Orange-Senqu River Basin

Orange-Senqu River Commission Secretariat
Governments of Botswana, Lesotho, Namibia and South Africa

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Estuary and Marine EFR assessment, Volume 3: Assessment of the role of freshwater inflows in the coastal marine ecosystem

Research project on environmental flow requirements of the Fish River and the Orange-Senqu River Mouth

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Report list

A list of the Technical Reports that form of this study is provided below. A diagram illustrating the linkages between the reports is also provided.

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22	Delineation of the Study Area – Resource Unit Report, Research project on environmental flow requirements of the Fish River and the Orange-Senqu River Mouth
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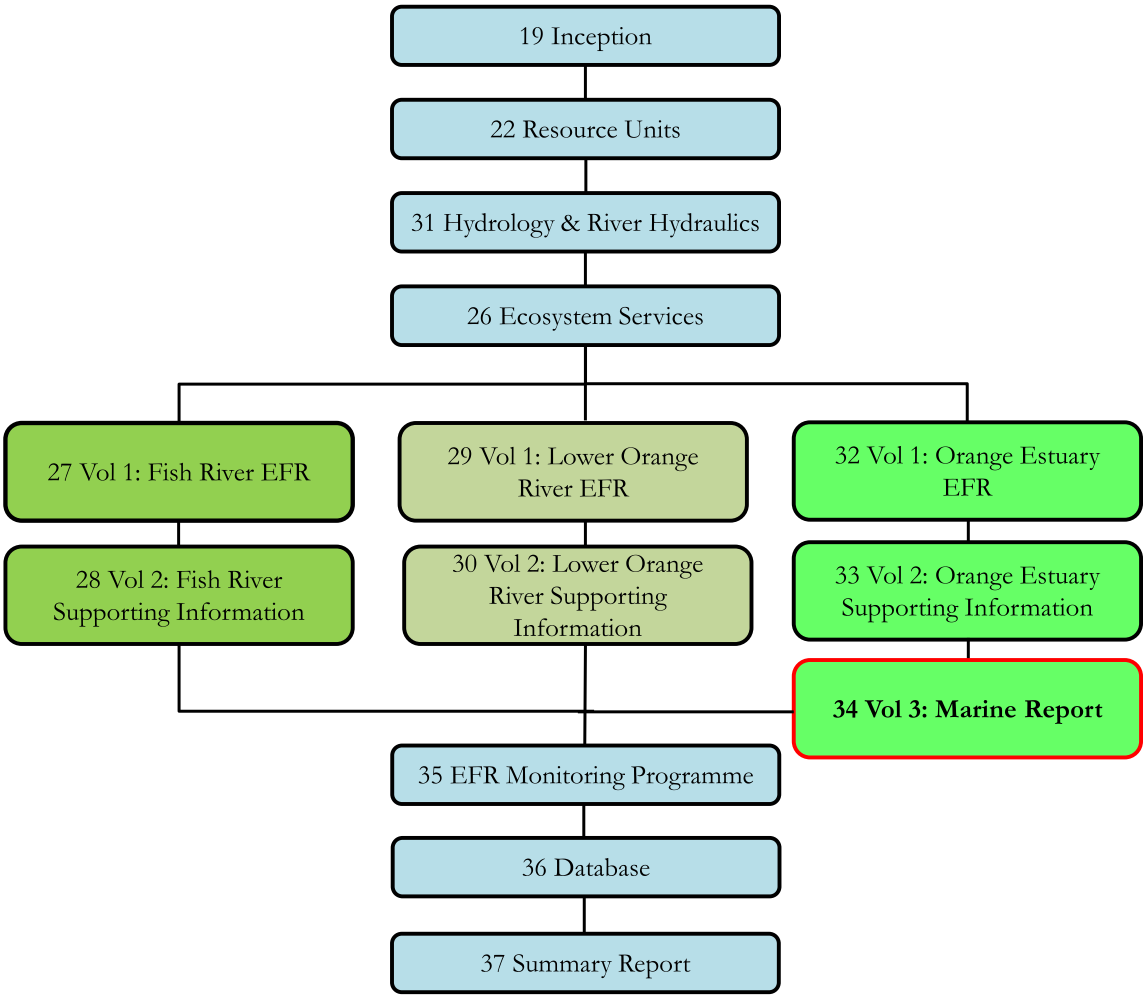
35	EFR monitoring programme, Research project on environmental flow requirements of the Fish River and the Orange-Senqu River Mouth
36	Database, Research project on environmental flow requirements of the Fish River and the Orange-Senqu River Mouth

Technical Report No	Report
37	Summary Report, Research project on environmental flow requirements of the Fish River and the Orange-Senqu River Mouth

Bold indicates current report.



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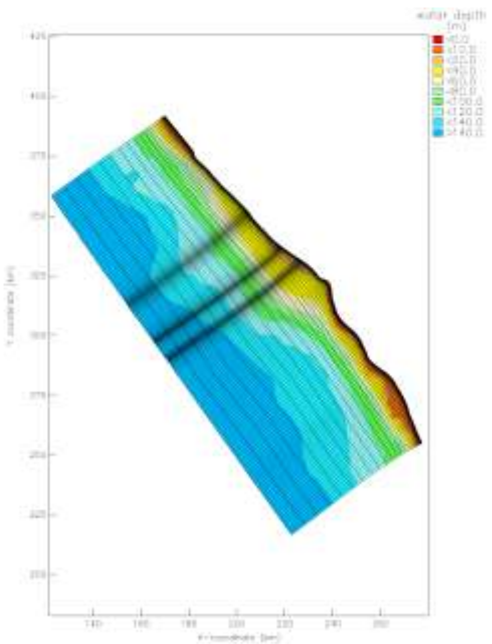
Executive summary

Introduction

This project proposes an assessment on the role of freshwater inflows and associated fluxes in the coastal marine ecosystems linked to the estuary and the potential effects of changes in the freshwater-related fluxes into these ecosystems. This will be done in order to recommend allowable changes in freshwater inflow into the marine environment within the constraints of maintaining or improving the present health status of the marine ecosystem and optimisation of the existing ecosystem services provided by the coastal ecosystem. This study will focus sediment and hydrodynamics, microalgae, invertebrates and fish using available catch data, remote sensing and numerical modelling.

Study area

For the purpose of this study the Orange River nearshore marine environment is defined as one degree north (~100 km) and one degree south (~100 km) of the Orange Estuary mouth, and offshore to the 200 m depth contour. The study area is situated near the centre of the Namaqua bioregion, a cool-temperate bioregion that extends from Sylvia Hill, north of Lüderitz in Namibia, to Cape Columbine in South Africa (Lombard et al., 2004). This bioregion is characterised by high levels of primary production both on the shore (algae) and offshore (phytoplankton). The computational grid and bathymetry used as basis for the nearshore marine flow and sediment dynamics model is provided below.



The Orange River drains into the southern section of the Benguela Current adjacent to the widest part of the continental shelf and at the southern boundary of the Lüderitz-Orange River Cone upwelling cell. This upwelling cell forms the boundary between the northern and southern Benguela. The surface currents in the vicinity of the river inflow are on average to the northeast. Since the freshwater inflow from the river lies on top of the seawater due to density differences, it can be assumed that on average the plume of the river will also flow in a north-eastern direction. The discharge from the estuary typically forms a plume of buoyant freshwater where it drains into the sea, the nature of which is shaped by the discharge volume and prevailing wind conditions (Shillington et al., 2006, Gan et al., 2009).

Nearshore marine ecological flow requirement method

No official method exists for determining the EFR of the nearshore marine environment in South Africa or Namibia. However, Van Ballegooyen et al. (2003) developed a comprehensive assessment framework for the evaluation of EFR of the nearshore marine environment based on international best practice. This study proposes to use a modified version of the propose framework to evaluate a range of freshwater flow scenarios by means of the following steps:

- Determine the legislative requirement for setting an EFR for the nearshore marine environment: Review the policies and legislation of relevance to the assessment and management of the freshwater requirements of the marine environment, including particular obligations under various treaties and international agreements.
- Definition of the ecosystem extent (biogeographic boundaries) and describe key properties: The boundaries of ecosystem of relevance to the assessment need to be defined based on the extent of the marine ecosystem potentially impacted by change of freshwater inflow (i.e. an appropriate definition of the ecological 'footprint'). A description of the ecosystem and its key components is also required to ensure a comprehensive EFR assessment and appropriate ecosystem management recommendations.
- Identify resource utilisation in ecosystem: The resource utilisation needs to be identified in order that, as a minimum, appropriate keystone/indicator species can be selected for the assessment of the freshwater requirements of the marine environment.
- Setting of environmental objectives: Based on the identified policy and legislative requirements, resource utilisation and characteristics of the ecosystem under consideration; specific management and environmental quality objectives need to be developed.
- Hydrological scenario assessment: Describe the changes in the past, present and future flow regime of the catchment to provide context to the assessment.
- Identification relevant abiotic components (habitat) and assess the response to flow modification: The critical abiotic components (e.g. salinity, nutrients, sediments, etc.) influencing the quality of the required habitats during the various life-cycle stages of the key biotic species need to be identified. The various abiotic (and biotic) components need to be integrated and/or aggregated, such that they are relevant to determining the biotic response.

- Describe the implications of flow alteration on selected biological components: This include the following:
    - selection of keystone or indicator species: Based on the management objectives, the defined ecosystem boundary and resource utilisation, keystone and/or indicator species need to be identified to minimise the complexity of the assessment, allow for the setting of clear and measurable environmental objectives, and ensure practical and effective management advice;
    - determination of life-cycle and habitat requirements: An analysis of the various life-cycle stages of the identified keystone or indicator species is required to identify the habitat requirements for the various life-cycle stages and consequently the abiotic (and biotic) components of relevance;
    - predict the possible responses, if any, to predicted change in abiotic components: Provide an analysis of the biotic responses to predicted abiotic change.
- Evaluation of socio-economic importance of marine aquatic ecosystems and resource uses: The outcomes of the scientific assessment of the potential impacts associated with changes in freshwater inflow into marine ecosystems need to be linked to the socio-economic implications of these changes as this is the primary basis upon which water resource allocations are likely to be made. Based on the outcome of this step, there may be modification of the recommended freshwater requirements for the nearshore marine ecosystems under consideration.
- Recommendation of Freshwater Requirements: The adequacy of the scientific assessment will be determined by whether or not there is sufficient understanding and/or measurements to translate management and environmental quality objectives into specific freshwater requirements or target values, based on usage of the nearshore marine environment as an existing or potential future resource. Typically this is only possible for a specific coastal and nearshore region once existing and potential future resource utilisation in the region of interest has been mapped and there is a reasonable understanding of the functioning of the ecosystems of relevance.

Marine environmental flow requirements

The influence of the Orange River on marine biota is likely to differ between nearshore and offshore, especially with regards to the surf zone which is often described as a closed system (McLachlan, 1981). The export of sediment, nutrients and detritus to the sea are undoubtedly important but it is sediment that shapes both near and offshore habitats. Nutrients from the river serve to stimulate phytoplankton and zooplankton production in the nearshore marine environment, ultimately benefitting the larval, juvenile and adult fish that depend on this food source. Floating debris offers refuge to juvenile fish whereas detritus may be broken down into useful nutrients, serve as a substrate for micro-flora and fauna or be consumed directly by detritivorous fish and invertebrates. Sediment export replenishes the nearshore habitats that are continuously eroded by oceanic currents and also provides a refuge for many fish by increasing turbidity. Turbidity, in turn, will serve to increase the catchability of many species, especially the larger individuals that move into the turbid environment in search of concentrated prey. The freshwater plume centred on the mouth of the estuary will provide cues for the migration of estuarine-dependent juvenile and adult fish into and out of the estuary. The strength of these cues will ultimately dictate how many individuals of these species recruit into the marine fisheries. River plumes also serve as a temperature refuge from cold upwelling for coastal migrants thereby maintaining connectivity between populations, habitats and biogeographical regions.

Implications of flow alteration on abiotic processes and primary production

Changes in the abiotic environment under the various proposed scenarios were assessed relative to reference condition. For all of the proposed development options relevant metrics were normalised relative to the magnitude of the metric under reference condition. When the metrics are normalised relative to reference condition, each of the metrics is reported as 100% under reference condition and typically at lower percentages for both present state and the remainder of the proposed development options. River inflows under reference conditions were typically a factor of 2 – 3 times greater than under the present state. The final step was to translate these percentage changes in nearshore marine habitats (under the various developments scenarios) into significance ratings. The mapping between percentage changes in the relevant metric and the significance of these changes under the various scenarios is summarised in the table below. The significance of the predicted changes relative to reference conditions typically range between 0 for reference conditions (i.e. minimal change) to -2 to -3 for present state conditions as well as expected conditions under proposed future scenarios (i.e. moderately significant to highly significant decreases).

Mapping used to translate changes (%) into significance ratings/indices of predicted change.

Significance rating	***Metric reported as a % of the magnitude of the metric under reference conditions***	***Comment***
3	175% to 200%	Highly significant increase
2	150% to 175%	Moderately significant increase
1	125% to 150%	Discernible increase
0	75% to 125%	Minimal change
-1	50% to 75%	Discernible decrease
-2	25% to 50%	Moderately significant decrease
-3	0% to 25%	Highly significant decrease

The changes in the freshwater, sediment and dissolved reactive silicates input into the marine environment were assessed based on the significance ratings described in table above. In terms of these significance ratings (with respect to inflows to the marine environment) it is not possible to discern between present state and proposed scenarios (Sc) 2 to 5. However Sc 6 and 7, at a rating of -3, compared to the rating of -2 for Sc 2 to 5, are significantly worse than the other proposed scenarios.

The above results (indices of change) characterise only the changes of freshwater and associated fluxes (sediments, nutrients, etc.) into the marine environment. This information can be used to assess potential changes in nearshore and offshore marine environments based on expert opinion and/or model simulated changes in water quality and/or sediment-related marine habitats. The model simulated changes in water quality and sediment-related marine habitats is reported in sections 9.1 (benthic habitats) and 9.2 (pelagic habitats).

Freshwater, dissolved reactive silicate (DRS), turbidity and sediment inflows to the nearshore marine environment under the various proposed future scenarios are provided below. The inflows for the various scenarios are expressed in terms of the significance ratings specified in the table preceding. All ratings are relative to reference conditions.

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume[1]	0	-2	-2	-2	-2	-2	-3	-3
Total discharge of sediments[1]								
Salinity	0	-2	-2	-2	-2	-2	-3	-3
DRS	0	-2	-2	-2	-2	-2	-3	-3
Turbidity	0	-2	-2	-2	-1/-2	-2	-3	-3
Sediments	0	-2	-2	-2	-2	-2	-3	-3

1 Annual average over a 66-year period.

Implications of flow alteration on selected biological components

Marine habitat and communities

Communities within marine habitats are largely ubiquitous throughout the southern African West Coast region, being particular only to substrate type or depth zone. Least variation is amongst seaweeds and invertebrates but only marginally more amongst fish. These biological communities consist of many hundreds of species, often displaying considerable temporal and spatial variability (even at small scales).

North of the Orange River mouth the shoreline is predominantly a sandy coast formed by the northward littoral transport of coarse marine sediments. Rocky intertidal habitats are represented only by occasional small rocky outcrops that host benthic communities strongly influenced by sediments. South of the river mouth, the coastline is dominated by rocky shores, with occasional short beaches interspersed between rocky headlands. The surf-zone immediately in front of the mouth is a highly reflective sandy shore. The deeper water marine ecosystems comprise primarily unconsolidated seabed sediments, much of it terrigenous, with subtidal reef habitats being limited to the shallow nearshore regions (<40 m). The pelagic ecosystem is also influenced by river flow. Reduced salinity and increased turbidity in the surface layers are associated with the influence of the Orange plume and are usually discernible in a 50 km radius from the mouth but may expand to 100 km or more during and after floods.

Historical changes in discharge volumes, shifts in seasonal flow variation and shifts in mouth closure events have most likely resulted in seasonal reversals of some abiotic drivers with potentially serious consequences for the estuary and ultimately the marine environment beyond (Taljaard, 2005). Such changes would almost certainly have influenced the community composition and abundance of fish and invertebrate communities surrounding the mouth of the estuary, and presumably must be having some impact on those species that rely on seasonal cues for entering or exiting the estuary.

Invertebrates

Invertebrates in the offshore soft sediments: An array of environmental factors and their complex interplay is ultimately responsible for the structure of benthic communities of which water depth and sediment composition are two of the major components of the physical environment determining invertebrate community structure off the Namibian and South African coastline. Diversity, distribution and abundance of invertebrate communities in the mixed terrigenous and marine deposits of the coastal zone are controlled by both the granulometric properties of the sediments and complex interactions between physical and biological factors at the sediment–water interface.

What must be kept in mind, however, is that marine communities in the Benguela are frequently exposed to naturally elevated suspended-sediment levels. They can thus be expected to have behavioural and physiological mechanisms and adaptations for coping with or capitalising on, this feature of their habitat, and are unlikely to be significantly affected by suspended sediment plumes generated by river discharges.

Pelagic invertebrate communities: The pelagic invertebrate communities of the Benguela are highly variable both spatially and temporally, their abundances largely being determined by the upwelling regime. Catchment flow is also highly variable and important but usually on a local scale adjacent to river mouths.

Along the southern African West Coast, where turbid water is a natural occurrence, inhibition of primary production in the near-shore environment is likely to be negligible. Suspended inorganic material can also enhance food availability to filter-feeding organisms by providing an extensive surface for the adsorption of dissolved organic material and microorganism colonization. However, the amount of organic matter ingested and assimilated generally increases with increasing particle concentration up to a threshold level above which the filter-feeding mechanism becomes overloaded, and filtration rate again declines in order to maintain assimilation rates and minimize energy loss. On the whole, the effect of suspended sediment loads on juvenile and adult invertebrates are usually beneficial, occasionally negative but at sub-lethal levels.

Rock lobster: West coast rock lobster *Jasus lalandii* sustain a large fishery and are an important invertebrate predator in kelp bed habitats in the region. Lobster are tolerant of high suspensed sediment loads but sedimentation or heavy siltation of nearshore reefs may reduce the carrying capacity of an otherwise suitable habitat, therefore potentially directly affecting rock-lobster populations, or reducing regional recruitment where sedimentation is widespread. This may consequently have important implications for the success of the commercial harvest of this resource in an area.

Declines in lobster catches off Namibia coincided with, or followed shortly after, Orange flood events in 1955, 1958, 1967, 1974, 1976 and 1988 Penney et al. (2008). Off South Africa, declines coincided with the 1938, 1944, 1955, 1958, 1967 and 1988 flood events. Even the 1948 flood appears to coincide with a minor dip in catches during the period of otherwise rapid expansion of this fishery between 1945 and 1951. Only the 1976 flood did not seem to coincide with a catch decline in the South African fishery.

The above said, a number of factors including massive over-fishing have contributed to the decline of Namibian and South African rock lobster resources over the past 50 years. Recovery of the over-fished resources has been severely limited by highly variable recruitment which, to a large extent, results from the extreme environmental variability, and generally harsh conditions, of the central Benguela region. Coupled with declines in growth rate since the late 1980s, productivity was further reduced. Furthermore, since 1988, large-scale environmental changes have contributed to increased frequency and severity of low oxygen events, and occasional massive floods have deposited substantial quantities of mobile sediment into the nearshore ecosystems.

Fish and fisheries

Types of fish (and fisheries) response to changes in freshwater inflow to the marine environment fall into four broad categories (Lamberth et al., 2009):

1. Apparent negative responses to reduced flow, that is most likely due to rainfall / climate patterns throughout a biogeographical region rather than local flow rates.
2. Negative responses to local reduced flows that are real, e.g. reduced flow from the catchment will result in reductions in turbidity, preferred sediments, nutrient loads, phyto- and zooplankton production and ultimately reduced biomass and catches.
3. Cases of zero or negligible response, either positive or negative, to changes in flow.
4. Situations where flow reduction has a positive effect on catches. Correlations like these often prove to be less due to ecological drivers than to various aspects of fleet behaviour (and fisheries management).

Influence on nearshore nomadic coastal fish: Given the predominantly cool water of the Benguela upwelling region, linefish such as silver kob *Argyrosomus inodorus* and west coast steenbras *Lithognathus aureti* tend to be distributed within the warmer-water areas along the west coast. These warm areas are limited and tend to be in shallow bays, estuaries or warm-water plumes in the vicinity of estuary mouths. Hypothetically, the southward distribution of Angolan dusky kob *Argyrosomus coronus* and west coast steenbras *L. aureti*, both non-estuarine marine species, to as far as Langebaan Lagoon, may depend on the availability of warm-water refugia offered by estuary mouths and plumes. Consequently, a reduction in river flow may influence the distribution of these species by reducing the extent and availability of these refugia. A similar process is likely to facilitate exchange between South African, Namibian and Angolan stocks of *Argyrosomus inodorus, Pomatomus saltatrix* and *Lichia amia*. All three of these species as well as *Lithognathus lithognathus* and *L. aureti*, are important commercial and/or recreational fish in the region.

