to the overall population size any population level mortality effects, or injuries that may be caused are judged to be insignificant.

Although occurring in the study area, whales and dolphins are infrequent visitors in the shallow nearshore regions, being more common further offshore. Because of their large sizes the risk of pathological injuries that may be caused by the proposed blasting appears to be constrained because of limited effect ranges. Young (1991) gives the following safe ranges for dolphins and whales, the equations indicating a reduction in sensitivity to underwater explosions with increasing size:

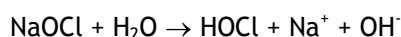
Juvenile dolphin	$R = 576 W^{0.28}$
Dolphin	$R = 434 W^{0.28}$
6m Whale	$R = 327 W^{0.28}$

Where R = range in m and W = charge weight in kg.

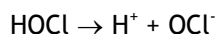
Due to the limited effect ranges and the distributions of whales and dolphins in the region any effects of the proposed blasting programme at the respective population levels are considered to be insignificant. As specified under South African environmental laws, disturbance of whales should be avoided. If whales are present in the blast area, disturbance cannot be ruled out. Consequently mitigation of the possible disturbance effect is required. It is recommended to visually search the area around the blasting area before blasting commences and to postpone the blasting should a whale be spotted.

A.2 Seawater Chlorine Chemistry and Associated Potential Impacts

The chemistry associated with seawater chlorination when using chlorine-based products is complex and only a few of the reactions are given below, summarised from ANZECC (2000), Lattemann & Höpner (2003) and UNEP (2008). Chlorine does not persist for extended periods in water but is very reactive. Its by-products, however, can persist for longer. The addition of sodium hypochlorite to seawater results in the formation of hypochlorous acid:

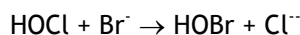


Hypochlorous acid is a weak acid, and will undergo partial dissociation as follows:



In waters of pH between 6 and 9, both hypochlorous acid and hypochlorite ions will be present; the proportion of each species depending on the pH and temperature of the water. Hypochlorous acid is significantly more effective as a biocide than the hypochlorite ion.

In the presence of bromide (Br^-), which like chloride is a natural component of seawater (average bromide concentration in seawater is 67 mg/ℓ), chlorine instantaneously oxidises bromide to form hypobromous acid and hypobromite (HOBr):



Hypobromous acid is also an effective biocide. It is worth noting that, for a given pH value, the proportion of hypobromous acid relative to hypobromite is significantly greater than the corresponding values for the hypochlorous acid - hypochlorite system. Thus, for example, at pH 8 (the pH of seawater), hypobromous acid represents 83% of the bromine species present, compared with hypochlorous acid at 28%. Hypobromous acid can also disproportionate into bromide and bromated, which is accelerated by sunlight.

In natural waters, chlorine can undergo a range of reactions in addition to those discussed above, leading to the formation of a range of by-products. The reaction of chlorine with organic constituents in aqueous solution can be grouped into several types:

(a) Oxidation,

where chlorine is reduced to chloride ion, e.g. $\text{RCHO} + \text{HOCl} \rightarrow \text{RCOOH} + \text{H}^+ + \text{Cl}^-$

(b) Addition,

to unsaturated double bonds, e.g. $\text{RC} = \text{CR}' + \text{HOCl} \rightarrow \text{RCOHCCIR}'$

(c) Substitution,

to form N-chlorinated compounds, e.g. $\text{RNH}_2 + \text{HOCl} \rightarrow \text{RNHCl} + \text{H}_2\text{O}$

or C-chlorinated compounds, e.g. $\text{RCOCH}_3 + 3\text{HOCl} \rightarrow \text{RCOOH} + \text{CHCl}_3 + 2\text{H}_2\text{O}$

Chlorine substitution reactions can lead to the formation of organohalogen compounds, such as chloroform, and, where HOBr is present, mixed halogenated and brominated organic compounds. The number of by-products can hardly be determined due to many possible side reactions. A major component, however, are the trihalomethanes (THMs) such as bromoform. Concentrations of other halogenated organics are considerably lower and usually in the nanogram per liter range. Substances of anthropogenic origin in coastal waters, especially mineral oil or diesel fuels, may give rise to compounds like chlorophenols (some of which can taint fish flesh at concentrations as low as 0.001 mg/ℓ (DWAf 1995)) or chlorobenzenes. However, THMs such as bromoform account for most of the compounds.