Most nearshore and estuarine fish either prefer or are tolerant of turbid waters and only move away when conditions approach tolerance levels. In turn, higher fish densities than those in surrounding waters were associated with turbidity plumes from marine mining activity to the north of the Orange River Clark et al. (1998). In reality, high flood-induced turbidity in the Orange and other nearshore areas appear to attract many 'turbidity-adapted' fish probably in response to potential refuge, parasite removal and/or more concentrated prey. Indeed, aggregations of 'turbidity-adapted' fish most notably silver kob *Argyrosomus inodorus*, start occurring in the surf-zone adjacent to the Orange Mouth up to two weeks prior to a flood event, probably in response to the first physico-chemical signals from the catchment. This said, the predictability of these aggregations and increased catchability make these fish vulnerable to over-exploitation.

Influence on demersal soft-sediment fish: Flow-driven changes in the magnitude and nature of sediment export to the marine environment will result in concomitant shifts in the diversity and abundance of fish that are distributed according to sediment preference or intensity of turbidity plumes off the Orange mouth. The best examples of these are bottom-dwelling flatfish species such as sole and skates which are distributed according to sediment type and particle size or small pelagic fish such as anchovy which find refuge in the surface layer turbidity plumes. Coincidental events such as floods, major dam construction and fishery collapse suggest a number of relationships, short and long-term between catchment flow and fish and fisheries in the marine environment. Species that were historically important in the demersal trawl fishery such as west coast sole *Austroglossus microlepis* now contribute less than 1% of the fish biomass in this soft sediment habitat.

West coast sole: West coast sole *Austroglossus microlepis* are targeted in South African and Namibian waters whereas east coast sole *Austroglossus pectoralis* are caught on South Africa's eastern seaboard. There are two recognized stocks of west coast sole a southern population centred on the Orange mouth and a northern population opposite the Skeleton Coast (Crawford et al., 1987). The trawl fishery for the southern population collapsed in the 1970s. The South African fishery has never recovered whereas there's been a resumption of the fishery in Namibian waters. It is not known whether this represents a recovery of the southern stock or a shift of the northern one southward. Worth mentioning, is that South Africa's east coast sole fishery has remained stable over the same time period.

Fishers in the sole trawl industry here and elsewhere in the world have long used rainfall (terrestrial runoff) as a predictor of catches in the following season. From 1970 to 1980, dam storage capacity on the Orange rose from 10% to 90% of that in the present state. The west coast sole fishery collapsed in the mid 1970s. Demersal trawl survey data (DAFF 1984 – 2011) indicate a weak but positive relationship between Orange flow and biomass estimates. However, there are stronger but negative relationships between sole and their predators e.g. gurnard *Chelidonichthys capensis* and smooth-hound shark *Mustelus mustelus*. Damming saw sediment discharge into the sea change in composition from predominantly silt to cohesive clays. Hypothetically, this influenced the burying ability and crypsis of juvenile sole leaving them more exposed to predators on the sediment surface and abrupt stock collapse. Changes in nutrient and food availability may also have played a role. In comparison, the stability in South Africa's east coast sole fishery may be partly attributed to there being no substantial change in the nature and volume of terrigenous sediment reaching the sea in that region

Small-pelagic fish: Small pelagic fish, notably anchovy, sardine and round-herring are the mainstay of the small pelagic purse-seine fishery on the South African and Namibian coast, the largest commercial fishery in the region. Small pelagic fish play a key role in regulating ecosystem function arising through their mid-level trophic position and influence on the abundance of both the plankton they feed on and the predators that feed on them (DAFF, 2012). Up until this project, there has been no real consideration given to the influence of catchment flows on the distribution of these pelagic fish in the Benguela Current Large Marine Ecosystem (BCLME).

Although ichthyoplankton (fish eggs and larvae) comprise a minor component of the overall zoo and phytoplankton biomass, it remains significant due to its dominance by small-pelagic fish (including pilchard, anchovy, and round-herring) and the commercial importance of this fishery in the region. High densities of larval and juvenile anchovy *Engraulis encrasicolus* are associated with the turbidity plume off the Orange and other estuaries on the west coast. The use of river plumes as refugia or juvenile nursery areas is characteristic of many small pelagic fish populations globally. Given the generally strong relationship between recruitment and end-of-the-year spawner biomass, it could also be expected that river flow, plume size and its influence on juvenile fish density may provide an additional useful predictor of said spawner biomass. Preliminary analysis of Orange River flow and pelagic fish biomass indicated a positive relationship between river-flow and juvenile densities of the three small-pelagic species but only the anchovy-flow one was significant. More robust analysis and numerical modelling of all variables should improve on this. Further, although on average only 10-20% of juvenile anchovy density can be explained by flow, higher densities of juveniles persist off the Orange and other river mouths in years when they're in low abundance or absent from other parts of the Namibian and South African coast.

Intertidal, subtidal and surf-zone fish: Most surf-zone, intertidal and subtidal fish on the Namibian and South African coast are common to sandy, rocky and mixed shore habitats but differ in abundance according to proportions of rock and sand and degree of exposure (Clark, 1997). In contrast to the low diversity of invertebrate communities on mixed shores in the region, fish species diversity and abundance is greatest at intermediate levels of exposure but also increases with habitat heterogeneity from sandy to mixed shores. These shores are characterised by extensive sand movement (terrigenous and marine) and the repeated scouring or burial of algal and invertebrate communities. Similarly, in the immediate nearshore, relationships between fish assemblages and flow from the Orange are likely to be indirect and according to the influence of catchment flow and sediment dynamics on the distribution of kelp and the burying or exposure of subtidal reefs.

Links between the rocky intertidal fish assemblage and the Orange Catchment, both positive and negative, are more tenuous than in other nearshore habitats. The 1988 Orange River floods diluted coastal waters causing mass mortalities of shallow-water invertebrates and kelps but fish escaped to deeper more saline waters. In contrast, floods frequently result in aggregations of fish adapted to low salinity and high turbidity in the Orange nearshore. In turn, the estuary and it's plume may offer refuge from low-oxygen events and other potentially lethal conditions in the sea.

Key ecosystem services/ biotic component	*Natural*	*Present*	*Sc 2*	*Sc 3*	*Sc 4*	*Sc 5*	*Sc 6*	*Sc 7*
Phytoplankton	-1	+1	+1	+1	+1	+1	+2	+2
Macrophytes	-2	+1	+1	+1	0	+1	+3	+3
Habitat-forming macrophytes (kelps)	-2	+1	+1	+1	0	+1	+3	+3
Soft-sediment macrofauna	+3	-1	-1	-1	-1	-1	-2	-2
Reef-associated macrofauna	-3	+1	+1	+1	+1	+1	+3	+3
Rock Lobster	-3	+1	+1	+1	+1	+1	+3	+3
Benthic biodiversity	+3	-1	-1	-1	-2	-1	-3	-3
Nomadic coastal fish (e.g. kob)	+3	-2	-2	-2	-1	-2	-3	-3
Demersal soft sediment fish (e.g. sole)	+3	-3	-3	-3	-3	-3	-3	-3
Small-pelagic fish (e.g. anchovy)	+3	-2	-2	-2	-1	-2	-3	-3
Intertidal, subtidal, surf-zone fish (e.g. clinids, galjoen)	+3	-1	-1	-1	0	-1	-1	-1

Marine EFR requirements

The nature and volume of sediment transport are the most important component of freshwater flow from the Orange to the nearshore marine environment in particularly with regards to shaping benthic habitats and maintenance of the refuge and foraging area provided by the river plume. There is unlikely to be any discernible change from scenarios (Sc) 1 – 4 and probably 5 well as most of the significant change has already occurred from reference to the present state. Scenarios 6 and 7 however, could be severe both in terms of flow and export of sediment to the sea. The severe flow reduction of both these scenarios is magnified by the development of a large dam at Vioolsdrift (relative close proximity to the sea that will reduce sediment transport to the nearshore marine environment).

Three selected objectives of a marine environmental flow requirement (EFR) for the Orange River would be re-establishment of benthic habitat suitable to sole and ultimately the fishery, maintenance of the river plume for juvenile small-pelagic fish, and maintenance of the turbid warm-water in the nearshore suitable to nomadic fish that occurs in the high-flow season. Collapse of the sole fishery occurred during the 1970s when 80% of the current dams came into existence and drastically altered sediment export to the sea. Re-establishment of this benthic habitat is likely unachievable under any of the scenarios as it would require redesign of most impoundments in the catchment to secure sediment releases. More feasible are maintenance and enhancement of the river plume and seasonal warm water ‘cell’ that occurs during high-flow in the nearshore. These are both likely to persist under Sc 2 – 5 and would be optimised in Sc 4.

Ecosystem services

Rivers carry nutrients from their catchments which they discharge into the nearshore marine environment. Sediment outputs from rivers can play an important role in maintaining benthic habitats offshore, which has knock-on effects for demersal (bottom dwelling) fisheries. The continental shelf offshore of the mouth of the Orange Estuary is thought to be a critical nursery area for several fish stocks that make up a large proportion of the value of commercial fisheries in South Africa. Juvenile anchovy *Engraulis encrasicolus*, a mainstay of the pelagic industry utilise the turbidity plume as a nursery whereas west coast sole *Austroglossus microlepis* distribution and abundance varies according to the amount and type of sediment discharged from the catchment. Although their importance is indisputable, these linkages are not well understood.

Monitoring

Nearshore marine environment: Improving the baseline

Additional surveys to improve the baseline information is summarised below.

Component	*Baseline survey*	*Temporal scale*
Sediments	Sample suspended sediment load at Vioolsdrift.	Daily
Remote sensing	Observations on turbidity, salinity, temperature and chlorophyll-a	Daily
Fish	Small pelagic acoustic surveys in South African and Namibian coastal	2X annual (i.e. quarterly)
Invertebrates	Benthic and beach monitoring on both Namibian and South Africa side.	Annual (i.e. quarterly)

Conclusions and recommendations

The main focus of the nearshore marine study was to establish if there is ecological connectivity between the Orange River and the nearshore marine environment. The work done as part of this study shows a clear link between the sediments being transported by the Orange Estuary and the nearshore marine habitat adjacent to the Orange Estuary. This in turn is driving a range of biological responses, e.g. increase fisheries production. However, the exact nature and strength of the dependencies would require intensive research into the future. Key aspects that needs investigation include studies in changes in trophic interactions (e.g. gurnard/sole predator interactions), dedicated satellite imagery study to investigate long term effects of floods on the nearshore environment, and extending the ocean tracking network in Southern Africa to include the marine environment adjacent to the Orange River mouth (acoustic array that currently only Mozambique to Cape Point).

The Namibian and South African legislative framework make provision for the sustainable management of their biodiversity and fisheries in the context of their respective international biodiversity commitments. Thus said, at present both the Namibian and South African National Water acts are silent on the freshwater requirements of the nearshore marine environment. There is therefore no legal requirement to provide an EFR to the sea. In order to achieve the fist, it is recommended that the various national frameworks and policies that regulate water resource management in these two countries be reviewed to address this aspect.

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Abbreviations

AMD	*Acid mine drainage*
BCLME	*Benguela Current Large Marine Ecosystem*
BUS	*Benguela upwelling system*
CSIR	*Centre of Scientific and Industrial Research*
DAFF	*Department of Agriculture, Forestry and Fisheries*
DEA	*Department of Environmental Affairs*
DIN	*Dissolved inorganic nitrogen*
DIP	*Dissolved inorganic phosphate*
DRS	*Dissolved reactive silicate*
EFR	*Environmental flow requirement*
EMP	*Environmental management programme*
HAB	*Harmful algal blooms*
HWL	*High water level*
HWS	*High water spring*
IDP	*Integrated development planning*
IWRM	*Integrated water resources management*
LC	*Lethal concentration*
LS	*Landsat*
LWL	*Low water level*
MODIS	*Moderate-resolution Imaging Spectroradiometer*
MSL	*Mean sea level*
MSY	*Maximum sustainable yield*
MUW	*Modified upwelled water*
ORASECOM	*Orange-Senqu River Commission*
PES	*Present ecological state*
PIM	*Particulate inorganic matter*
SACW	*South Atlantic central water*
SADC	*South African Development Community*
SADCO	*South African data centre for oceanography*
Sc	*Scenario*
SST	*Sea surface temperature*
TAC	*Total allowable catch*
TP	*Total phosphorus*
TSS	*Total suspended solids*

1 Introduction

1.1 Study scope

The Orange-Senqu Strategic Action Programme supports the Orange-Senqu River Commission (ORASECOM) in developing a basin-wide plan for the management and development of water resources, based on integrated water resources management (IWRM) principles. The project is currently in the proses of finalising a Transboundary Diagnostic Analysis that will serve as the scientific basis for developing a set of interventions under the framework of a basin-wide Strategic Action Programme and associated National Action Plans in the riparian States. In 2009 the ORASECOM commissioned a study into environmental flow requirements (EFR) in the lower Orange-Senqu River basin. This study was recently completed and has defined both the present ecological state (PES) and the environmental flows that would be required to maintain a range of ecological states at eight representative sites upstream of the confluence of the Fish and Orange Rivers. EFRs of the ephemeral but nevertheless significant Fish River and the Orange River from its confluence with the Fish down to the mouth were not covered in any detail by the GIZ study. This outstanding work is to be the subject of this research project. One of the focus areas of this larger project is the Orange River mouth (the Estuary) and the adjacent marine environment.

The Orange Estuary study will focus on sediment and hydrodynamics, water quality, microalgae, vegetation, invertebrates, fish and birds. For the Orange Estuary component the following is to be undertaken:

- develop and implement a baseline monitoring programme covering flow-related biophysical parameters;
- research and assess non-flow-related impacts on the estuary;
- describe the present ecological state of the estuary;
- determine the environmental flows that would be required to maintain a range of ecological conditions in the estuary;
- recommend attainable and satisfactory environmental flows for the estuary; and
- design a long-term monitoring programme to assess the efficacy of environmental flows and other management interventions for the estuary.

Furthermore, this project proposes an assessment on the role of freshwater inflows and associated fluxes in the coastal marine ecosystems linked to the estuary and the potential effects of changes in the freshwater-related fluxes into these ecosystems. This will be done in order to recommend allowable changes in freshwater inflow into the marine environment within the constraints of maintaining or improving the present health status of the marine ecosystem and optimisation of the existing ecosystem services provided by the coastal ecosystem. This study will focus on sediment and hydrodynamics, microalgae, invertebrates and fish using available catch data, remote sensing and numerical modelling.

1.2 Approach

No official method exists for determining the EFR of the nearshore marine environment in South Africa or Namibia. However, Van Ballegooyen et al. (2003) developed a comprehensive assessment framework for the evaluation of EFR of the nearshore marine environment based on international best practice. This study proposes to use a modified version of the propose framework to evaluate a range of freshwater scenarios by means of the following steps:

- Determine the legislative requirement for setting an EFR for the nearshore marine environment: Review the policies and legislation of relevance to the assessment and management of the freshwater requirements of the marine environment, including particular obligations under various treaties and international agreements.
- Definition of the ecosystem extent (biogeographic boundaries) and describe key properties: The boundaries of ecosystem of relevance to the assessment need to be defined based on the extent of the marine ecosystem potentially impacted by change of freshwater inflow (i.e. an appropriate definition of the ecological 'footprint'). A description of the ecosystem and its key components is also required to ensure a comprehensive EFR assessment and appropriate ecosystem management recommendations.
- Identify resource utilisation in ecosystem: The resource utilisation needs to be identified in order that, as a minimum, appropriate keystone/indicator species can be selected for the assessment of the freshwater requirements of the marine environment.
- Setting of environmental objectives: Based on the identified policy and legislative requirements, resource utilisation and characteristics of the ecosystem under consideration; specific management and environmental quality objectives need to be developed.
- Hydrological scenario assessment: Describe the changes in the past, present and future flow regime of the catchment to provide context to the assessment.
- Identification relevant abiotic components (habitat) and assess the response to flow modification: The critical abiotic components (e.g. salinity, nutrients, sediments, etc.) influencing the quality of the required habitats during the various life-cycle stages of the key biotic species need to be identified. The various abiotic (and biotic) components need to be integrated and/or aggregated, such that they are relevant to determining the biotic response.
- Describe the implications of flow alteration on selected biological components: This include the following:
    - selection of keystone or indicator species: Based on the management objectives, the defined ecosystem boundary and resource utilisation, keystone and/or indicator species need to be identified to minimise the complexity of the assessment, allow for the setting of clear and measurable environmental objectives, and ensure practical and effective management advice;

  - determination of life-cycle and habitat requirements: An analysis of the various life-cycle stages of the identified keystone or indicator species is required to identify the habitat requirements for the various life-cycle stages and consequently the abiotic (and biotic) components of relevance;
  - predict the possible responses, if any, to predicted change in abiotic components: Provide an analysis of the biotic responses to predicted abiotic change.
- Evaluation of socio-economic importance of marine aquatic ecosystems and resource uses: The outcomes of the scientific assessment of the potential impacts associated with changes in freshwater inflow into marine ecosystems need to be linked to the socio-economic implications of these changes as this is the primary basis upon which water resource allocations are likely to be made. Based on the outcome of this step, there may be modification of the recommended freshwater requirements for the nearshore marine ecosystems under consideration.
- Recommendation of Freshwater Requirements: The adequacy of the scientific assessment will be determined by whether or not there is sufficient understanding and/or measurements to translate management and environmental quality objectives into specific freshwater requirements or target values, based on usage of the nearshore marine environment as an existing or potential future resource. Typically this is only possible for a specific coastal and nearshore region once existing and potential future resource utilisation in the region of interest has been mapped and there is a reasonable understanding of the functioning of the ecosystems of relevance.

2 Policy and legal framework

For transboundary systems like the Orange Estuary and nearshore marine environment, the applicable legal framework can be extensive and complex where the laws of more than one country apply, in the case South Africa and Namibia. This chapter aims to provide a brief overview of the key international conventions and national legislation, binding to the Orange River Estuary and near shore marine environment.

The information presented below is by no means comprehensive, but is intended to provide background to the identification of environmental objectives and the related freshwater requirements.

2.1 International conventions

- Agreement between Botswana, Lesotho, Namibia and South Africa on the establishment of the Orange River Commission (ORASECOM) (November 2000): The ORASECOM agreement recognises the Helsinki Rules, the United Nations Convention on the Non-Navigational Uses of International Watercourses and the previous South African Development Community (SADC) Protocol on Shared Watercourse Systems. It also refers to the Revised Protocol on Shared Watercourses with respect to definitions of the key concepts 'equitable and reasonable' and 'significant harm' (Earle et al., 2005). The main result of this agreement is the establishment of ORASECOM as an international organization with legal personality and certain institutions and powers.
- United Nations Convention on the Law of Non-navigational Uses of International Watercourses (1997) that codified international water law: In itself the convention is a framework agreement which allows for ad hoc watercourse agreements to be adopted for specific international watercourses. Watercourse states may enter into one or more agreements, called 'watercourse agreements', which apply and adjust the provisions of the Convention to the characteristics and uses of a particular international watercourse or part thereof. It contains a foundational provision with respect to "equitable and reasonable utilization and participation" and details the factors relevant to equitable and reasonable utilization.
- SADC Revised Protocol on Shared Watercourses (August 2000): Gives effect to some most of the principles of the United Nations Convention on the Law of Non-navigational Uses of International Watercourses (1997). The overall objective is to foster closer cooperation for judicious, sustainable and coordinated management, protection and utilization of shared watercourses and to advance the SADC Agenda of regional integration and poverty alleviation. In particular, the Protocol wants to promote the establishment of shared watercourse agreements and institutions, to advance sustainable, equitable and reasonable utilization, sound environmental management, harmonization and monitoring

of legislation of the states involved and the promotion of research, technology development, information exchange and capacity building (Hiddema and Erasmus, 2007).

- Convention of Migratory Species of Wild Animals (1979) (Bonn Convention): A response to the need for nations to co-operate in the conservation of animals that migrate across their borders. These include terrestrial mammals, reptiles, marine species and birds. Special attention is paid to endangered species. Currently only South Africa is a signatory to this convention
- United Nations Convention on Biological Diversity (1992): Has three main objectives: the conservation of biological diversity; the sustainable use of biological resources; and the fair and equitable sharing of benefits arising from the use of genetic resources. Parties to the convention are required to develop national strategies, plans or programmes, or adapt existing ones, to address the provisions in the convention, and to integrate the conservation and sustainable use of biodiversity into sectoral and cross-sectoral plans, programmes and policies. Both Namibia and South Africa are signatories to this convention.
- Convention on Wetlands of International Importance especially as Waterfowl Habitat (1971) (Ramsar Convention): Aims to stem the loss and to promote wise use of all wetlands. The Convention includes estuaries in its definition of wetlands. The Convention defines wetlands as 'areas of marsh, fen, peat land or water whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres' (which includes estuaries). Both South Africa and Namibia are signatories to this convention.
- Agreement on the Vioolsdrift and Noordoewer Joint Irrigation Scheme between the Government of the Republic of Namibia and The Government of the Republic of South Africa (1992) (Earle et al., 2005).
- Agreement between the Government of the Republic of South Africa and the Government of the Republic of Namibia on the Establishment of a Permanent Water Commission (1992) (Earle et al., 2005).

2.2 National legislation

2.2.1 Namibia

National legislation considered potentially relevant to management of the Orange River Estuary includes:

- Water Resources Management Act (No. 24 of 2004), with the Ministry of Agriculture, Water and Forestry, governs the management, development, protection, conservation, and use of water resources in Namibia and includes estuaries as a water course. This act repeals Water Act (No. 54 of 1956). For example, Part VIII deals with the abstraction and use of water, while Part XI covers water pollution control, including the control of effluent discharges;

- Environmental Management Act (No. 7 of 2007), with the Ministry of Environment and Tourism as lead agent, aims to promote the sustainable management of the environment and the use of natural resources by establishing principles for decision making on matters affecting the environment and provide for a process of assessment and control of activities which may have significant effects on the environment;
- Namibia Water Corporation Act (No. 12 of 1997), with the Namibia Water Corporation (NamWater) as lead agent, primarily deals with water supply and the role and objectives of NamWater;
- Minerals (Prospecting and Mining) Act (No. 33 of 1992), with the Ministry of Mines and Energy as lead agent, provides for the reconnaissance, prospecting and mining for, and disposal of, and the exercise of control over, minerals in Namibia.

2.2.2 South Africa

Key national acts considered potentially relevant to management of the Orange Estuary are briefly summarised below. The key acts include:

- National Environmental Management Act (No. 107 of 1998), with the Department of Environmental Affairs as lead agent, the act provides for co-operative environmental governance through the establishment of national environmental management principles and their incorporation into decisions affecting the environment. The act emphasises co-operative governance and assists in ensuring that the environmental right in the Constitution are protected. The act provides that sensitive, vulnerable, highly dynamic or stressed ecosystems, such as estuaries or coastal ecosystems, require specific attention in management and planning procedures, especially where subjected to significant human resource usage and development;
- National Water Act (No. 36 of 1998), with Department of Water Affairs as lead agent, has one of its important objectives as ensuring protection of the aquatic ecosystems of such South Africa's water resources. The act requires policies to be in place that provide guidance in developing resource quality objectives, i.e. specifying aspects such as freshwater inflow, water quality, habitat integrity, biotic composition and functioning requirements. The act requires classification and resource quality objectives to be determined for all water resources. The act does not recognise the nearshore marine environment as an aquatic ecosystem in need of freshwater.
- National Environmental Management: Integrated Coastal Management (No. 24 of 2008), with the Department of Environmental Affairs as lead agent, aims at establishing a system of integrated coastal and estuarine management in South Africa, including norms, standards and policies, in order to promote the conservation of the coastal environment, and the ecologically sustainable development of the coastal zone, to define rights and duties in relation to coastal areas, to determine responsible organs of state in relation to coastal areas and to give effect to South Africa's international obligations in relation to coastal matters and to provide for related matters;

- Marine Living Resources Act (No. 18 of 1998) amended 2000, with the Department of Agriculture, Forestry and Fisheries as lead agent, has objectives and principles that deal with the utilization, conservation and management of marine living resources (including estuarine resources), the need to protect whole ecosystems, preserve marine biodiversity and minimize marine pollution, as well as to comply with international law and agreements and to restructure the fishing industry. Chapter 4 of the act deals with the declaration of Marine Protected Areas and empowers the Minister to declare Marine Protected Areas where various activities are prohibited;
- Mineral and Petroleum Resources Development Act (No. 28 of 2002), with the Department of Minerals as lead agent, contains the statutory requirements regarding the enforcing of environmental protection and management of mining impacts, including sand and coastal mining. The Act requires environmental management programmes (EMP) that identify a mine's impact on the environment and provide a clear programme on how these will be managed, based on an environmental impact assessment;
- Local Government: Municipal Systems Act (Act 32 of 2000), with the Department of Provincial and Local Government as lead agent, deals with integrated development planning (IDPs) (municipalities are obliged to prepare and to update IDPs regularly). An IDP is intended to encompass and harmonise planning over a range of sectors such as water, transport, land use and environmental management;
- National Environmental Management: Biodiversity Act (Act 10 of 2004), with the Department of Environmental Affairs as lead agent, provide for the conservation of biological diversity, regulate the sustainable use of biological resources and to ensure a fair and equitable sharing of the benefits arising from the use of genetic resources The Act states that the state is the custodian of South Africa's biological diversity and is committed to respect, protect, promote and fulfill the constitutional rights of its citizens. It also recognizes that South Africa is party to, amongst others, the Convention on Biological Diversity, the Convention on Wetlands of International Importance especially Waterfowl Habitat (Ramsar Convention) and the Convention on Migratory Species (Bonn Convention);
- Conservation of Agricultural Resources Act (No. 43 of 1983), with the Department of Agriculture, Forestry and Fisheries as lead agent, provide for the conservation of the natural agricultural resources of South Africa by: the maintenance of the production potential of land; the combating and prevention of erosion and weakening or destruction of the water sources; and the protection of the vegetation and the combating of weeds and invader plants.

2.3 Conclusion

The Namibian and South African legislative framework make provision for the sustainable management of their biodiversity and fisheries in the context of their respective international biodiversity commitments. Thus said, at present both the Namibian and South African National Water acts are silent on the freshwater requirements of the nearshore marine environment. There is therefore no legal requirement to provide an EFR to the sea. In order to achieve the fist, it is recommended that the various national frameworks and policies that regulate water resource management in these two countries be reviewed to address this aspect.

3 Characterisation of Orange River inflow

To provide some indication of flow alteration, Vioolsdrift flows was analysed for a range of flow regimes (Figure 1). All flows that exceed 75% of daily flows for the analysed period was classified as 'high flows'. A number of flow ranges were identified as being important (see Figure 1 and Table 2) in terms of floods and pulses of high flow in the Orange River and its estuary. A small flood event was defined as initial high flow with a peak flow greater than two year return interval event (associated with monthly volumes exceeding 1,000 million cubic metres (Mm^3). A large flood event was defined as an initial high flow with a peak flow greater than 10 year return interval event (associated with monthly volumes exceeding 5,000 Mm^3). Large floods can be further subdivided into two categories, namely large floods with monthly volumes exceeding 5,000 Mm^3 but less than 8,000 Mm^3, and very large floods having flood volumes that exceed 8,000 Mm^3. Extreme low flow is defined as an initial low flow below 10% of daily flows for the period. The high flows seem to correspond quite well with the Southern Oscillation Index (Figure 2), i.e. with El Niño and La Niña events. The larger floods seem to coincide with period of rapid transition between El Niño and La Niña conditions, reflecting a strong modulation of flows by these large-scale forcing mechanisms. However Figure 2 also indicates significant changes over time that is related to dam development in the Orange River catchment. The most noticeable change is that the number of large floods and particularly the number of small flood have declined significantly since the early 1970's when a number of large dams were developed in the Orange River Catchment (see Table 1).

Table 1. Major impoundments in the Orange River catchment

Name of Dam	***Year commissioned***	***River on which located***	***Full capacity (10^6 m^3)***
Van Wyksvlei	1884	Van Wyksvlei	145
Smart Syndicate	1912	Ongers	100
Vaalharts	1936	Vaal	63
Vaal	1938	Vaal	2 536
Kalkfontein	1938	Riet	324
Erfenis	1960	Grootvet	211
Allemanskraal	1960	Sand	176
Krugersdrif	1970	Modder	77
Bloemhof	1970	Vaal	1 269
Gariep (Hendrik Verwoerd)	1972	Orange	5 670
Welbedagt	1973	Caledon	40
Vanderkloof (PK le Roux)	1977	Orange	3 236
Sterkfontein	1977	Wilge	2 617
Groot Draai	1978	Vaal	359

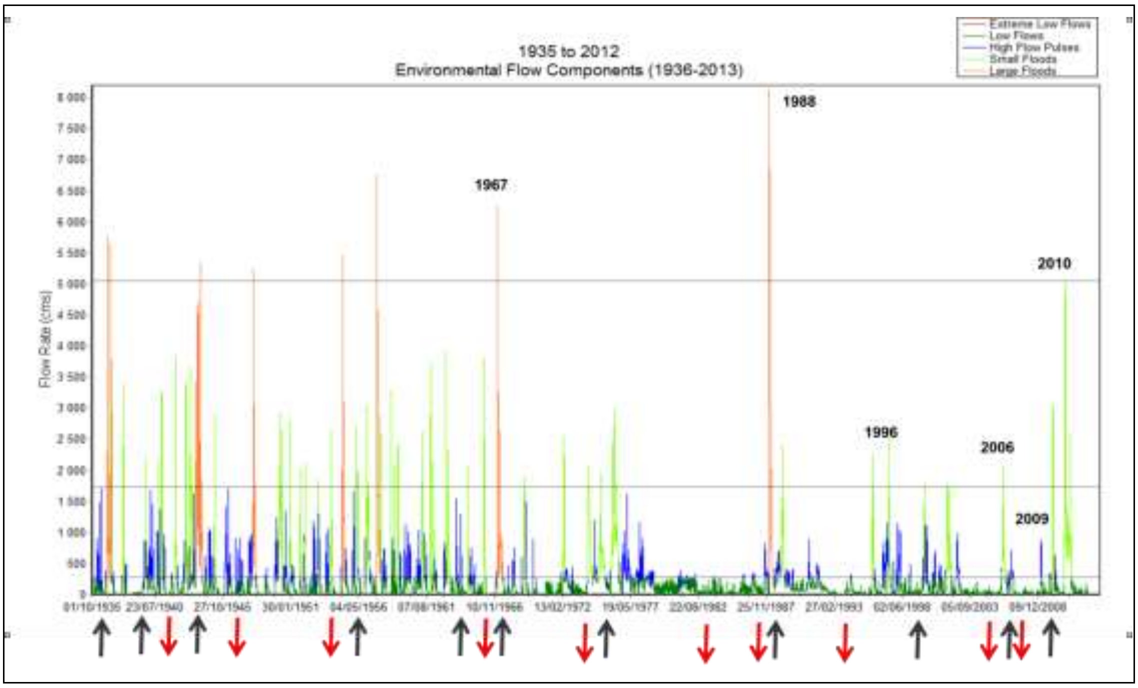


Figure 1. Measured Orange River flows for the period 1936 to 2011. The arrows at the bottom of the figure indicate elevations (blue) or decreases (red) in the Southern Oscillation index (Source: http://www.bom.gov.au/climate/current/soihtm1.shtml).

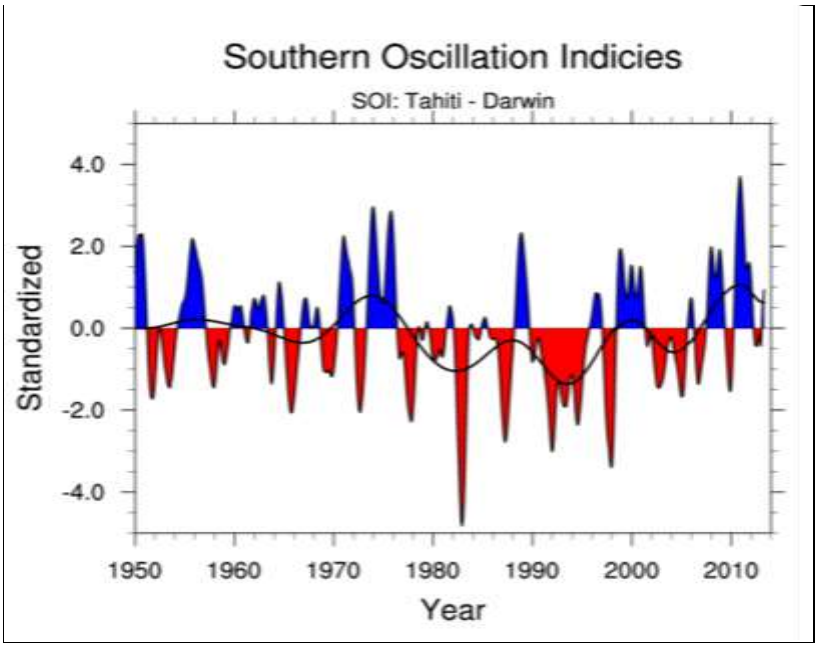


Figure 2. Southern Oscillation Index for the period 1950 to 2013 (Source: http://www.cgd.ucar.edu/cas/catalog/climind/soi.html).

The number of occurrences of these flood sizes and freshwater 'pulses' were then determined for a 66-year simulated monthly volume data extending from 1930 to 1996 (see Table 2 below) for estimated reference conditions and present state.

As part of the modelling study only the larger flow categories reported in Table 2 were simulated in the model for their response in the marine environment. For the lowest flow category, the 'Inside estuary' flow category, the magnitude of the freshwater 'pulses' are such that there are no significant freshwater flows (and associated fluxes) into the marine environment. Therefore in this study only the four largest flow categories have been simulated and evaluated, namely freshwater pulses, small flood, large floods and very large floods (see Table 2)

Table 2. Number of occurrences of flood and freshwater pulses in a 66-year simulation period in the Orange River under reference conditions and present state

Flood type	***Flood volume (Mm^3)***	***Number of flood occurrences***	
		Reference	***Present***
Inside Estuary	< 150	238	571
Pulse	150 – 1000	350	139
Small floods	1000 – 5000	179	74
Large Floods	5000 – 8000	24	7
Very Large Floods	>8000	1	1

It is important to take cognisance of the changes in frequency of the defined flood events under the various simulations. In the change from reference conditions to the present state there has been a dramatic decrease in occurrence of all but the very largest flood events. This is particularly true for the smaller flood and high flow pulse events.

3.1 Inorganic nutrients

To characterise the inorganic nutrient load in the Orange River – typical of what could enter the marine environment – data collected by South Africa's Department of Water Affairs at the Ernst Oppenheimer Bridge was used. This monitoring station is the closest to the sea. Also displayed on the graphs is median monthly flow, measured at Vioolsdrift.

Median monthly dissolved inorganic nitrogen (DIN), dissolved inorganic phosphate (DIP) and dissolved reactive silicate (DRS) concentrations measured in river inflow (1995 – 2011 is presented in Figure 3). Also presented in these graphs are the median monthly flow measured at Vioolsdrift over this same period. Median DIN concentrations (mainly comprising nitrate- N) correlate well with median monthly flow, where highest concentrations are measured during months of high river inflow. DIN concentration averaged about 100 μg/ℓ, peaking at 400 μg/ℓ in April. De Villiers and Thiart (2007) estimated DIN concentrations in river flow in the lower Orange catchment under reference condition to be <50 μg/ℓ. Higher concentration under the present state are mostly associated with runoff agricultural land-use in the catchment.

DIP concentrations in river flow did not show a strong correlation with flows (Figure 3). DIP concentrations averaged 30 µg/ℓ with a slight peak in April (50 µg/ℓ). De Villiers and Thiart (2007) estimated DIP concentrations in river flow from the lower Orange catchment under Reference Condition to be <10 µg/ℓ. Higher concentration under the present state are mostly associated with runoff from agricultural land-use in the catchment.

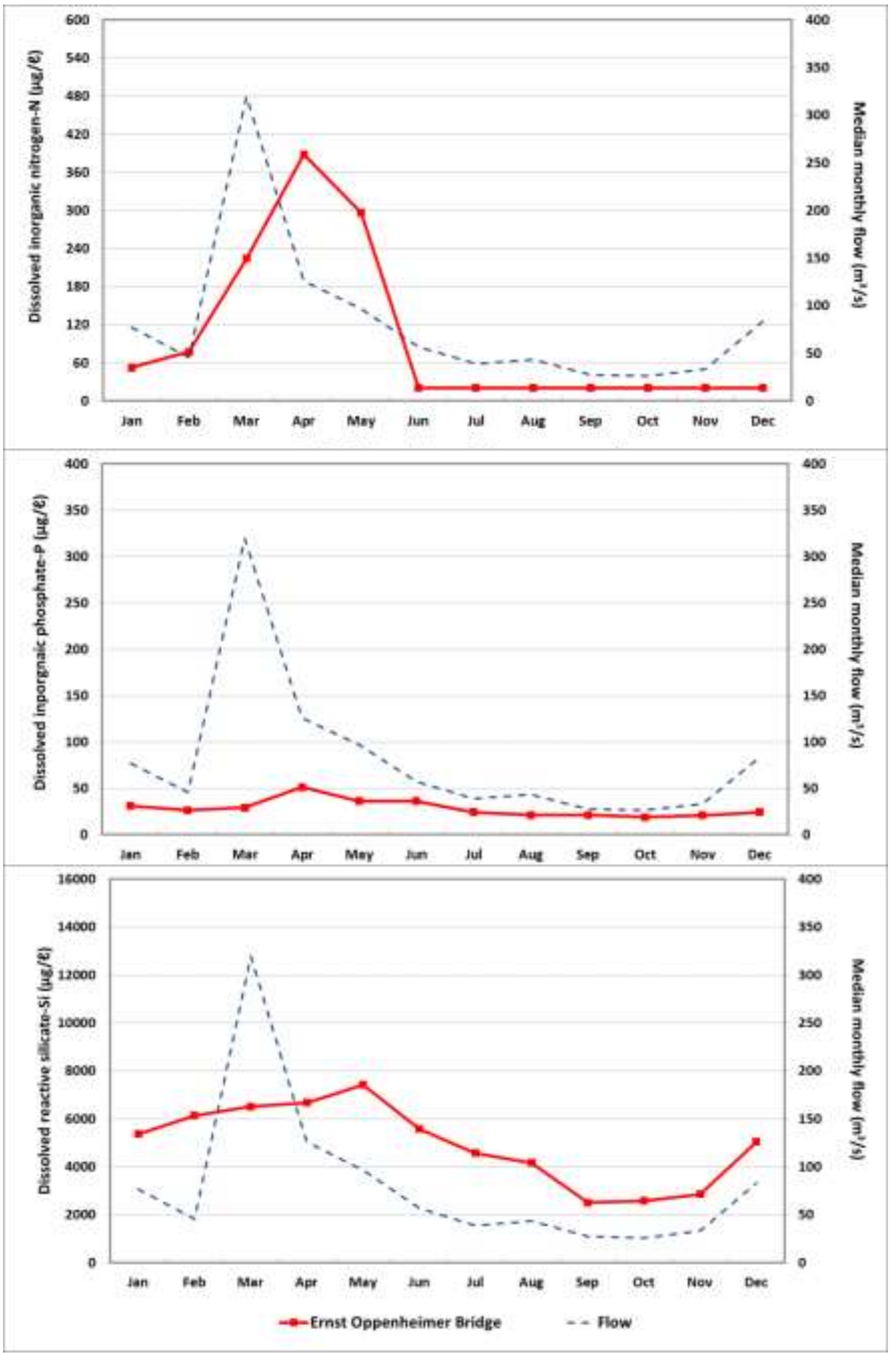


Figure 3. Monthly median DIN, DIP and DRS concentrations (1995 – 2011) measured at the Ernst Oppenheimer Bridge (red), as well as median monthly river flow measured at Vioolsdrift (Source: DWA)

Median DRS concentrations correlate well with median monthly flow, where highest concentrations (7,500 µg/ℓ) are associated with high flow periods and lower concentrations (2,500 µg/ℓ) with low flow periods. Concentrations under the reference condition was most likely similar to those measured during 1995 to 2011 as DRS can be naturally high in river water, compared with seawater, linked to catchment geological characteristics (Eagle and Bartlett, 1984).

3.2 Sediment and particulate organic inputs

The Lesotho Highlands, where the Orange system originates, is characterized by steep gradients. In the Lesotho Highlands soils are classed as Mountain Black Clays, shallow at high altitude and easily eroded by cultivation and overgrazing (CSIR, 2011). During summer, soils on the summit become waterlogged and in winter they usually freeze, increasing their susceptibility to erosion. Most of the remainder of the Orange catchment is covered by sands or weakly developed soils. Most of the basin is regarded as being medium to high risk in terms of soil erosion, with the exception of mainly the Kalahari component.