A number of other source water characteristics are likely to have an impact on the concentrations of organic by-products present in brine water discharges: natural organic matter in water is the major precursor of halogenated organic by-products, and hence the organic content of the source water (often measured as total organic carbon, TOC) may affect the concentration of by-products formed. In general, the higher the organic content of the source water, the higher the potential for by-product formation. The ammonia concentration is likely to affect the extent of by-product formation, through reaction with chlorine to form chloramines. Although seawater generally contains low concentrations of ammonia than freshwater, under certain conditions (dependent on chlorine dose: ammonia nitrogen concentration) it can compete with bromide for the available chlorine to form monochloramine. In addition, hypobromous acid can react with ammonia to form bromamines. Although the sequence of reactions is complex, it is likely that the reaction of either hypochlorous or hypobromous acid with ammonia to form halamines will reduce organic by-product

formation during the chlorination of seawater. Chlorine can also react with nitrogen-containing organic compounds, such as amino acids to form organic chloramines. The pH of the incoming feed water could also affect the nature of the by-products formed. In general, while variations in pH are likely to affect the concentrations of individual by-products, the overall quantity formed is likely to remain relatively constant. Little is known about the biocidal properties of these compounds.

Paradoxically, chlorine chemistry thus establishes that no free chlorine is found in chlorinated seawater where bromide oxidation is instantaneous and quantitative. However, the chlorinated compounds, which constitute the combined chlorine, are far more persistent than the free chlorine. After seawater chlorination, the sum of free chlorine and combined chlorine is referred to as total residual chlorine (TRC).

Marine organisms are extremely sensitive to residual chlorine, making it a prime choice as a biocide to prevent the fouling of marine water intakes. Many of the chlorinated and halogenated by-products that are formed during seawater chlorination (see above) are also carcinogenic or otherwise harmful to aquatic life (Einav *et al.* 2002, Lattemann & Höpner 2003). Values listed in the South African Marine Water Quality Guideline (DWA 1995) show that 1500 µg/ℓ is lethal to some phytoplankton species, 820 µg/ℓ induced 50% mortality for a copepod and 50% mortality rates are observed for some fish and crustacean species at values exceeding 100 µg/ℓ (see also ANZECC 2000). The lowest values at which lethal effects are reported are 10 - 180 µg/ℓ for the larvae of a rotifer, followed by 23 µg/ℓ for oyster larvae (*Crassostrea virginica*). Sublethal effects include valve closure of mussels at values <300 µg/ℓ and inhibition of fertilisation of some urchins, echinuroids, and annelids at 50 µg/ℓ. Eppley *et al.* (1976) showed irreversible reductions in phytoplankton production, but no change in either plankton biomass or species structure at chlorine concentrations greater than 10 µg/ℓ. Bolsch & Hallegraeff (1993) showed that chlorine at 50 µg/ℓ decreased germination rates in the dinoflagellate *Gymnodinium catenatum* by 50% whereas there was no discernable effect at 10 µg/ℓ. This indicated that particularly the larval stages of some species may be vulnerable to chlorine pollution. The minimum impact concentrations reported in the South African Water Quality Guidelines are in the range 2 to 20 µg/ℓ at which fertilisation success in echinoderm (e.g. sea urchin) eggs is reduced by approximately 50% after 5 minute exposures.

A.3 Environmental Fate of Cleaning Chemicals used in the CIP Process

The membranes in the desalination plant will need periodical cleaning (CIP = Cleaning in Place) to remove any biofouling. The currently suggested cleaning interval for the proposed desalination project is three times per year. Typical cleaning chemicals include weak acids, detergents, oxidants, complexing agents and/or non-oxidising biocides for membrane disinfection. These chemicals are usually generic types or special brands recommended by the membrane manufacturers. The exact list of chemicals used will only be known once the desalination plant operator has been appointed. Common cleaning chemicals, however, include Sulphuric acid, Ethylenediaminetetra-acetic acid (EDTA), Sodium tripolyphosphate (STPP), and Trisodium phosphate (TSP), and Dibromonitripropionamide (DBNPA) as non-oxidising biocide. Below follows a short summary of the environmental fates and effects of these chemicals.