The mass of sediment discharged by the Orange River is estimated to be around 17 million tons per year (Bremner et al., 1990). While this may be small in comparison to the world's leader, the Ganges-Brahmaputra which discharges in the order of 1,670 million tons per year, it still represents a significant volume of sediment entering the Benguela Current System each year. Present state discharges of sediment from the Orange system are considerably lower than those recorded prior to the 1960s but of a similar magnitude to those reported for geological time scale (Table 3). A concomitant change has also occurred in the texture of the suspended sediment load carried by the river, which has changed from silt-dominance in pre-1970 material, to clay dominance since this time (Bremner et al., 1990). Both of these effects have been attributed to agricultural malpractices in the parts of the catchment (Northeastern Cape in South Africa) in this early period (Rooseboom and Mass, 1974), and the fact that easily erodible topsoil was stripped from the upper Orange catchment during the early 1930s, and the rapid increase in the number and size of dams that were constructed in the catchment in the early 1970s (Bremner et al., 1990). These changes are mirrored in the changes in the suspended sediment concentrations measured in flood waters in March 1988 (7,4 mg/ℓ) compared with similar sized floods in April 1961 (17,4 mg/ℓ), March 1965 (15,5 mg/ℓ) and November 1955 (14,5 mg/ℓ) (Bremner et al., 1990). Historically, the bulk of this sediment carried by was reportedly derived from the upper portion of the catchment from whence most of the runoff is also derived (Rooseboom and Mass, 1974; Rooseboom 1975; Rooseboom and Harmse, 1979), whereas this has now shifted to the lower catchment, below the major impoundments on the system. Bremner et al. (1990), list bank erosion and river bed scour, derived from the river channel downstream of the major dams situated near the Orange-Vaal confluence, as the main sources of sediment in the river in the 1988 floods. Large amounts of sediment were also removed from the estuary as well, with vertical scour of at least eight metres deep being recorded at the bridge, and lateral erosion of the salt marsh of about 400 m (Bremner et al., 1990). The total volume of sediment discharged during the 1988 flood was estimated at 64,2 million tons, very similar to the mean annual sediment discharge of 60,4 million tons measured at Prieska/Upington (close to the mouth) between 1930 and 1969 (Bremner et al., 1990).

Table 3. Variation in sediment discharge rates of the Orange River (after Bremner et al., 1990)

Period	***Information source***	***Sediment discharge rate (in million tons per year)***
Geological time		
Late Cretaceous	Dingle and Hendey (1984)	24
Palaeogene	Dingle and Hendey (1984)	4.5
Neogene	Dingle and Hendey (1984)	0.8
Historical time		
?	Lisitizin (1972)	153
Pre-1921	Perry (1988)	119
1929 – 1934	Rooseboom and Mass (1974)	89
1934 – 1943	Rooseboom and Mass (1974)	56
1943 – 1952	Rooseboom and Mass (1974)	52
1952 – 1960	Rooseboom and Mass (1974)	46
1960 – 1969	Rooseboom and Mass (1974)	34
1980's	Rooseboom (pers. comm.) cited in Bremner et al. (1990)	<17

However, not all of the terrigenous sediment inputs onto the Benguela continental shelf region are of riverine origin. The powerful easterly adiabatic offshore wind ('berg-wind') that occur along the southern African West Coast typically during the austral winter (Figure 4) also play a significant role in sediment input into the coastal marine environment, potentially contributing the same order of magnitude of sediments as the annual estimated input of sediment by the Orange River (Shannon and Anderson, 1982). Transport up 150 km offshore has been observed for a single wind event and was particularly pronounced in the vicinity of the Orange River (Shannon and Anderson, 1982). Whitaker (1984) undertook a detailed study of winter-season berg-winds using daily METEOSAT satellite images from 1978, 1979, 1982 and 1983, supplied by the Centre of Scientific and Industrial Research (CSIR). Seven offshore dust plumes, oriented from NE onshore to SW offshore, were observed in 1978, 12 in 1979, five in 1982 and eight in 1983 (Penney et al., 2008).

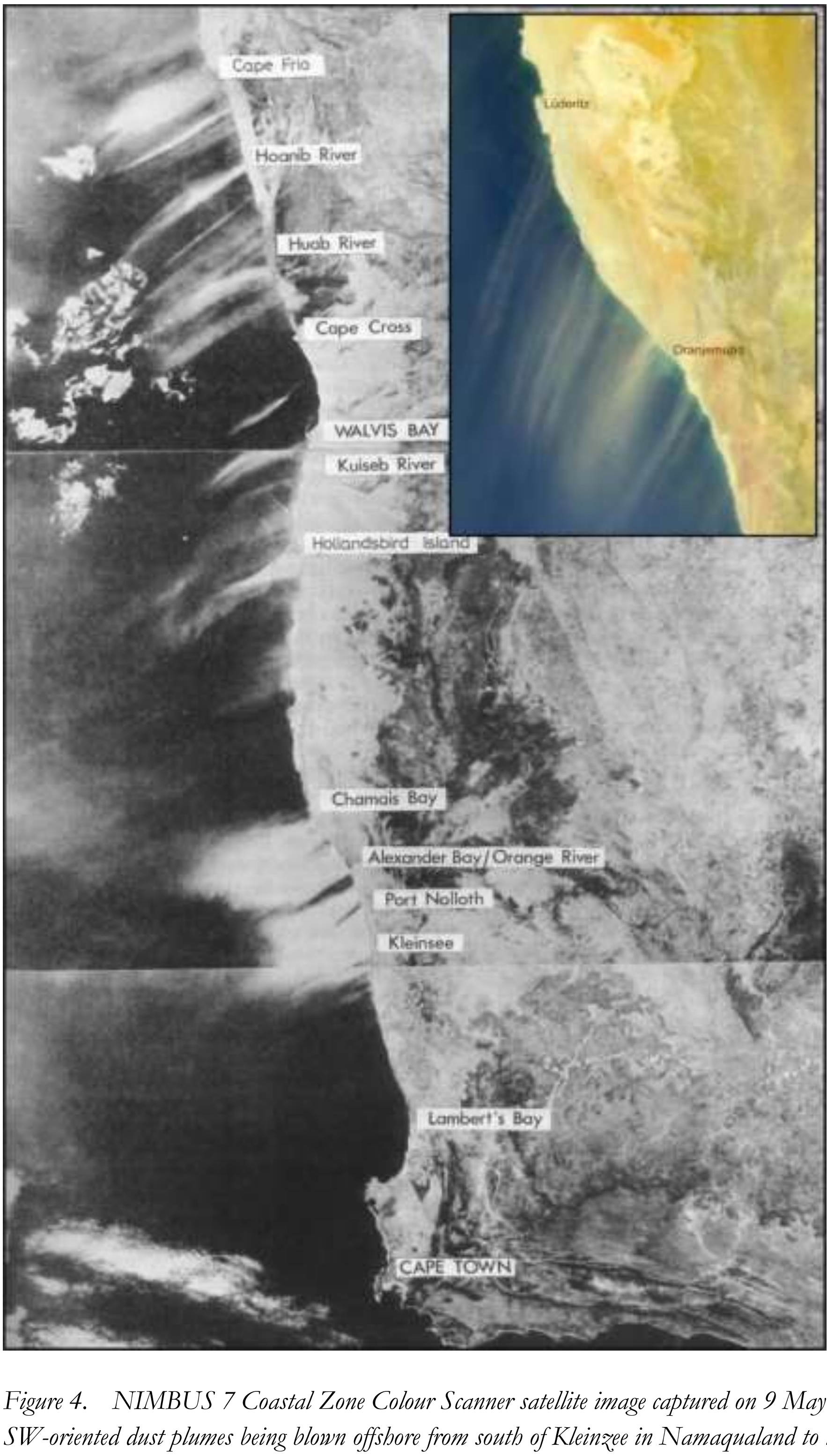


Figure 4. NIMBUS 7 Coastal Zone Colour Scanner satellite image captured on 9 May 1979 showing a series of SW-oriented dust plumes being blown offshore from south of Kleinzee in Namaqualand to the Kunene River on the border between Namibia and Angola (from Shannon and Anderson, 1982). Inset: satellite image of a similar berg-wind event along the southern Namibian coastline, between Lüderitz and Oranjemund.

3.2.1 *Particulate organic matter*

No data could be obtained on particulate organic matter concentrations in river inflow. The only measured data that was available on organic input is the Total phosphorus (TP) data measured by South Africa's DWA at the Ernest Oppenheimer Bridge (1995 – 21011) (Figure 5).

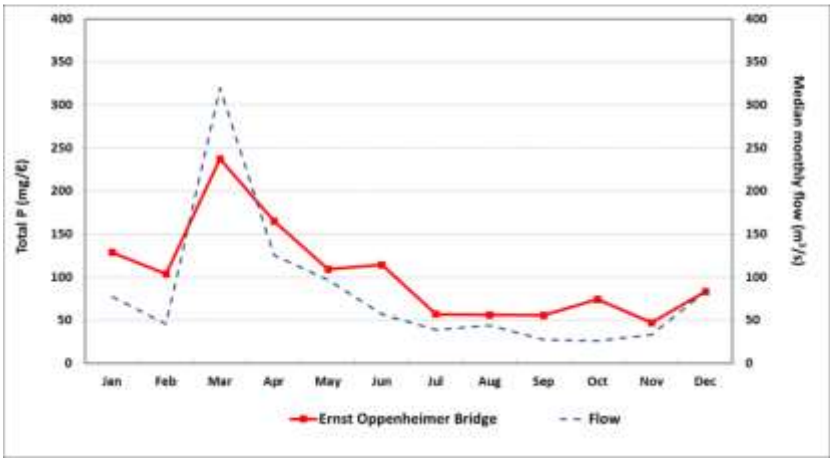


Figure 5. Monthly median TP concentrations (1995 – 2011) measured at the Ernst Oppenheimer Bridge (red), as well as median monthly river flow measured at Vioolsdrift (Source: DWA)

TP concentrations include both dissolved and particulate P. Comparing the TP results to the DIP (Figure 5), it is clear that a substantial component of TP in river flow is present in the organic fraction – most likely particulate organic matter. It can therefore be assumed that during high flows river input may also be a source of particulate organic nutrients to the near-shore, although this could not be quantified at this stage.

4 Characterisation of nearshore marine ecosystem

4.1 Water column processes

The marine environment bordering the Orange River mouth forms part of the dynamic and productive Benguela Upwelling System (BUS) along the coasts of South Africa and Namibia (Figure 6). The Orange River mouth itself is nestled in between two major upwelling cells of the BUS, i. e., the Namaqua Upwelling Cell to the south and the Lüderitz Upwelling Cell to the north (Figure 6). It is therefore necessary for the physical and biogeochemical processes of the greater Benguela Upwelling System to be taken into consideration when describing the marine environment in to which the Orange River flows.

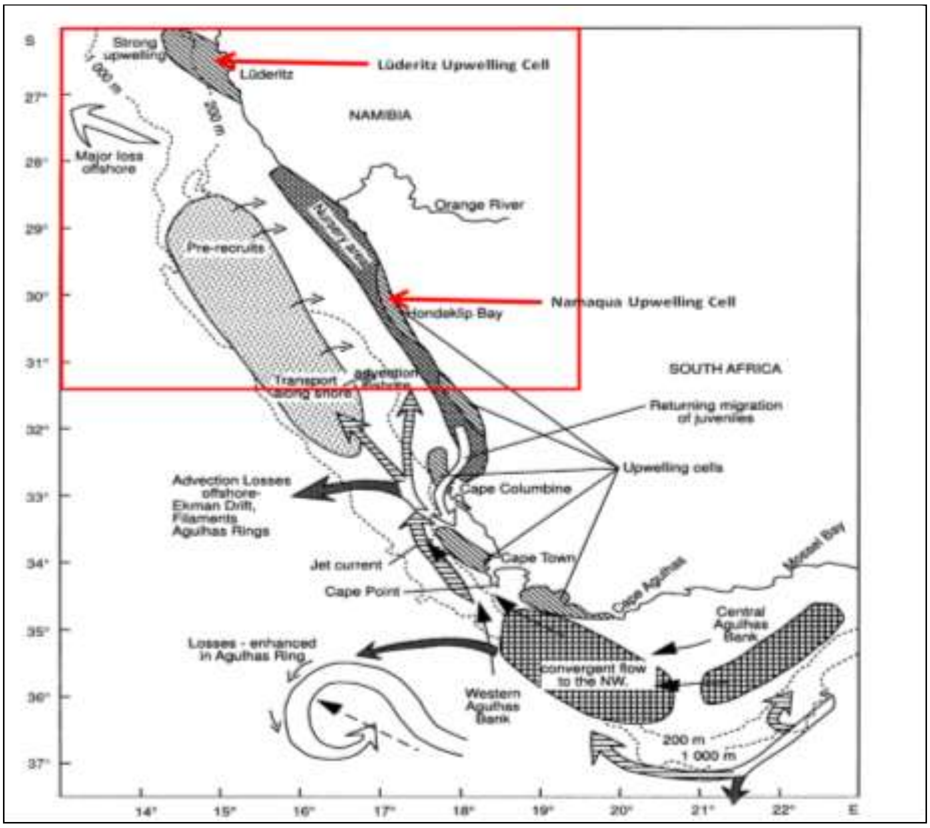


Figure 6. Physical and biological dynamics of the entire BUS along the coasts of South Africa and Namibia (after Hutchings et al., 2002). The red box indicates the area of interest, where the Orange River mouth is shown to lie between the two major upwelling cells of Lüderitz to the north and Namaqua to the south

The BUS forms the eastern boundary upwelling regime of the South Atlantic Subtropical Gyre. BUS is naturally divided into a northern and a southern upwelling sub-system, with the northern sub-system exhibiting perennial upwelling and the southern sub-system seasonal (spring and summer) upwelling (Hutchings et al., 2009).

The upwelling of the southern BUS is driven by spring and summer south-easterly wind regimes, while the northern BUS is driven by the north-eastward drift of the Benguela Current (Duncombe Rae, 2005; Monteiro, 2009). These physical mechanisms are responsible for the slope to shelf upwelling flux which brings mainly South Atlantic Central Water from slope to the outer shelf and from the outer shelf to the inner shelf before upwelling to the surface layers commences (Monteiro, 2009; Waldron et al., 2009). This is especially true for the marine region into which the Orange River flows, where the shelf is more than 150 km wide from the coastline to the edge of the continental shelf.

For the purposes of this study, the geographical boundaries of the marine study area were selected as a box approximately 100 km north and south of the river mouth, and 200 km seawards (Figure 7). This selection was based on the possible sphere of influence of the Orange River plume. The white line on the seaward side of the map demarcates the edge of the continental shelf. Station locations for the South African Data Centre for Oceanography (SADCO) data extraction are indicated in red, while the Department of Agriculture, Forestry and Fisheries (DAFF) sampling locations are indicated in green

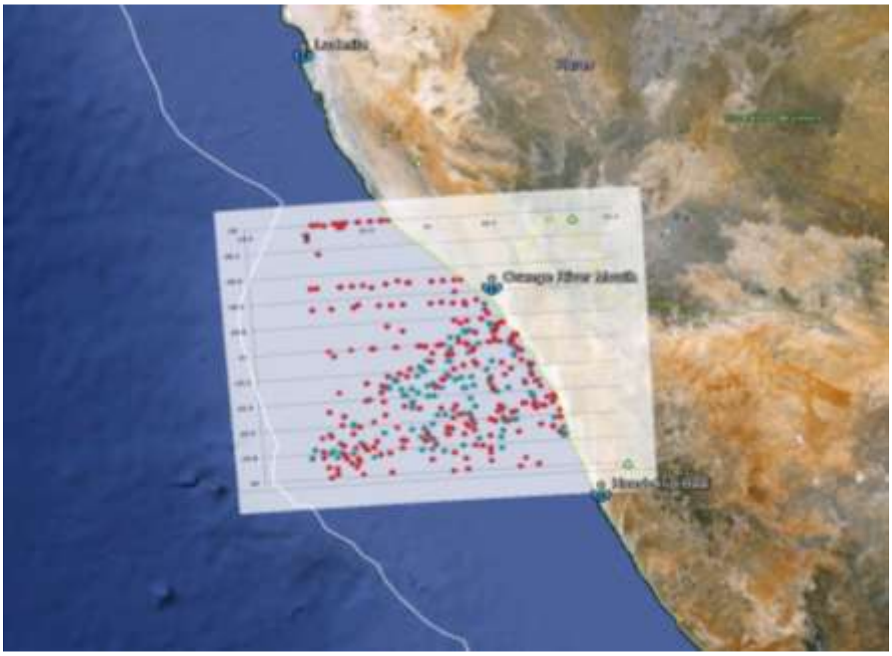


Figure 7. The study area in the marine environment selected for this study.

Physical and biogeochemical data were extracted for the study area from SADCO (http://sadco.csir.co.za/) for the period 1970 to 2000 (red dots in Figure 7). In addition data obtained from South Africa's DAFF collected during summer for the period 1996 to 2011 were also used (green dots in Figure 7).

4.1.1 *Temperature and salinity*

The temperature and salinity data from SADCO and DAFF are presented in Figure 8. The temperature and salinity data allowed for the differentiation of two distinct water masses, i. e., Modified Upwelled Water (MUW) and South Atlantic Central Water (SACW), according to the definitions of Duncombe Rae (2005). The SACW, which forms the main source of high nutrient upwelling waters, was found with temperatures between 6 and 15° C and salinity between 34,50 to 35,40 psu. MUW, which has a lower density and found on top of the SACW, has a temperature range of between 10 to 20° C, and salinity between 34,70 to 35,40 psu. These temperatures and salinities found for the SADCO and DAFF data differ slightly from those described by Duncombe Rae (2005) (see below in Figure 8).

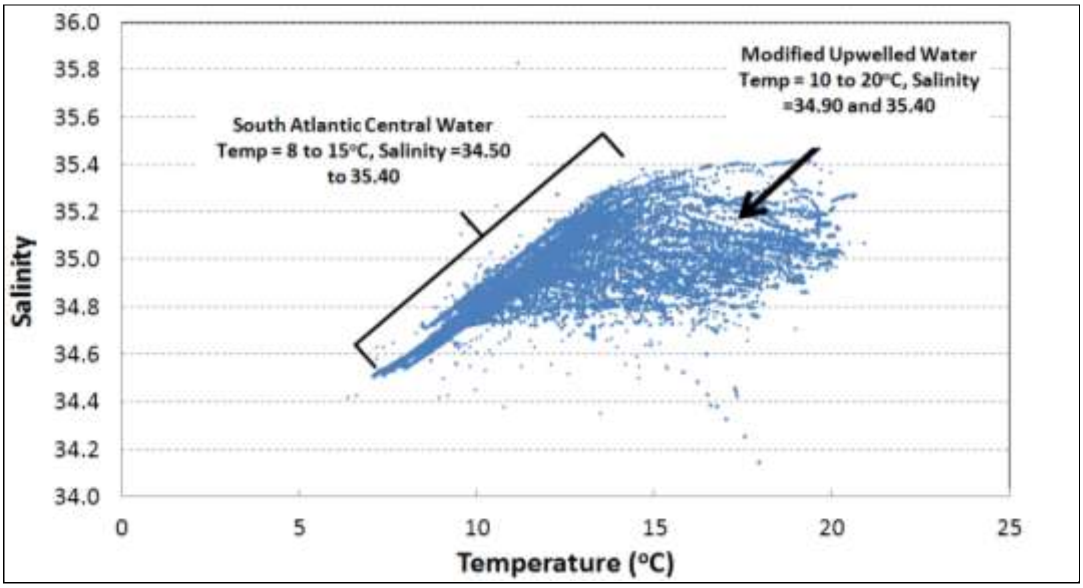


Figure 8. Salinity (psu) and temperature property plot from data collected in the area adjacent to the Orange River collected over the period 1970 to 2011 displaying the SACW and the MUW. These water masses were identified using the definitions of Duncombe Rae (2005).

Duncombe Rae (2005) describes a hydrographic survey line extending from the Orange River mouth directly offshore which allowed for temperature-salinity comparisons, as well as for a depth profile of the water masses described above (Figure 9).

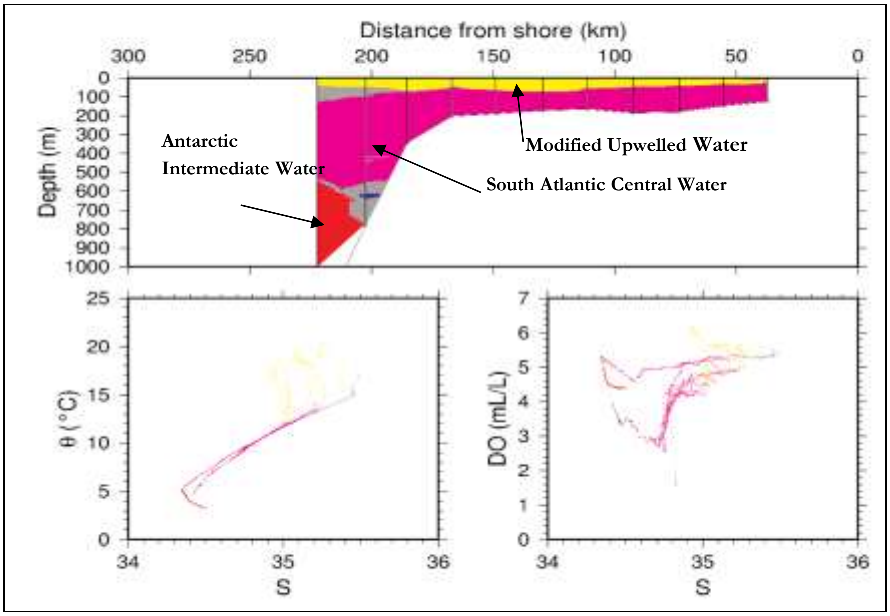
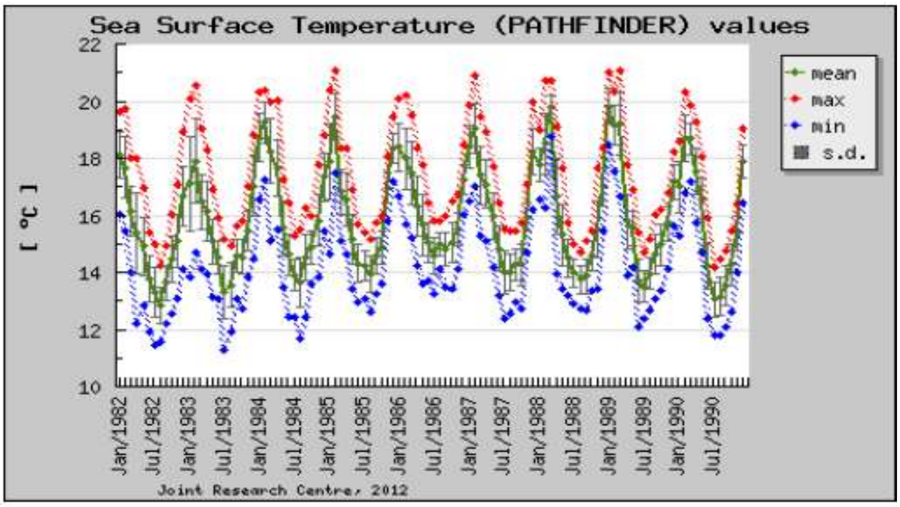


Figure 9. The graphic shows the oceanic water masses with distinct temperature and salinity characteristic found along a survey line directly in front of the Orange River mouth. The yellow indicates MUW, the pink indicates SACW and the red indicates Antarctic Intermediate Waters (after Duncombe Rae, 2005).

Duncombe Rae (2005) found SACW as having a temperature range of 6 to 14° C and a salinity of 34,70 to 35,65 psu. The SACW is derived from the adjacent Cape Basin and can be found on the Benguela continental shelf at the Orange River mouth between 50 m and 200 m depth, and between 200 m and 600 m depth, 200 km offshore (Duncombe Rae, 2005) (see Figure 9). The SACW is nestled in between Modified Upwelled Water on top and Antarctic Intermediate Water, below (Figure 10). Modified Upwelled Water has a temperature of between 12 and 21° C and a salinity of between 34,90 and 35,50. The Antarctic Intermediate Water found below 600 m along the shelf slope has a temperature of between 3 and 5° C and a salinity of between 34,35 and 34,50 (Figure 9). These three water masses describe the physical environment into which the Orange River deposits its freshwater and suspended particulate matter

Infrared satellite data obtained for the study area from the Pathfinder Data Set (1982 – 2000) (Casey, 2010), consisting of 4 km x monthly sea surface temperature (SST; in °C) show a strong seasonal signal for surface waters of the region (Figure 10). Summer SSTs are markedly higher (19 – 21 °C) compared with winter SSTs (11 – 14 °C) (Figure 10). This seasonal SST signal is also observed in the SADCO dataset (not shown) but is not as strongly portrayed as in the satellite SSTs.

At first this strong seasonal signal in SSTs appears to be counter-intuitive since, the BUS is highly active during spring-summer, thus colder SSTs would be expected during spring and summer. However, the Benguela upwelling is concentrated in well-defined upwelling regions (cells), where cooler waters are observed in spring and summer, compared to the rest of the region (Weeks et al., 2006). These intense upwelling regions fall outside the region for which satellite data were extracted for Figure 10.



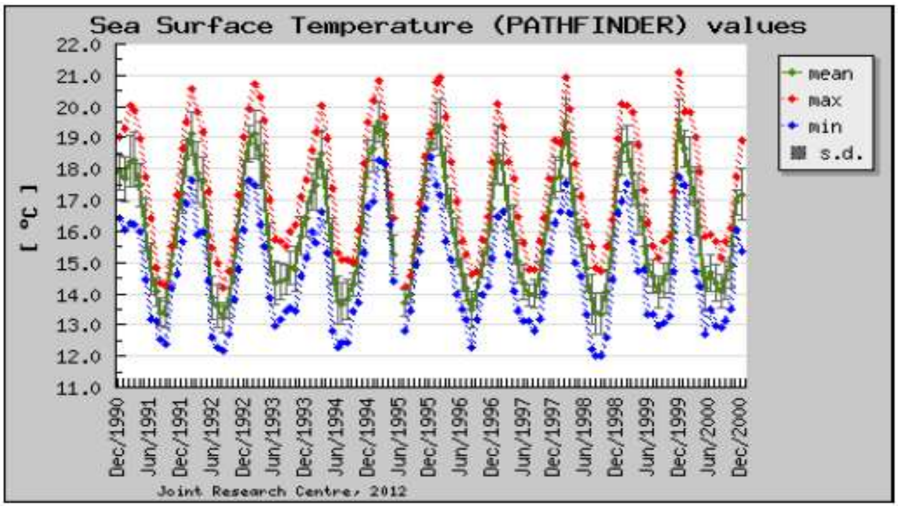


Figure 10. SST (°C) for the study area derived from the Pathfinder satellite for the period 1982 to 2000 (Source: European Union's Joint Research Centre's Global Marine Information System http://gmis.jrc.ec.europa.eu)

Weeks et al. (2006) using SST satellite data for the period 1998 to 2003 clearly show that outside the major upwelling cells of the southern Benguela; incoming solar radiation (insolation) has a dominant effect. They found that during December to February SSTs to be the highest, while during June to August SSTs to be the coolest in the continental shelf regions away from major upwelling cells. Thus the seasonal signal observed in the satellite data in Figure 10 can be ascribed mainly to solar insolation differentials between seasons.

4.1.2 *Dissolved oxygen*

Dissolved oxygen data collected for the period 1970 to 2011 (SADCO and DAFF data) is presented in Figure 11. Dissolved oxygen (mg/ℓ) varied between a low of close to zero to a high of 14 mg/ℓ, over the temperature range 6 to 21° C. The higher dissolved oxygen concentrations of between 6,0 to 14,0 mg/ℓ are characteristic of MUW found in the upper 50 m (Figure 11). The reason for the high dissolved oxygen concentrations could be due to wind induced aeration, as well as the production of oxygen due to photosynthesis. The lower dissolved oxygen concentration of <6 mg/ℓ exhibits a downward peak towards 10° C, which can be ascribed to SACW, which lies below the MUW. The lower dissolved oxygen content of the SACW is due to dissolved oxygen depletion via organic matter remineralisation. The latter is confirmed by the dissolved oxygen minimum found between 50 and 150 m depth, a depth range occupied by SACW (Duncombe Rae, 2005).

The dissolved oxygen data obtained from SADCO and DAFF appears to higher than what were found for the same region by Duncombe Rae (2005). According to Duncombe Rae (2005) upper MUW (0 to 50 m) contained dissolved oxygen concentrations of between 6,29 to 9,43 mg/ℓ, and SACW contained dissolved oxygen concentrations of between 2,0 to 7,72 mg/ℓ. The apparent differences between the dissolved oxygen concentrations from SADCO and DAFF are not immediately apparent.

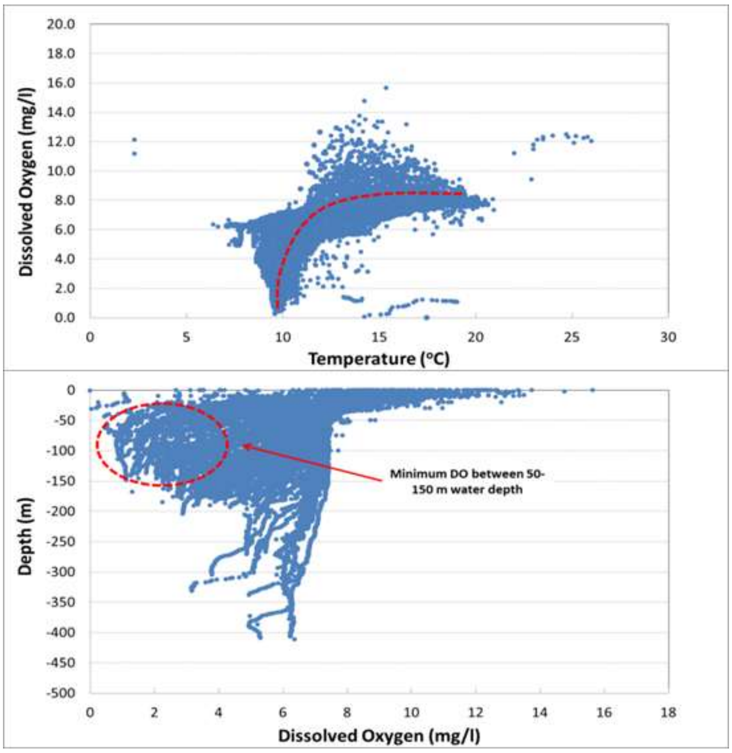


Figure 11. Relationship between temperature and dissolved oxygen in the near-shore study area adjacent to the Orange River measured over the period 1970 to 2011, as well as variation in dissolved oxygen with water depth over the same period.

4.1.3 Dissolved inorganic nutrients

Dissolved inorganic nutrient concentrations from SADCO are presented in Figure 12. Nitrate concentrations varied from near 0 to 420 µg/ℓ, phosphate concentrations varied from near 0 to 100 µg/ℓ, and silicate concentrations varied from near 0 to 1,120 µg/ℓ (Figure 12).

The maximum dissolved nutrient concentrations appear to reflect previous measurements made for the southern BUS, e. g., Brown and Hutchings (1987) found maximum nitrate concentrations of 375 µg/ℓ (26,8 mmol/m^3), maximum phosphate concentrations of 85 µg/ℓ (2,74 mmol/m^3), and maximum silicate concentrations of 703 µg/ℓ (25,1 mmol/m^3). The dissolved silicate concentrations might be considered high, as both Brown and Hutchings (1987) and Pitcher et al., (1998) obtained silicate concentrations below 840 µg/ℓ (30 mmol/m^3) for the southern BUS. Dissolved nutrient concentrations close to zero are documented by Pitcher et al. (1998) for the southern BUS.

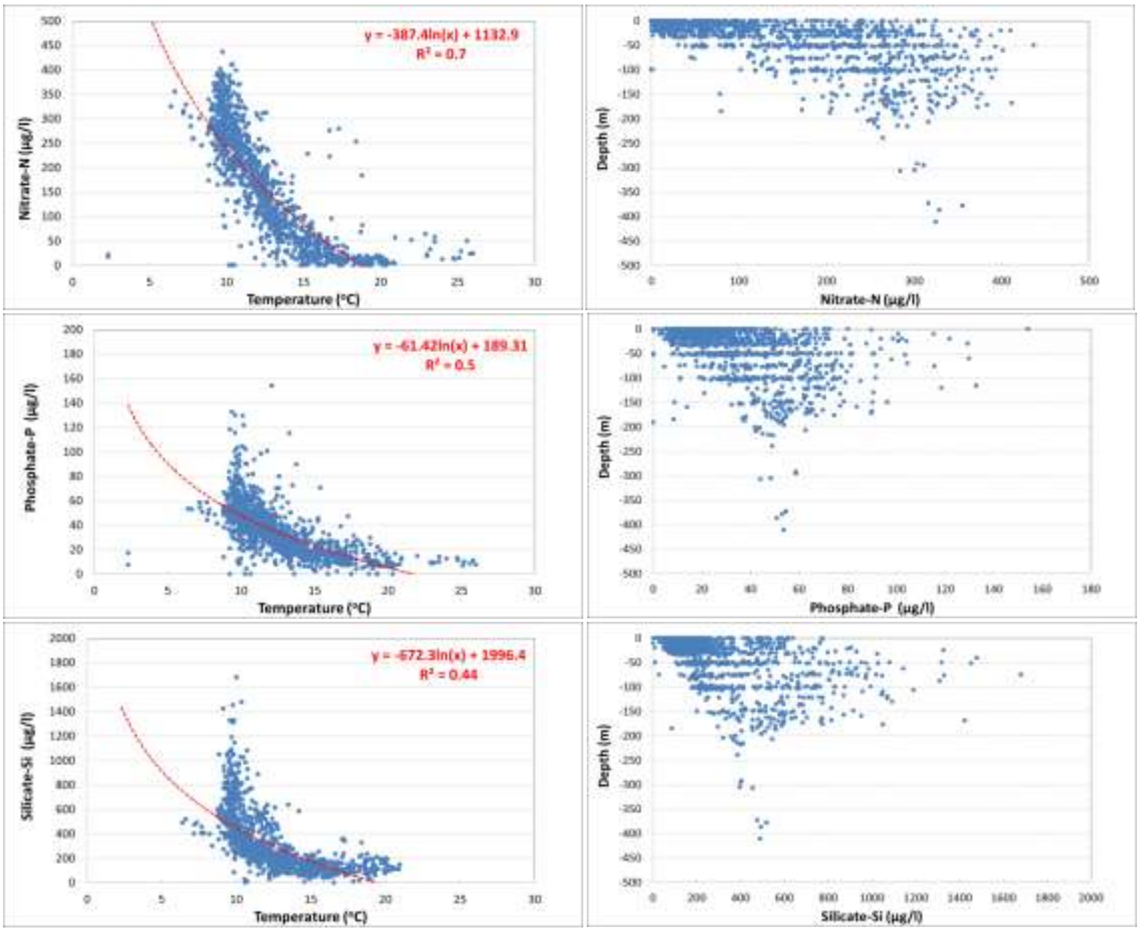


Figure 12. Relationship between temperature and dissolved nitrate, phosphate and silicate concentrations in the near-shore study area adjacent to the Orange River (see Figure 7) measured over the period 1996 to 2000.

To convert units for nitrate from µg-at N/ℓ or mmol/m³ to µg N/ℓ : Nitrate(as µg N/ℓ) = Nitrate(as µg-at N/ℓ or mmol/m³) x 14

To convert units for phosphate from µg-at P/ℓ or mmol/m³ to µg P/ℓ : Phosphate(as µg P/ℓ) = Phosphate(as µg-at P/ℓ or mmol/m³) x 31

To convert units for silicate from µg-at Si/ℓ or mmol/m³ to µg Si/ℓ : Silicate(as µg Si/ℓ) = Silicate(as µg-at Si/ℓ or mmol/m³) x 28

The relationships between dissolved nutrients and temperature exhibit logarithmic curvatures, with lower dissolved nutrient concentrations at higher temperatures and higher dissolved nutrient concentrations for lower temperatures. These are the expected relationships between dissolved nutrients and temperature, also found by Pitcher et al. (1996) and Pitcher et al. (1998) for the southern BUS. The higher dissolved nutrient concentrations between 8 to 12° C reflects the high dissolved nutrient content of the SAACW which is the main source of dissolved nutrients for the surface waters. The lower nutrient concentrations of the surface waters reflect the depletion of the dissolved nutrient due active photosynthesis in the euphotic zone. Nutrient data showed no clear relationship with depth (Figure 12).

In their study Holzwarth and others (2007) attributed the influence of river inflow on phytoplankton communities to be restricted to a handful of diatom species that appear more abundant off the mouth of the Orange River and the Kuenene River (Holzwarth et al., 2007). These authors speculate that this may be due to reduced salinity in the surface waters and/or other river-specific influences (e.g. nutrient, trace metals and sediment inputs). Could it be dissolved reactive silicate, a key component for diatoms.

4.1.4 Chlorophyll-a

Chlorophyll-a (chl-a) concentrations measured in the study area over the period 1970 to 2000 (SADCO data) is presented in Figure 13. The chlorophyll-a data plotted against temperature shows concentrations between 0 and 18,0 µg/ℓ (or mg/m³) over the temperature range 9 to 21° C. These chl-a concentrations are in agreement with previous measurements made for the southern BUS in the euphotic zone, e. g., Pitcher et al. (1996), Brown and Hutchings (1987) and Weeks et al. (2006) found chl-a concentrations of between 0 and 20 µg/ℓ. It is interesting to note that Weeks et al. (2006) also found a similar almost bell-shape pattern in chl-a distribution over a similar temperature range of between 8 to 23° C (see Figure 6 of Weeks et al., 2006). Although they only measured chl-a in the surface, compared to the dataset presented here where depth measurements are included, they attribute the less than 2 µg/ℓ chl-a concentrations to offshore chl-a poor waters. Chl-a concentrations of between 2 and 7 µg/ℓ were attributed to phytoplankton in recently upwelled waters (Weeks et al., 2006). Their highest chl-a concentrations of greater than 7 µg/ℓ coincided with temperatures of between 12,5 to 17,5° C, which is similar to the temperature range in which the highest chl-a concentrations were found for the SADCO dataset.

Sea surface chl-a concentrations were obtained from the MODIS satellite instrument at a resolution of 4 km x monthly for the region of interest (black box, Figure 14). The monthly frontal lines shown in Figure 14 demarcate the 1 mg/m^3 chlorophyll-a concentration boundary. These frontal lines, thus, separate the chlorophyll-poor oceanic waters from the chlorophyll-rich coastal waters. The maps show that although the frontal lines moves to and fro from the edge of the continental shelf, the region of interest is not significantly impacted by the frontal variability. In other words, most of the time the surface concentrations within the open black box (Figure 14) are higher than 1 μg/ℓ.

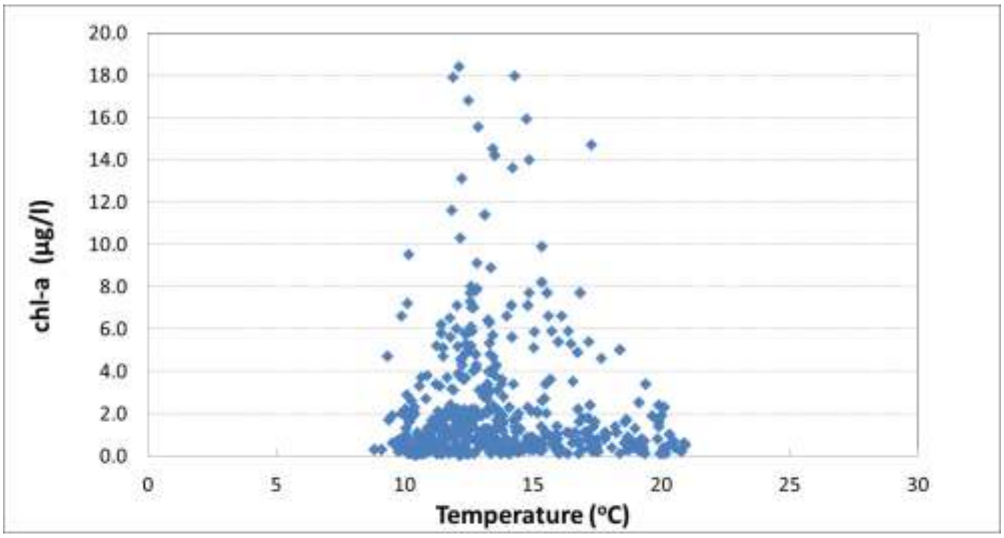


Figure 13. Relationship between temperature and chlorophyll a concentrations (μg/ℓ) in the near-shore study area adjacent to the Orange River measured over the period 1970 to 2000 from SADCO

The gradient in chlorophyll-a concentration within the black box (see inserts in Figure 14) is shown to increase from offshore towards the coastline. The lower chl-a concentration of <2 μg/ℓ (yellow) is characteristic of the offshore oceanic waters (Weeks et al., 2006), while higher concentrations of > 2 μg/ℓ (red) is characteristic of productive coastal waters.