Sulphuric acid (H₂SO₄) is used for pH adjustment in the desalination process to reduce the pH for the acid-wash cycle. It is a strong mineral acid that dissociates readily in water to sulphate ions and hydrated protons, and is totally miscible with water. At environmentally relevant concentrations, sulphuric acid is practically totally dissociated, sulphate is at natural concentrations and any possible effects are due to acidification. This total ionisation also implies that sulphuric acid, itself, will not adsorb on particulate matters or surfaces and will not accumulate in living tissues (<http://www.chem.unep.ch/irptc/sids/oecdsids/7664939.pdf>). Sulphuric acid can be acutely toxic to aquatic life *via* reduction of water pH. Most aquatic species do not tolerate pH lower than 5.5 for any extended period. No guideline values are available for this substance but *No Observed Effect Concentration* (NOEC) values were developed from chronic toxicity tests on freshwater organisms and range from 0.058 mg/ℓ for fish populations to 0.13 mg/ℓ for phytoplankton and zooplankton populations, respectively (<http://www.chem.unep.ch/irptc/sids/oecdsids/7664939.pdf>). As seawater is highly buffered, the limited sulphuric acid

discharges are not expected to have significant impacts in the marine environment. The pH of the effluent is predicted to be between 7.3 and 8.2.

EDTA is an aminopolycarboxylic salt that is used as a chelating agent to bind or capture trace amounts of iron, copper, manganese, calcium and other metals. In water treatment systems, EDTA is used to control water hardness and scale-forming calcium and magnesium ions to prevent scale formation. Because of the ubiquitous presence of metal ions, it has to be assumed that EDTA is always emitted as a metal complex, although it cannot be predicted which metal will be bound. EDTA will biodegrade very slowly under ambient environmental conditions but does photodegrade. EDTA is not expected to bioaccumulate in aquatic organisms, adsorb to suspended solids or sediments or volatilize from water surfaces (European Union Risk Assessment Report 2004). Toxicity tests on aquatic organisms have shown that adverse effects occur only at higher concentrations (the lowest concentrations at which an adverse effect was recorded is 22 mg/ℓ) (European Union Risk Assessment Report 2004). On the other hand, if trace elements like Fe, Co, Mn, and Zn are low in the natural environment, an increased availability of essential nutrients caused by the complexing agent EDTA is able to stimulate algal growth. Heavy metal ions in the water are complexed by free EDTA, and a comparison of the toxicity of those compared to the respective uncomplexed metals and free EDTA have shown a reduction in toxicity by a factor of 17 to 17000 (Sorvari & Sillanpää 1996). Experiments (albeit with significantly higher trace metal concentrations than are typically observed in the environment) indicate that EDTA decreases the accumulation of metals such as Cd, Pb and Cu, however the absorption of Hg by mussels is seemingly promoted through complexation with EDTA (Gutiérrez-Galindo 1981, as cited in the European Union Risk Assessment Report 2004). Potential promotion of the accumulation of metals in sediments is unlikely to be a concern as in high concentrations EDTA prevents the adsorption of heavy metals onto sediments and even can remobilise metals from highly loaded sediments (European Union Risk Assessment Report 2004). Within the framework of marine risk assessment, the European Union has published a risk assessment report in which a *Predicted No Effect Concentration* (PNEC) of 0.64 mg/ℓ was calculated (European Union Risk Assessment Report 2004). The EDTA concentration expected in the brine is 0.013 mg/ℓ and lies thus under the PNEC value.