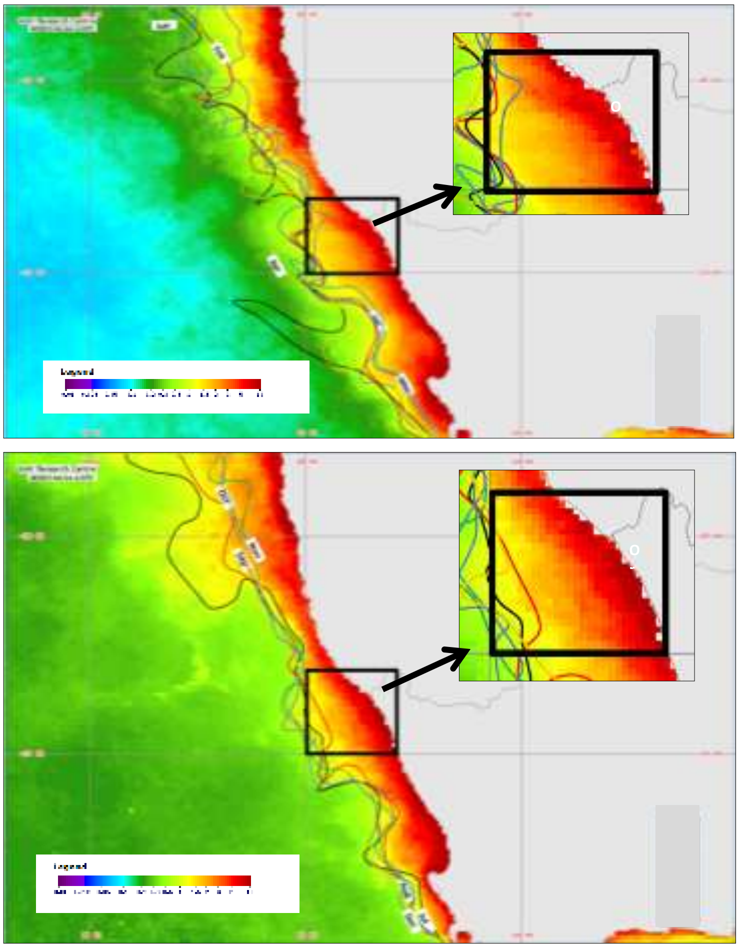


Figure 14. Graphs of sea surface chlorophyll-a concentration (mg/m³, equivalent to µg/ℓ) derived from MODIS satellite instrument. The maps show monthly lines of the chlorophyll-a front at 1 mg/m³ for the region of interest. The top map is for the months January to June and bottom map for July to December.

4.1.5 Turbidity

Turbidity data collected in the study area for the period 1996 to 2011 (data obtained from DAFF) are presented in Figure 15. Turbidity data are plotted against temperature and water depth. The turbidity-temperature plot shows that the turbidity for the small triangular regions where the DAFF stations are located (see Figure 15) just south of the Orange River mouth, has a turbidity range of between 0 and 5,0 NTU. The majority of measurements, however, are found in the range 0 to 1,5 NTU along the entire water column from the surface to the bottom. A sharp peak is found at 10° C with maximum values of 5 NTU, with a few station measuring higher concentrations of up to 6,8 NTU. This higher turbidity values are found in the 50 to 100 m depth range (Figure 15). It is between these depths that the SACW and MUW form a temperature-salinity frontal system (Duncombe Rae, 2005). It is also here where organic and inorganic particles (marine snow) accumulate after settles out from the surface waters. It can thus be assumed that this higher turbidity values at 10° C, and between 50 to 100 m depth are due to the accumulation of marine snow which can consists of senescent phytoplankton cells, zooplankton faecal pellets, and inorganic particles that adhere to sinking organic material.

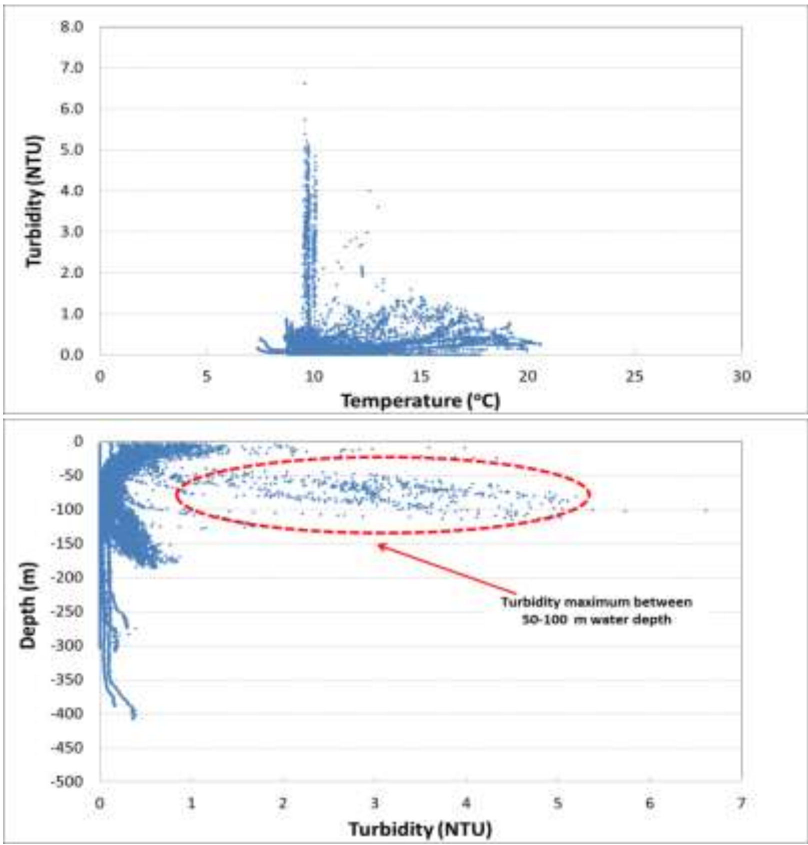


Figure 15. Graphs of sea surface chlorophyll-a concentration (mg/m³, equivalent to µg/ℓ) derived from MODIS satellite instrument. The maps show Relationship between temperature and turbidity in the near-shore study area adjacent to the Orange River measured over the period 1996 to 2011, as well as variation in turbidity with water depth over the same period.

4.2 Sediment processes

4.2.1 Background information and conditions

A site inspection of the estuary, mouth and adjacent beach areas was conducted from 7 to 9 February 2012. The main purpose was to obtain information and field data on estuarine and adjacent coastal hydrodynamics, sediment dynamics and morphology. Spring tides occurred during this period, with the maximum sea tides predicted on 9 February, namely High water level (HWL) at + 2 m CD (16h26) and Low water level (LWL) at + 0.26 m CD (10h14) at Port Nolloth. Wave conditions during the site visit were approximately average to slightly below average, with breaking wave heights (Hbs) off the mouth in the 1,5 m to 3 m range. No sea storms occurred during the field survey period. Winds ranged from about westerly to southwesterly with lighter wind in the mornings ranging from about 3 to 8 knots, and stronger winds in the afternoons in the order of 20 knots.

4.2.2 *Coastal sediment characteristics*

Exposed sandy shores consist of coupled surf-zone, beach and dune systems, which together form the littoral active zone of sand transport (Short & Hesp, 1985). The nature of these zones is primarily dependent on various physical parameters that influence the rate of sand transport: wave energy, sediment particle size and beach slope, which in turn play a dominant role in determining the composition and abundance of faunal communities that inhabit beaches (Short and Wright, 1983). Using a combination of the physical parameters, beaches are classified into low, moderate and high wave-energy environments, each with specific beach characteristics. These beach habitats are occupied by specially adapted beach-dwelling organisms living either on (epifauna), or in (infauna) the unconsolidated superficial layers of the sediment, usually to a depth limit of ~30 cm.

Wright et al. (1982) have combined these wave parameters and sediment characteristics into an index, the "dimensionless fall velocity", Ω (also referred to as Dean's parameter), which incorporates wave height and period, as well as sand grain size, to distinguish between different beach morphodynamic types. Microtidal beaches (beaches with a tidal range of 2 m or less), which typify those on the southern African West Coast, can be classified as dissipative, intermediate or reflective beaches. Dissipative beaches (Ω>6) are characterised by fine sand, high waves and flat intertidal beach gradients. Wave energy is generally dissipated in the surf zone, so that the conditions experienced in the intertidal area are fairly calm. Such beaches harbour the richest intertidal faunal communities (McArdle and McLachlan 1991, 1992; McLachlan et al. 1993; Borzone et al., 1996). Reflective beaches (Ω<2), at the other extreme, are coarse grained (>500 μm sand) with steep intertidal beach faces. The shortened surf-zone results in most of the wave energy being dissipated in the intertidal area, and the waves break directly on the shore. This causes a high turnover of sand and a harsh intertidal climate, with resultant poor faunal communities (McLachlan, 1996; Janssen & Mulder, 2005). Intermediate beach conditions occur between Ω = 2 and 6.

There is, however, considerable small-scale spatial and temporal variability in wave energy, beach slope and sand particle size. Cyclical variations occur in response to the seasonal supply of sediment to the marine environment and seasonal changes in the nearshore wave regime. As part of a natural sedimentary cycle, the southern African coastline is subject to gradual accumulation of sand deposits during the constructive (low swell) summer season. The beaches subsequently retreat as a result of erosive waves during winter. Superimposed on the seasonal pattern are bi-weekly, daily, and storm-associated sand movements, when temporary reversal in the direction of sediment transport occurs in response to short-term changes in wave conditions. The frequency and duration of sand inundation resulting from such events are highly dependent on local topography, and the northward littoral drift characterising the coastline will typically result in sediment accumulating on the southern sides of rocky outcrops or promontories (Daly & Mathieson, 1977; D'Antonio, 1986).

Superimposed on the usual seasonal and longer-term inter-annual changes, there are high-frequency fluctuations resulting from daily changes in sea conditions, spring/neap tidal cycles and the semi-diurnal tidal cycle. Spatial variability results from differences in wave exposure, height on the shore and water depth, and the intensity of perturbation caused by suspended sediments is largely determined by the rates of sediment movement and deposition, the characteristics of the sedimenting material, and the degree and duration of inundation.

Storm events, other natural processes and human interference can thus alter the physical characteristics of the beaches, resulting in temporary or permanent alterations in faunal communities (Brown & McLachlan 1994; McLachlan et al., 1994; Defeo & Alava, 1995; Alonso et al., 2002; Borges et al., 2002; Brown & McLachlan, 2002; Gomez-Pina et al., 2002). Such changes may alter the manner in which beaches function as an interface between the marine and terrestrial environments, either in terms of their physical behaviour, or their role in nutrient cycling (McLachlan 1980; 1988). The magnitude of the impact depends on an interactive balance between the relative sensitivity of particular beaches to physical disturbance, and the degree of anthropogenic disturbance imposed. On the southern African west coast, the geological deposition of diamond-bearing gravels along historic beach terraces at various past sea levels has resulted in extensive stretches of beach being specifically targeted by diamond mining operations, particularly to the north and south of the Orange River mouth. These mining operations have over the past 100 years had a significant impact on the sand particle size and morphodynamics of the affected beaches.

The beaches north of the Orange River mouth are extremely dynamic, being characterised by high wave energy, and narrow and steep beach faces, and often a low-tide step at the low water mark. In the immediate vicinity of the Orange River mouth, beach sediments are well-sorted with a mean particle size of 340 – 370 μm. Further northwards in Mining Area 1, sediments become coarser (460 – 600 μm) as the beaches are located westwards of seawall operations and artificially accreted shorelines, typically using overburden sands that differ substantially from the original beach sediments. As fine sediments are continuously winnowed out of the beaches by the strong wave action, extremely coarse sand typically remains on the low shore, with gravel, pebbles and even boulders being deposited on the beaches opposite eroding seawalls. The mid-shore is characterised by heavy mineral sands, with coarse sand, shell fragments and washed-up kelp indicating the high shore and drift line. A high-tide berm, often as high as mean sea level (MSL)+4 m, is common on the back-beach.

The comparatively harsh characteristics of these beaches are reflected in the number of invertebrate macrofaunal species occupying them. Around the Orange River mouth the beaches boast on average 9,4 species, whereas further north they harbour on average only 6,5 species. In contrast, the Elizabeth Bay and Grossebucht beaches near Lüderitz, which have a mean particle size of 352 μm and 112 μm, have a species richness of 15 and 19 species, respectively (Pulfrich et al., 2012).

Beaches to the south of the river mouth are finer grained with a mean particle size of 280 μm reported for Port Nolloth (Harris, 2013), with particle size for beaches between the Spoeg and

Groen River mouths ranging from 250 – 360 µm (Soares, 2003; Harris, 2013). Corresponding species richness was 14 species and 19 species, respectively.

As the beaches on the southern African West Coast are continuously subjected to substantial natural environmental disturbance (wind, wave and tidal impacts), they and their faunal communities are comparatively robust to such disturbance (Brown & McLachlan 2002).

Figure 16. The beach immediately north of the river mouth (left) and the steep, coarse-grained beaches backed by seawalls characterising the Namibian mining area north of the mouth (right) (Source: Pulfrich et al., 2012).

Beach sediments observed on the shoreline adjacent to the mouth displayed significant variation in characteristics (Figure 17 and 18), ranging from fine sand to coarse sand, but also included some granules, gravels, pebbles and cobbles, as well as seemingly heavy mineral traces.

Figure 17. Examples of beach sediments observed on the shoreline adjacent to the mouth (Source: A Theron, 08/02/2012).

A photograph of the beach sand near the mouth (as depicted in Figure 18*Figure 18*) gives a rough visual indication of the varied nature of the sand and individual grains .

Figure 18. Photograph of beach sediment near mouth (Source: A Theron, 08/02/2012).

The generally linear shoreline adjacent to the mouth has an overall northwest-southeast orientation. The coastline features a sandy foreshore of relatively steep slope (Figure 19a and b), leading onto a flatter sandy backbeach with a medium high dune-ridge running parallel to the shore near the mouth. The surf zone appears to have a sandy bottom and to exhibit a shore parallel bar and trough profile (intersected by some rip channels). The beach sediments near the mouth range from fine to coarse grained sands, while sparse pebbles and cobbles are observed in some areas (Figure 19b). The distinctly darker coloured beach sediments also observed on the shoreline adjacent to the mouth (Figure 19b), are thought to be traces of heavy mineral deposits. This whole shoreline is completely exposed to incident deep sea waves from virtually all seaward sectors (especially from waves propagating from the dominant south-west). To some extent the surf zone sandbar (multi-barred in some places) helps to dissipate wave energy and reduces wave impact on the shoreline (Figure 19c). The beach sand of on average about medium grain size, along with the relatively steep intertidal beach face slope, is indicative of a moderate to high wave regime.

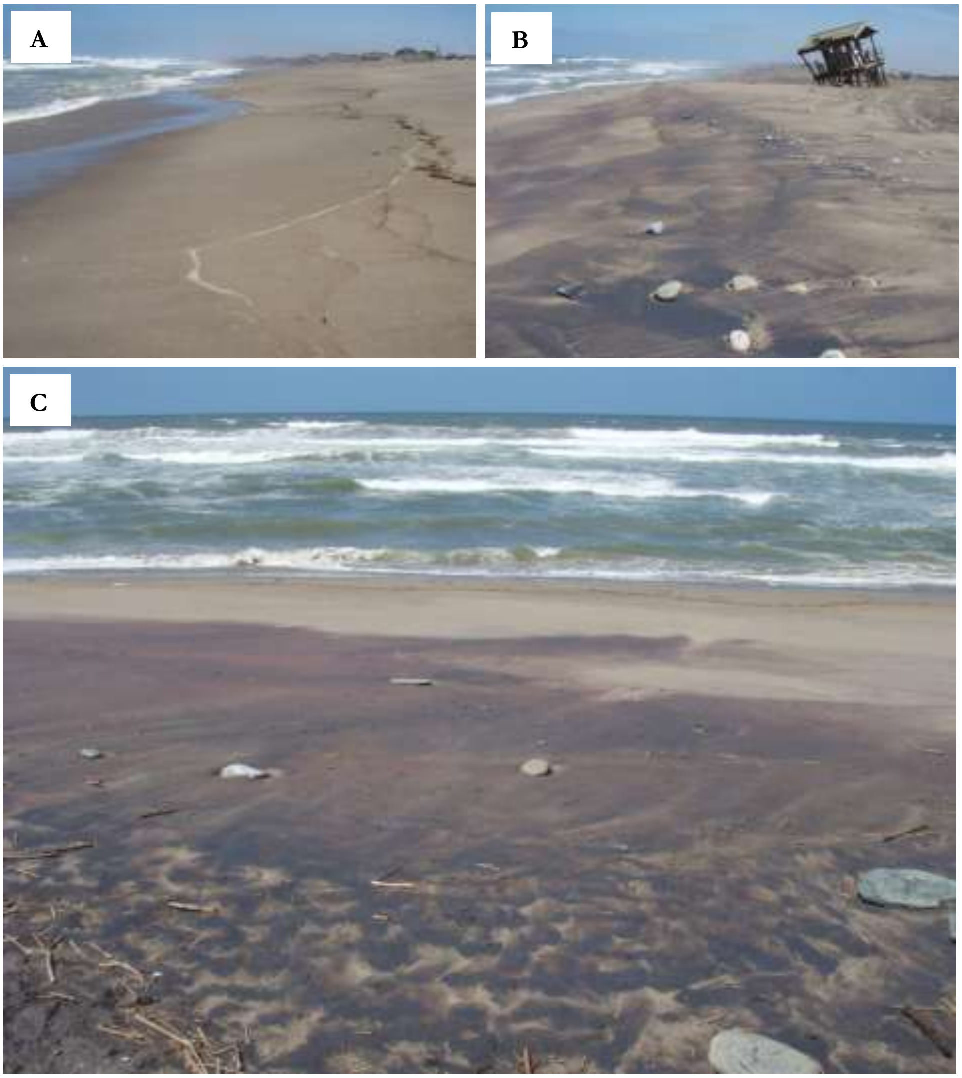


Figure 19. Shoreline adjacent to the Estuary Mouth - a: Steep beach profile; b: Dark coloured beach sediment deposits; c: Illustration of wave energy dissipation on surf-zone sand bar (Source: A Theron, February 2012)

The shoreline to the southeast of the Orange River Mouth is predominantly rocky and subject to diamond mining, as illustrated in Figure 20.

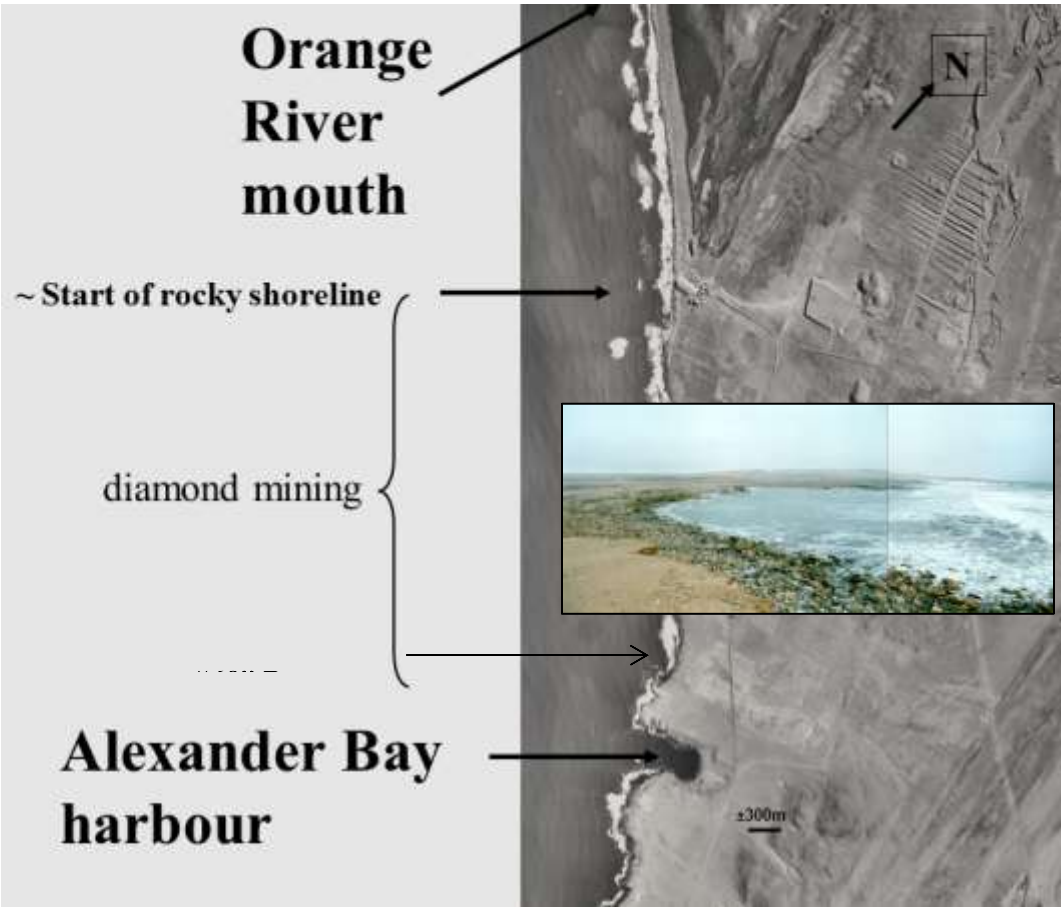


Figure 20. Coastal characteristics to the southeast of the Orange River Estuary Mouth

A summary of marine environmental (metocean) characteristics for the vicinity of the Orange Estuary mouth is as follows:

- sand grain sizes (D_{50} - median): In 68 Bay: 0,97 mm (mining nourishment: 0,26 mm);
- tides: spring tidal range: 1,57 m;
- winds: dominant: S to SE; 6 m/s;
- sediment transport: oblique onshore Aeolian;
- currents: at 30 m depth: < 10 cm/s;
- in surf zone: low to high, NW-bound;
- waves: height (significant): typically 1,5 m – 2 m; mean 2,3 m;
  - storm wave height < 6 m;
  - period (peak): 11 s – 12,6 s; max. < 20 s
  - angle: Usually SSE - SW; few from N sectors;
  - based on refraction study -Rcpwave model;
  - local shore normal breaker angle: +300 to -180 (confirmed by measurements).