Sodium tripolyphosphate (STPP, Na₅P₃O₁₀) is the sodium salt of triphosphoric acid. It is a typical ingredient of household cleaning products, and is thus commonly present in domestic waste-waters. STPP is an inorganic substance that when in contact with water (waste-water or natural aquatic environment) is progressively hydrolysed by biochemical activity, finally to orthophosphate. Acute aquatic ecotoxicity studies have shown that STPP has a very low toxicity to aquatic organisms (all EC/LC₅₀ are above 100 mg/ℓ) and is thus not considered as environmental risk (HERA 2003). The final hydrolysis product of STPP, orthophosphate, however, can lead to eutrophication of surface waters due to nutrient enrichment. However, phosphate as a nutrient is not limiting in marine environments unless there are significant inputs of nitrogen (nitrates, ammonia), which is the limiting nutrient in the marine environment. Depending on the presence of cationic ions, STPP can, in addition to the hydrolysis into orthophosphate, precipitate in the form of insoluble calcium, magnesium or other metal complex species (HERA 2003).

Trisodium phosphate (TSP) (Na₃PO₄) is a highly water-soluble cleaning agent. When dissolved in water it has an alkaline pH. The phosphate can act as a plant nutrient, and can thus increase algal growth, however, as noted above, phosphate as a nutrient is not limiting in marine environments unless there are significant inputs of nitrogen.

The non-oxidising biocide DBNPA, which could potentially be added during the RO cleaning process, or may be an alternative to chlorine based biocides, has extremely fast antimicrobial action and rapid degradation to relatively non-toxic end products (US EPA 1994). The ultimate degradation products formed from both chemical and biodegradation processes of DBNPA include ammonia, carbon dioxide, and bromide ions. Degradation end products (e.g. ammonia) will seemingly not be problematic in the marine environment, however, it is the specific biocidal action of residual DBNPA in the effluent streams that is the major concern. The dominant degradation pathway of DBNPA involves reaction with nucleophilic substances or organic material invariably found in water. Additional degradation reactions include hydrolysis, reaction with soil, and breakdown through light (US EPA 1994). The uncatalyzed hydrolysis of DBNPA proceeds via decarboxylation to the generation of an array of degradation products. These degradates include dibromoacetoneitrile, dibromoacetamide, dibromoacetic acid, monobromoacetamide, monobromonitrilo-propionamide,

monobromoacetic acid, cyanoacetic acid, cyanoacetamide, oxoacetic acid, oxalic acid, and malonic acid. The rate of hydrolysis is a function of pH and temperature, and increasing either or both pH and temperature will increase the decomposition rate. For instance, at pH 5 the half-life of DBNPA is 67 days as opposed to 63 hours at pH 7 and 73 minutes at pH 9. In natural waters (seawater has a pH of 8), DBNPA hydrolyses rapidly (half life < five hours) into the above mentioned degradates which continue to degrade rapidly by aerobic and anaerobic aquatic metabolism (US EPA 1994). Although the hydrolysis and aquatic photolysis rate is rapid under aquatic conditions, the primary degradation pathway is through aerobic and anaerobic metabolism. In aerobic and anaerobic aquatic metabolism studies, DBNPA degraded with a half-life of <four hours, with a further rapid decrease of the degradate concentrations (US EPA 1994). Exposure to sunlight is a further factor increasing the rate of decomposition which results in the formation of inorganic bromide ion. For example, the half-life of DBNPA was reported to be approximately 7 days when exposed to sunlight even at a pH of 4 (Dow Chemicals, Fact Sheet No. 253-01464-06/18/02). Aquatic toxicological studies have shown that DBNPA appears to be moderately toxic to estuarine fish and shrimp, highly toxic to estuarine mysids and very highly toxic to estuarine shellfish and larvae. It must be noted though that, due to the fast degradation of DBNPA, toxic effects are generally acute occurring within 24 hours of exposure, and chronic effects will not occur. Due to the rapid degradation of DBNPA in natural waters, some risk assessment studies have concluded that the use of DBNPA in cooling systems (once through and recirculating systems) does not pose an unacceptable risk to the environment (Klaine et al., 1996). Mitigation measures to ensure low residuals of DBNPA in any discharge to the marine environment include appropriate design of the brine basin so as to ensure greater and sufficient dilution of the DBNPA residuals in the effluent stream before discharge.