- Longshore sediment transport: ~1 Mm^3/year, NW-bound, but the rocky coastline south of the mouth limits the actual transport to a much lower rate.

4.2.3 Sediment dispersal and distribution off the Orange Estuary

Once they arrive in the sea, sediments from the Orange River are deposited in a submarine delta, and are dispersed north and south wards of the river mouth by wave action, longshore drift and subsurface currents. The submarine delta off the mouth of the river extends approximately 26 km seaward of the Orange Estuary and 112 km laterally (Rogers and Rau, 2006). Littoral drift, driven by the south-westerly swells, moves most of the coarse material (sand and gravel) toward of the river mouth, i.e. into Namibia waters (Rogers, 1977), while the weak poleward undercurrent (De Decker, 1970;) carries silt and clay south of the mouth, i.e. into South African waters (Rogers and Bremner, 1991).

The section of the continental shelf opposite the Orange River is termed the 'Orange Shelf', and is the widest part of the Namaqualand shelf. It was formed by high sedimentation rates off the Orange River in the Cretaceous period (Rogers and Rau, 2006). It is up to 100 km wide and 200 m deep. Most of the terrigenous sediments off the west coast of southern Africa are in fact derived from the Orange River, with smaller contributions coming from other rivers in the region (Olifants and Swartlynjies) (Rogers and Bremner, 1991). Some of the material on the shelf is comprised of marine biogenic carbonates that are transported northward by longshore drift (De Decker, 1988; Rogers and Rau, 2006). Much of the fine silt and clay carried south by the inshore undercurrent, accumulates along a mudbelt south of the river mouth (Bremner et al., 1990; Compton and Wiltshire, 2009). The mudbelt extends approximately 500 km southward from the Orange River mouth to St Helena Bay and lies at a depth of 40 to 130 m (Rogers and Rau, 2006). It is at its thickest (35 m) at the mouth of the Orange River (De Decker, 1986). Mean particle size of the sediments in the mudbelt decreases southward due to the reduced influence of the river and the ability of the poleward undercurrent to transport only very fine materials (Rogers and Rau, 2006). The marine biogenic component in the sediments, by contrast, increases southward of the mouth, indicating that there is a significant marine influence on the inner shelf, and that the Namaqualand mudbelt is not primarily derived from the southward transport of terrigenous sediment as was previously thought (Rogers and Rau, 2006).

4.3 River plume behaviour in nearshore marine environment

The Orange River drains into the southern section of the Benguela Current adjacent to the widest part of the continental shelf and at the southern boundary of the Lüderitz-Orange River Cone upwelling cell. This upwelling cell forms the boundary between the northern and southern Benguela Currents and is characterised by strong winds, high turbulence, strong offshore transport and low phytoplankton levels (Hutchings et al., 2009).

The surface currents in the vicinity of the river inflow are on average to the northeast (Figure 21). The north-eastern flow direction is mainly driven by the flux of the cold Benguela Current which forms the eastern limp of the greater South Atlantic Subtropical Gyre. Since the freshwater inflow from the Orange River lies on top of the oceanic surface water due to density differences, it can be assumed that on average the plume of the river will also flow in a north-eastern direction.

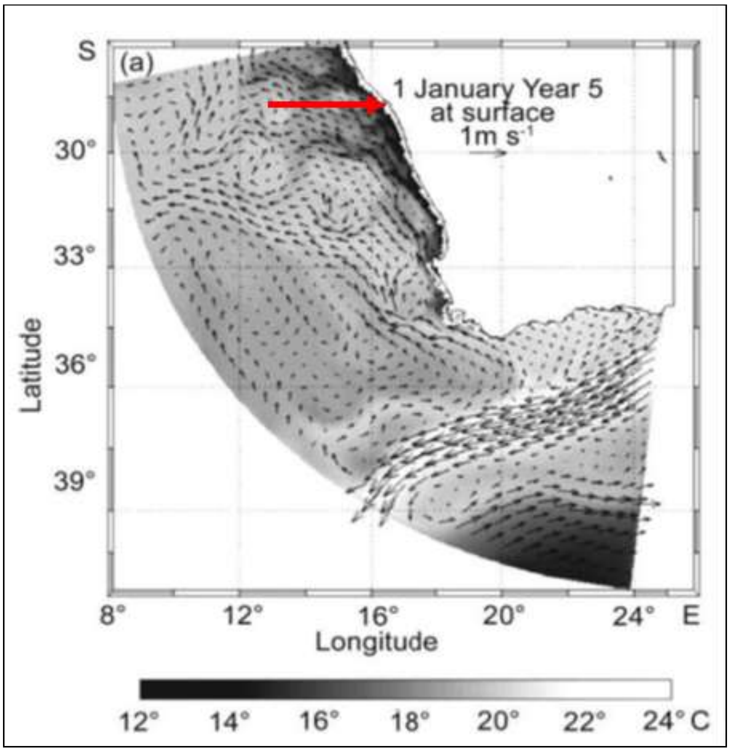


Figure 21. Results from a circulation model shows the stream functions of surface currents and SST for the BUS (Hutching et al., 2009). The red arrow indicates the mouth of the Orange River

The discharge from the Orange Estuary typically forms a plume of buoyant, nutrient-rich freshwater where it drains into the sea, the nature of which, is shaped by the discharge volume and prevailing wind conditions (Shillington et al., 2006; Gan et al., 2009). Buoyant discharge plumes have been known to modify alongshore and cross-shelf upwelling circulation in the upper water column, and can strongly influence near-shore circulation patterns (Gan et al., 2009). During upwelling favourable conditions, the surface-trapped plumes move offshore becoming thinner, thereby strengthening the seaward transport of the plume and the shoreward transport beneath. The actual upwelling intensity is unaffected, however, as there is little to no effect on the water column below 20 m (Gan et al., 2009; Chao and Boicourt, 1986). During down-welling favourable conditions, the freshwater plume typically forms a downwind coastal jet which elongates, accelerates and deepens along the coast (Gan et al., 2009; Chao and Boicourt, 1986). Alongshore currents are enhanced geostrophically along the inshore edge of the plume and weakened along the off-shore edge, due to pressure gradients created by differences in buoyancy between the plume and seawater (Gan et al., 2009).

Under normal circumstances, the flow from the Orange River is so small that it plays a negligible role in near-shore circulation. During severe floods, however, the river plume has exerted some control over coastal circulation patterns (Shillington et al., 2006). In the absence of strong winds the buoyant discharge plume from the 1988 Orange River flood formed an eddy with a diameter of 42 km, and a 10 to 15 km band of coastally-trapped shallow, warm, low-salinity water which travelled up to 200 km southwards of the mouth (Shillington et al., 2006). When south-easterly wind intensified, however, the discharge plume moved north with a deflection to the left caused by the Coriolis Force (Shillington et al., 2006). In terms of run-off, this flood was the largest historic flood on record (24.3 km^3) (Rogers and Rau, 2006), but was by no means exceptional, its return period was estimated to be a 1 in 10 to 15 year event (Swart et al., 1990).

The study area off the Orange Estuary is situated near the centre of the Namaqua bioregion, a cool-temperate bioregion that extends from Sylvia Hill, north of Lüderitz in Namibia, to Cape Columbine in South Africa (Lombard et al., 2004). This bioregion is characterised by high levels of primary production both on the shore (algae) and offshore (phytoplankton. The influence of the Orange River on primary production is most likely small and localised though, due to rapid dilution and distribution of material discharged from the river. It has already been highlighted, for example, that nutrients output from the river is unlikely to influence primary production in the offshore marine environment to any great extent. The influence of outputs from the river on phytoplankton communities appears to be restricted to a handful of diatom species that appear more abundance off the mouth of the Orange River and the Kuenene River (Holzwarth et al., 2007). These authors speculate that this may be due to reduced salinity in the surface waters and/or other river-specific influences (e.g. localised nutrient, trace metals and sediment inputs).

During 4/5 February 2011 biogeochemical data were collected in the vicinity of the Orange River during a flood event (data provided by Department of Environmental Affairs (DEA), South Africa, Hutchings et al., in prep). The positions of the sampling stations are shown in Figure 22). Comparing salinity with depth, a thin surface layer of fresher water overlying saline marine waters is clearly visible (Figure 23). No signal were apparent in the temperature and dissolved oxygen data, however the high turbidity in the Orange River during floods was evident in surface turbidity.

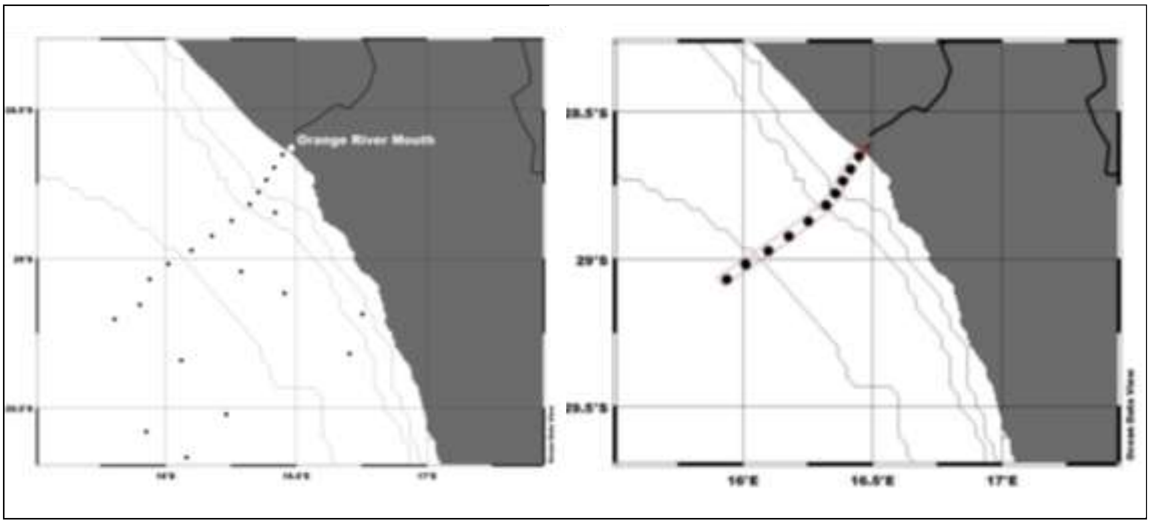


Figure 22. Map showing marine sampling stations during the flood in 4 and 5 February 2011

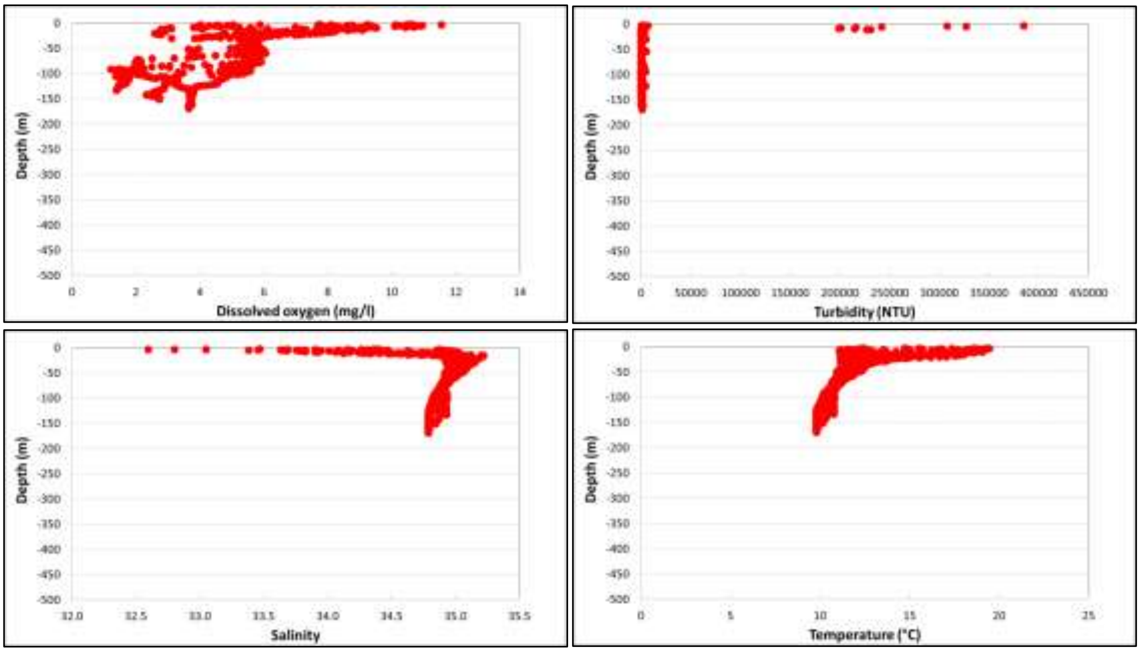


Figure 23. Physico-chemical data collected during a flood event (Feb 2011) in the vicinity of the Orange River relative to water depths: salinity, temperature, dissolved oxygen and turbidity (data provided by DEA, Hutchings et al in press.).

Inorganic nutrient concentrations (nitrate, phosphate and silicate) measure in the near-shore during the February 2011 flood event did not show any correlation with the salinity signal as illustrated in Figure 24 and 25. Phosphate-P did show significantly elevated concentration in surface waters. However, when comparing these against salinity, the highest concentrations were measured in most saline waters. This could not be explained as these high concentrations were outside the natural ranges for PO_4-P in the regions. It was expected to see a fresh water signal in the silicate data (given that silicates is often much higher in freshwater than in marine waters), but this was not apparent from this data set. Nitrates also revealed no freshwater signal, suggesting that the nutrient input from the river during this event was largely masked by the natural nutrient signal in the near-shore environment.

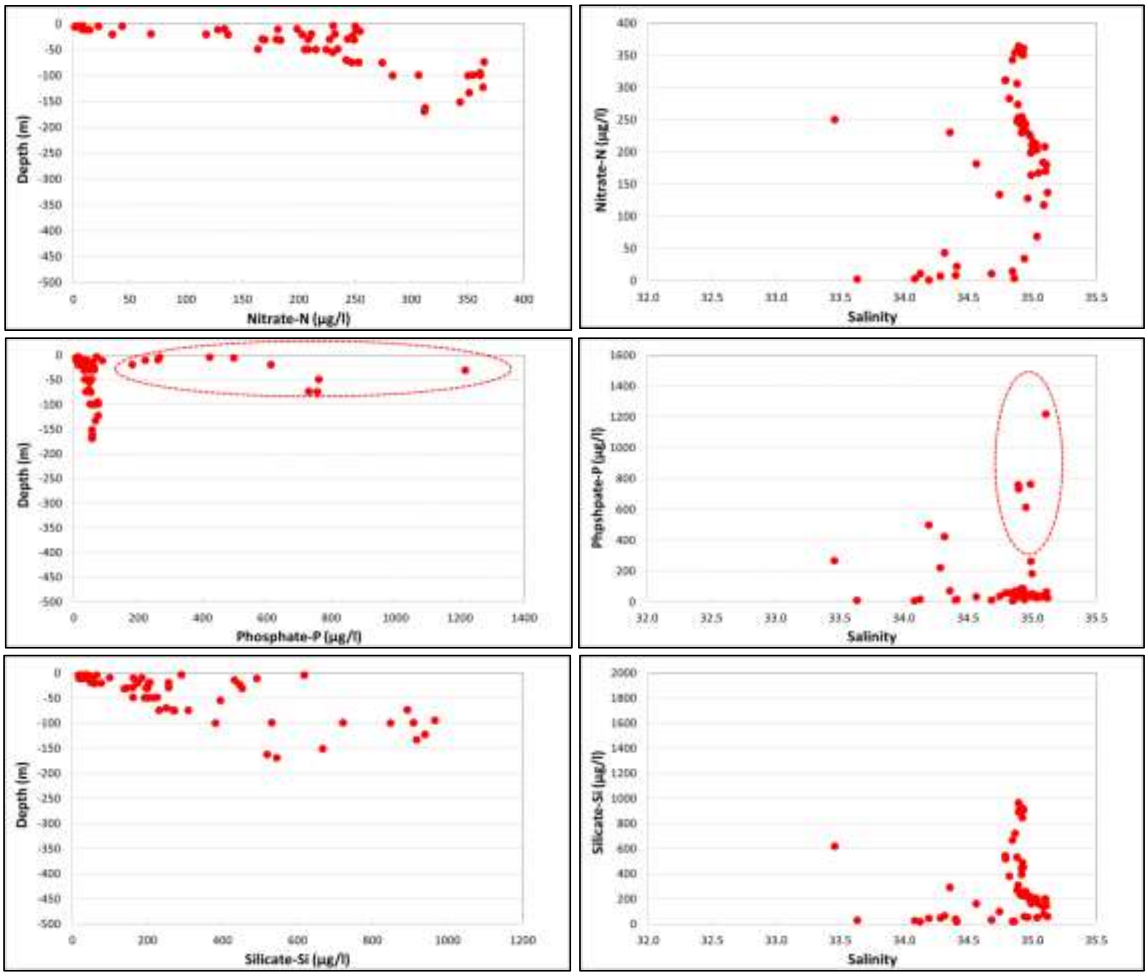


Figure 24. Inorganic nutrient data collected during a flood event (Feb 2011) in the vicinity of the Orange River relative to water depths: nitrate-N, phosphate-P and silicate-Si (data provided by DEA, Hutchings et al in press.)

Backscatter proxy from satellite instrumentation can also be applied to delineate the impact from river plumes in terms of suspended particulate matter loads. The backscattering coefficients from the MODIS data set at a resolution of 4 km x monthly were used. Backscattering is a property of a particulate component in a liquid medium to divert the incident light in the backward direction from its original path. The backscattering coefficient at 490 nm gives a good indication of the concentration of suspended organic and inorganic particles in the water. Away from the influence of terrestrial material, backscattering is a good proxy for suspended particulate matter in coastal waters. Monthly frontal lines delineating the boundary between high suspended particulate matter and low suspended particulate matter for the Benguela continental shelf are presented in Figure 25.