A.4 References

- ANZECC, (2000). Australian and New Zealand guidelines for fresh and marine water quality. Volume 2, Aquatic ecosystems. National water quality management strategy; no.4. Australian and New Zealand Environment and Conservation Council, Agriculture and Resource Management Council of Australia and New Zealand, Canberra Australia. ISBN 0 642 19562 5 (www.deh.gov.au/water/quality/nwqms/introduction/).
- Bolsch, C.J. and G.M. Hallegraeff (1993). Chemical and physical treatment options to kill toxic dinoflagellate cysts in ship's ballast water. *Journal of Marine Environmental Engineering* 1, 23-29.
- CSIR, 1997. Saldanha Bay general cargo quay dredging: Effects of underwater blasting on benthic invertebrates and fish. CSIR Report ENV/C 97055. 6pp.
- CSIR (1997). Saldanha Bay general cargo quay dredging: Effects of underwater blasting on benthic invertebrates and fish. CSIR Report ENV/C 97055. 6pp.
- Department of Water Affairs and Forestry (DWAF), 1995. South African water quality guidelines for coastal marine waters. Volume 1. Natural Environment. Volume 2. Recreation. Volume 3. Industrial use. Volume 4. Mariculture. Pretoria.
- Duronslet, M.J., Caillouet, C.W., Manzella, S., Indelicato, K.W., Fontaine, C.T., Revera, D.B., Williams, T. and D. Boss (1986). The effects of an underwater explosion on the sea turtles *Lepidochelys kempi* and *Caretta caretta* with observations of effects on other marine organisms. Unpublished report submitted to National Marine Fisheries Service Biological Laboratory, Galveston, Texas.
- Einav, R., Harussi, K. and D. Perry (2002). The footprint of the desalination processes on the environment. *Desalination*, 152, 141-154.
- Eppley, R.W., Renger E.H. and P.M. Williams (1976). Chlorine reactions with seawater constituents and inhibition of photosynthesis of natural marine phytoplankton. *Estuarine and Coastal Marine Science*, 7, 291-301.
- European Union Risk Assessment Report (2004). Edetic Acid (EDTA). CAS No: 60-00-4. EINECS No: 200-449-4. Risk Assessment Vol. 49. Final Report, 2004. http://ecb.jrc.it/documents/Existing-Chemicals/RISK_ASSESSMENT/REPORT/edtareport061.pdf
- Fitch, J.E. and P.H. Young (1948). Use and effect of explosives in California waters. *California Fish and Game*, 34, 53-70.
- Gitschlag, G.R., (1990). Sea turtle monitoring at offshore oil and gas platforms. In: Richardson, T.H., Richardson, J.I. & M. Donnelly (Eds). *Proceedings of the 10th Annual Workshop on Sea Turtle Biology and Conservation*. NOAA Technical Memorandum NMFS-SEFC-278. pp. 223-246.
- Gitschlag, G.R. and B.A. Herzog (1994). Sea turtle observations at explosive removals of energy structures. *Marine Fisheries Review*, 56, 1-8.
- Gitschlag, G.R. and M. Renaud (1989). Sea turtles and the explosive removal of offshore oil and gas structures. In: Eckert, S.A., Eckert, K.L. & T.H. Richardson (Eds). *Proceedings of the 9th Annual Workshop on Sea Turtle Conservation and Biology*, NOAA Technical Memorandum NMFS-SEFC-232, pp. 67-68.
- Goertner, J.F., Wiley, M.L., Young, G.A. and W.W. McDonald (1994). Effects of underwater explosions on fish without swimbladders. Technical Report NSWC TR 88-114. Naval Surface Warfare Center, Dahlgren Division, White Oak Detachment Silver Spring, MD.

- Gutiérrez-Galindo E.A. (1981). Effect de l'EDTA sur l'accumulation et l'élimination du Mercure par la moule *Mytilus edulis*. *Chemosphere*, **10**, 971-976.
- HERA (2003). Human & Environmental Risk Assessment on ingredients of European household cleaning products. Sodium Tripolyphosphate (STPP) CAS: 7758-29-4. Draft June 2003. (<http://www.heraproject.com>).
- Hill, S.H., (1978). A guide to the effects of underwater shock waves on Arctic marine mammals and fish. *Pacific Marine Science Report* 78-26. Institute of Ocean Sciences, Patricia Bay, Sidney, B.C. 50 p.
- Keevin, T.M. and G.L. Hempen (1997). The environmental effects of underwater explosions with methods to mitigate impacts. U.S. Army Corps of Engineers St. Louis District 1222 Spruce Street St. Louis, Missouri 63103-2833.
- Klaine, S. J., Cobb, G. P., Dickerson, R. L., Dixon, K. R., Kendall, R. J., Smith, E. E. and Solomon, K. (1996). An ecological risk assessment for the use of the biocide, Dibromonitripropionamide (DBNPA), in industrial cooling systems. *Environmental Toxicology and Chemistry*, **15**, 21-30.
- Klima, E.F., Gitschlag, G.R. and M.L. Renaud (1988). Impacts of the explosive removal of offshore petroleum platforms on sea turtles and dolphins. *Marine Fisheries Review*, **50**, 33-42.
- Kostyuchenko, L.P., (1973). Effects of elastic waves generated in marine seismic prospecting on fish eggs in the Black Sea. *Hydrobiological Journal*, **9**, 45-48.
- Lattemann, S. and T. Höpner (2003). Seawater desalination: Impacts of brine and chemical discharge on the marine environment, Balaban Desalination Publications, Italy. pp. 142 + Appendices.
- Lewis, J.A., (1996). Effects of underwater explosions on life in the sea. Ship Structures and Materials Division, Aeronautical and Maritime Research Laboratory. Document # DSTO-GD-0080. 38pp.
- O'Keeffe, D.J. and G.A. Young (1984). Handbook on the environmental effects of underwater explosions (Technical Report NSWC TR 83-240). Silver Spring, MD: Naval Surface Weapons Centre.
- OECD (Organisation for Economic Co-operation and Development) (1997). Screening Information Data Set: (SIDS) Initial Assessment Profile: Sodium Dodecyl Sulfate. <http://www.chem.unep.ch/irptc/sids/OECD/SIDS/151213.htm>
- Smith, M., (1996). Effects of underwater explosions on aquatic vegetation, submerged and emergent. Report Submitted to the U.S. Army Corps of Engineers, Project #DACW43-96-P1584. Southern Illinois University, Edwardsville, IL.
- Sorvari, J. and M. Sillanpää (1996). Influence of metal complex formation on heavy metal and free EDTA and DTPA acute toxicity determined by *Daphnia magna*. *Chemosphere*, **33(6)**, 119-127.
- UNEP, (2008). Desalination Resource and Guidance Manual for Environmental Impact Assessments. United Nations Environment Programme, Regional Office for West Asia, Manama, and World Health Organization, Regional Office for the Eastern Mediterranean, Cairo.
- US EPA (1994) Reregistration Eligibility Decision (RED) - 2,2-dibromo-3-nitripropionamide (DBNPA) - List C (Case 3056), EPA 738-R-94-026, Office of Pesticide Programmes, Washington DC, 179pp.
- Wambach, D.A., Traxler, M.A. and K.W. Eakin (2001). Montana Department of Transportation - a fine feathered friend. In: Irwin, C.L., Garrett, P. & K.P. McDermott (Eds). *Proceedings of*

the 2001 International Conference on Ecology and Transportation, Center for Transportation and the Environment, North Carolina State University, Raleigh, NC: pp. 546-548. (Abstract).

Yelverton, J.T., Richmond, D.R., Hicks, W., Sanders, K. and E.R. Fletcher (1975). The relationship between fish size and their response to underwater blast. Topical Report DNA 3677T. Defense Nuclear Agency, Department of Defense, Washington, D.C.

Young, G.A., (1991). Concise methods for predicting the effects of underwater explosions on marine life. NAVSWC NO 91-220. Naval Surface Warfare Center. Silver Spring, Maryland.