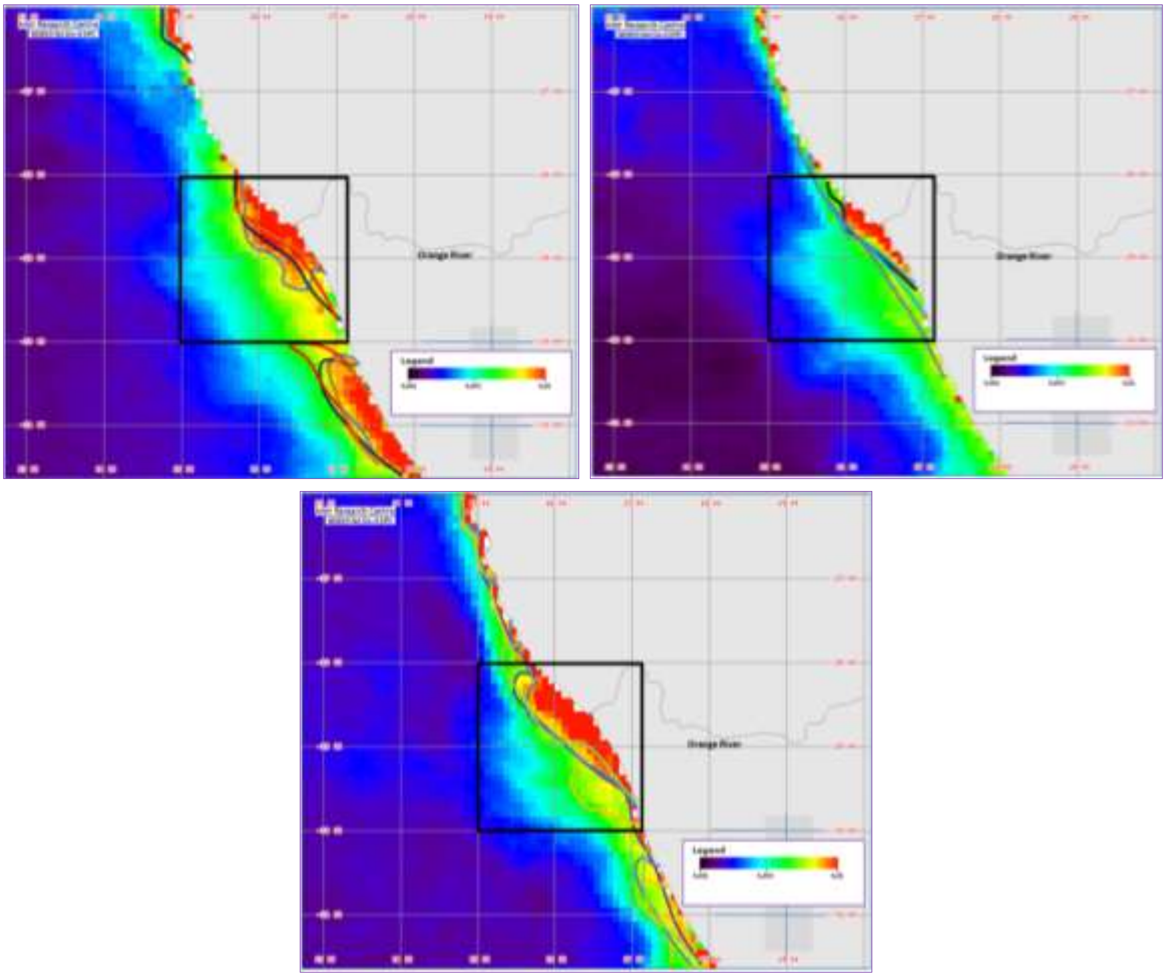


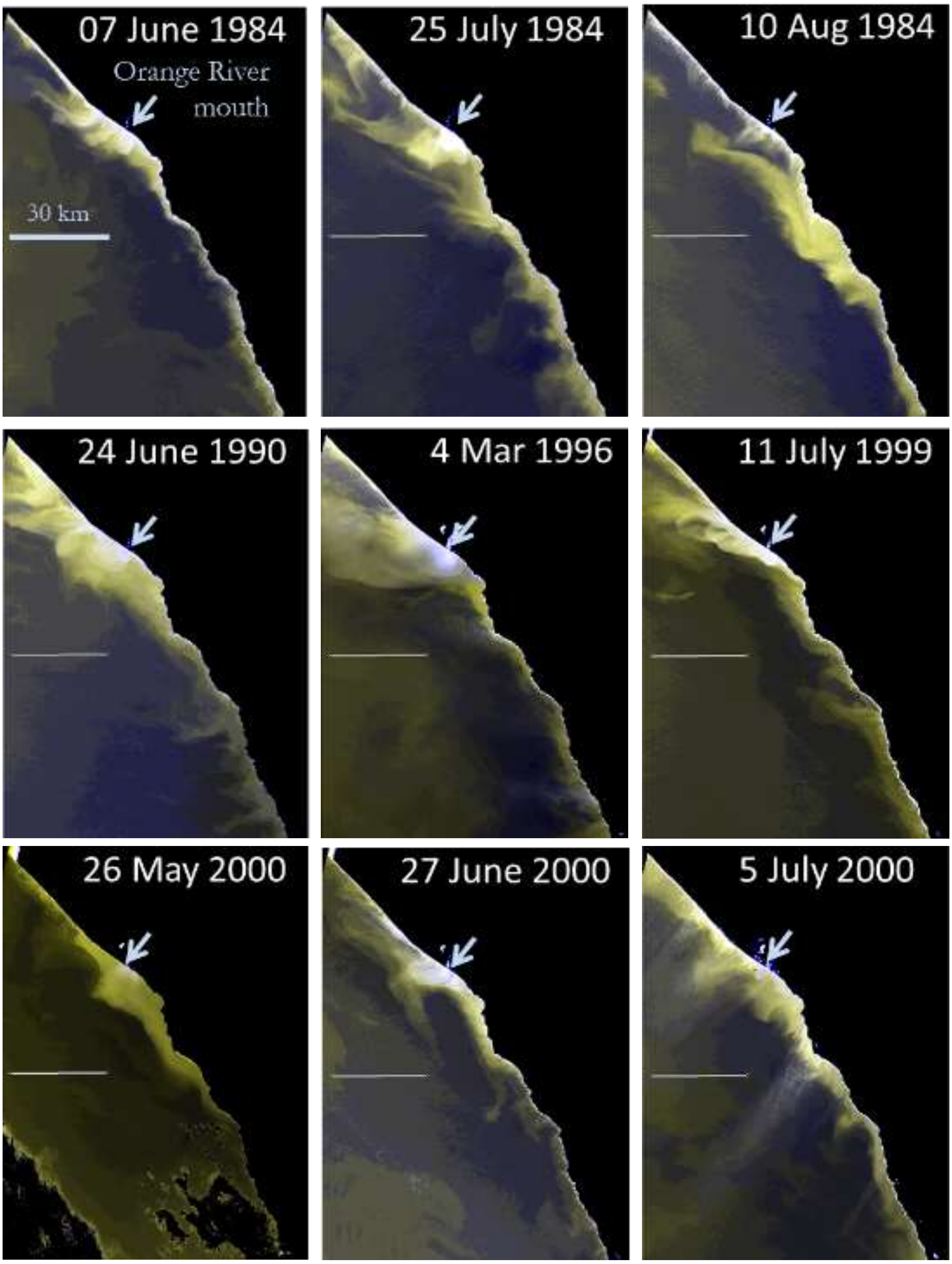
Figure 25. Images of the backscatter proxy for suspended particulate matter derived from MODIS. Each figure contain monthly frontal lines delineating the boundary between high and low particle concentrations, e. g. for the months of January to April (top left), for the months of May to August (top right), and for the months of September to December (bottom)

The frontal lines for the months of January to April (Figure 25) clearly show that the dominant upwelling cells of the BUS can be distinguished using the backscatter proxy. The data for January to April show that Orange River's contribution to the suspended particulate load of BUS is significantly masked by the Namaqua Upwelling Cell directly south of its mouth. The suspended particulate frontal lines of May to August demarcates the influence of the Orange River more clearly, since during these months the contributions of the major upwelling cells to the total concentration of suspended particulate matter is markedly diminished. The decline of suspended particulate matter from the upwelling cells may be due to the decrease in organic matter production during these months when the BUS is least productive. It can be assumed here that during the months of May to August the suspended particulate matter footprint is mainly due to Orange River input to the marine environment, either directly or indirectly. The frontal lines for the months of September to December appear to again show mainly the suspended particulate matter influence from the Orange River, as it is clear from the frontal lines that during this period the contributions from the Lüderitz and Columbine Upwelling Cells are still not fully activated.

The caveat in using backscatter coefficients during the winter months as a proxy for river particulate loading to the marine environment, when the influence of the Lüderitz and Namaqua Upwelling cells are not active, is that during this winter periods intense water column mixing due to strong mid-latitude cyclone activity (e. g. increase wave action and storm surges) can stir up sediment which the Orange River has deposited at its mouth and the surrounding vicinity. It is also during the winter months that the outflow of the Orange River is at its lowest. Thus it should be taken in consideration that the backscatter proxy may not be purely due to suspended particulates from the river plume, but could also be indirectly related to resuspension of river sediment due to wind and wave action cannot be factored out.

For a more detailed spatial analysis, multispectral Landsat 5 TM and Landsat 7 ETM+ images were used. Altogether 18 images of the scene 177 – 80 (WRS-2 Path and Row) were acquired from the US Geological Survey (http://glovis.usgs.gov/) covering the time period from 1984 to 2012. The spatial resolution of 30 x 30 m allows for a more detailed analysis of spatial patterns of sea surface turbidity.

Subsets of all images for the study area are displayed in Figure 26. In most of the images the prevalence of an expressed turbidity cloud is visible. This seems to indicate influence of the river discharge during most of the observation dates. The spatial extent in terms of area, and area influenced along shore and maximum distance of the plume from the shore were assessed visually on each image (Table 4). These values largely coincide with the values mentioned by Shillington et al. (2006), see above. However, the almost permanent visibility contradicts Shillington et al. (2006) who stated that the river flow plays a negligible role in near-shore processes.


07 June 1984
Orange River mouth
30 km
25 July 1984
10 Aug 1984
24 June 1990
4 Mar 1996
11 July 1999
26 May 2000
27 June 2000
5 July 2000

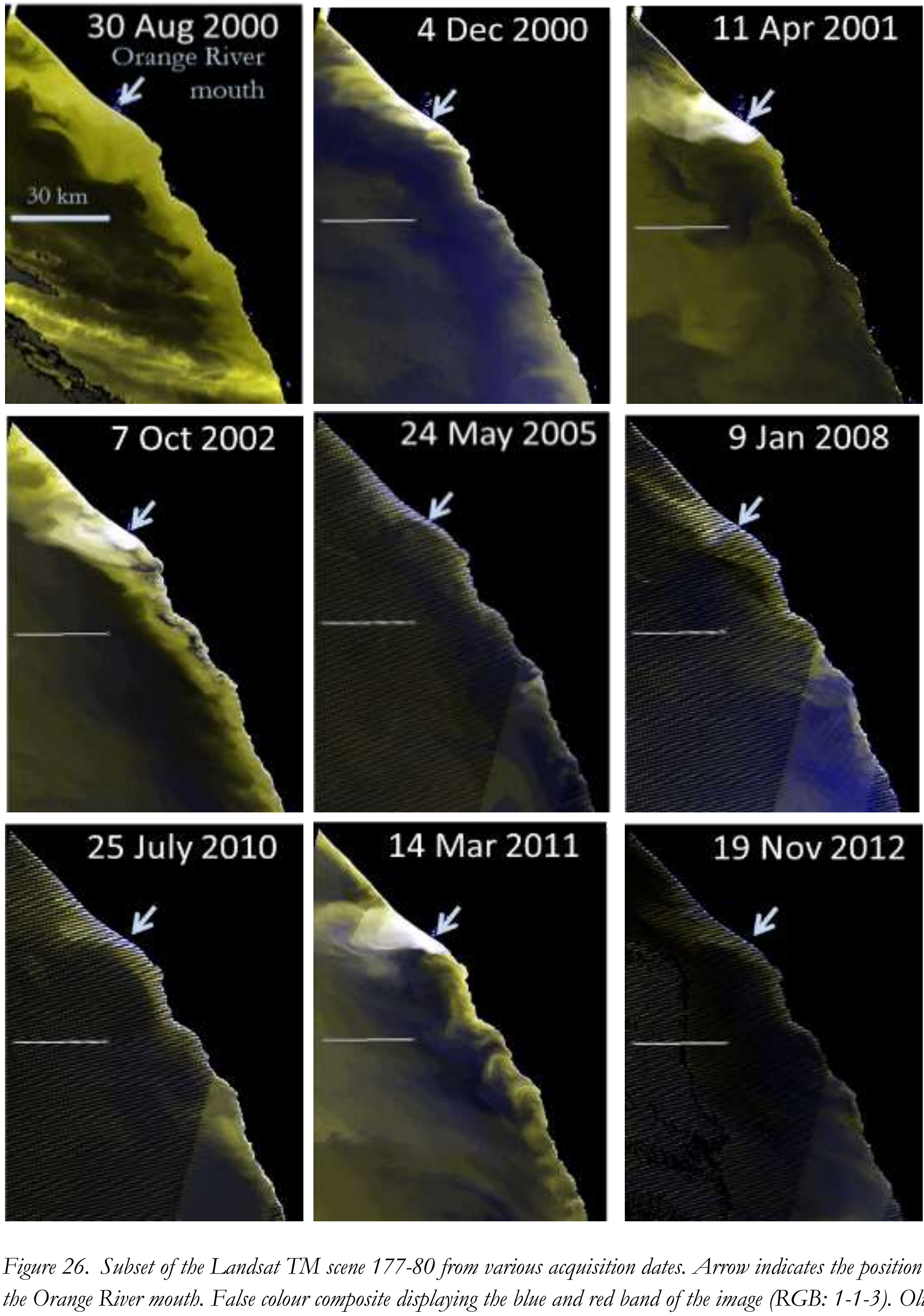


Figure 26. Subset of the Landsat TM scene 177-80 from various acquisition dates. Arrow indicates the position of the Orange River mouth. False colour composite displaying the blue and red band of the image (RGB: 1-1-3). Olive to light violet tones indicate increasing turbidity of the near shore waters. Black: masked land areas and clouds

Table 4. Turbidity plume sizes derived from Landsat (LS) imagery

	LS acquisition date	Plume area (km^2)	Plume extent along shore (km)	Plume extent seaward (km)
1	07/06/1984	297	40	10
2	25/07/1984	184	22	15
3	10/08/1984	67	9	10
4	24/06/1990	670	51	18
5	04/03/1996	1114	60	29
6	11/07/1999	500	44	17
7	26/05/2000	440	47	14
8	27/06/2000	248	17	19
9	05/07/2000	424	33	20
10	30/08/2000	853	43	21
11	04/12/2000	409	40	14
12	11/04/2001	663	35	30
13	07/10/2002	600	45	19
14	24/05/2005	178	31	8
15	09/01/2008	230	32	13
16	25/07/2010	178	30	11
17	14/03/2011	700	26	31
19	19/11/2012	412	38	14

The observed plumes were compared to flow data from the Vioolsdrift as proxy for river discharge. The result indicates no direct relation between the extent of the plume and single flood events (Figure 28). The figure seems to indicate however a cumulative effect of moderate discharges over a longer period on the total plume size (compare the 1995 – 2003 period). This would indicate that the turbidity persists for a longer period than initially anticipated. For example Figure 27 and 28 shows the relationship between river discharge values from the month of the image acquisition (if day was later than 15th of respective month, or the month before the image acquisition date (if the day was before 15th of respective month). None of the 3 measured parameters shows any clear relation to the actual river discharge (black line).

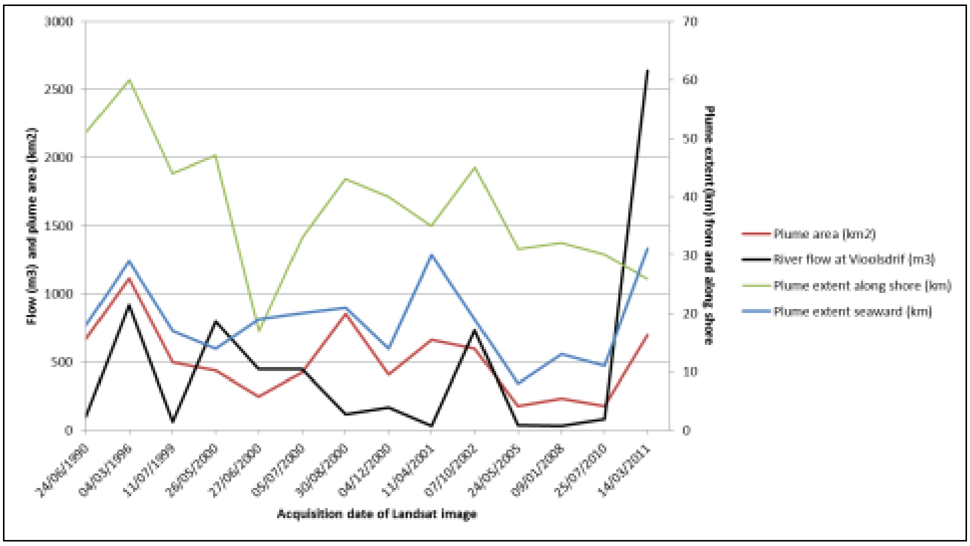


Figure 27. Flood plume area (in km²) and extent along shore (km) in relation to monthly river discharge volumes (flows in m³)

The preliminary analysis of the Landsat images was conducted purely on visual image interpretation of spectrally uncalibrated images. Therefore the delineation does not always follow identical turbidity values/gradients. This approach could not always clearly distinguish between turbidity caused by river discharge and turbidity caused by local upwelling. Also, while in some of the July and August images the influence of upwelling cells seems to dominate over river discharge patterns, this trend is not consistent. These matters would require further investigation beyond the scope of this study.

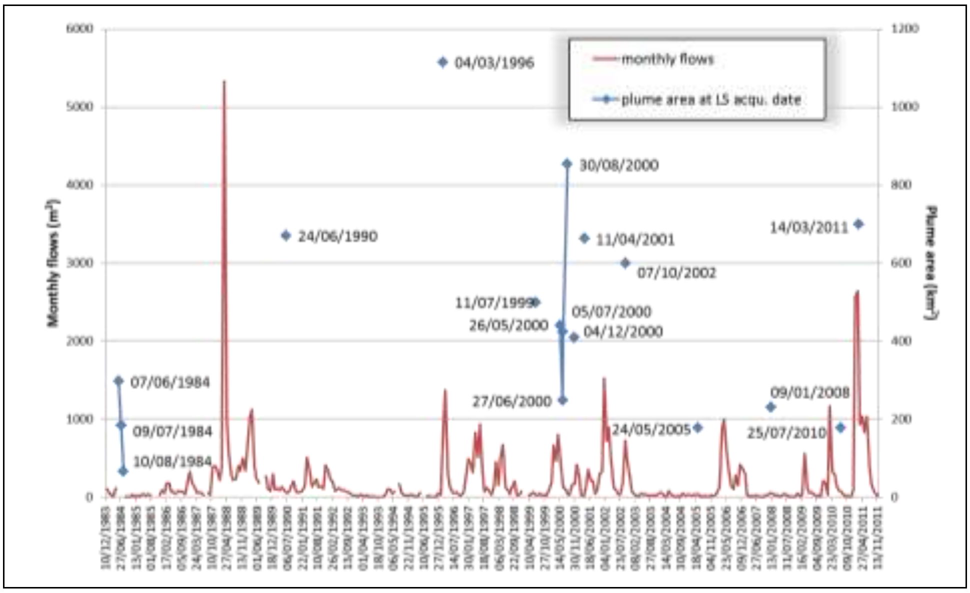


Figure 28. Flood plume extensions (in km²) in relation to monthly river discharge volumes (flows in m³). Dates at dots give respective Landsat acquisition date

5 Important biotic components

Biogeographically, the coastline around Orange Estuary falls into the cool temperate Namaqua Province, which extends from Cape Point to Lüderitz, and is primarily driven by the coastal upwelling characteristic of the region (Emanuel et al., 1992; Lombard et al., 2004).

Communities within marine habitats are largely ubiquitous throughout the southern African West Coast region, being particular only to substrate type or depth zone. Least variation is amongst seaweeds and invertebrates but only marginally more amongst fish. These biological communities consist of many hundreds of species, often displaying considerable temporal and spatial variability (even at small scales).

North of the Orange Estuary the shoreline is predominantly a sandy coast formed by the northward littoral transport of coarse marine sediments. Rocky intertidal habitats are represented only by occasional small rocky outcrops that host benthic communities strongly influenced by sediments (Pulfrich, 2012; Pulfrich et al., 2012). South of the river mouth, the coastline is dominated by rocky shores, with occasional short beaches interspersed between rocky headlands. The deep-water marine ecosystems comprise primarily unconsolidated seabed sediments with subtidal reef habitats being limited to the shallow nearshore regions (<40 m).

5.1 Beach and surfzone

5.1.1 Invertebrates

The ecosystem components typical of the littoral active zone of sandy beaches include the microphytobenthos, terrestrial and semi-terrestrial arthropods, zoobenthos and avifauna, with ichthyofauna being present in the surf-zone and beyond. The benthic invertebrates of soft bottom substrates comprise animals that live either on (epifauna), or burrow within (infauna) unconsolidated surficial layers of the sediment, usually to a depth limit of ~30 cm, and are generally divided into macrofauna (animals >1 mm) and meiofauna (<1 mm). These biological communities are described briefly below, focussing primarily on the macrofauna, of which most area known from southern Namibian beaches (McLachlan & De Ruyck, 1993; McLachlan et al., 1994; Nel et al., 1997; Meyer et al., 1998; Clark et al., 1998; Clark & Nel, 2002; Nel et al., 2003; Clark et al., 2004; Pulfrich, 2004a; Pulfrich et al., 2004; Clark et al., 2005, 2006; Pulfrich & Atkinson, 2007; Pulfrich et al., 2007, 2008, 2010, 2011, 2012). Information on the Namaqualand beach is limited to the research of Soares (2003) and Harris (2013) from the area around the Groen and Spoeg Rivers and Port Nolloth.

Numerous methods of classifying beach zonation have been proposed, based either on physical or biological criteria. The general scheme for intertidal beaches proposed by Branch & Griffiths (1988) is used below, with the description of species present being drawn from reports specifically focussing on the area north of the Orange River mouth.

Supralittoral zone: The supralittoral zone is situated above the high water spring (HWS) tide mark, 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, most often associated with, and feeding on kelp deposited near or on the driftline, but also venturing into the intertidal zone. Terrestrial species include a diverse array of beetles (Insecta: Coleoptera) and kelp flies (Insecta: Diptera) as well as mole crickets (Insecta: Orthoptera). Semi-terrestrial fauna include the giant pillbug *Tylos granulatus* (Crustacea: Isopoda), and the sand hoppers *Talorchestia australis* and *T. quadrispinosa* (Crustacea: Amphipoda). Community composition depends on the nature and extent of wrack, in addition to the physical factors structuring beach communities.

Midlittoral zone: The intertidal zone, also termed the midlittoral zone, has a vertical range of about 2 m. The mid-shore region is characterised by the cirolanid isopods *Pontogeloides latipes* and *Eurydice kensleyi*, the amphipods *Exosphaeroma truncatitelson*, *Paramoera capensis*, *Griffithsia* (*Mandibilophoxus*) *latipes* and *Bathyporeia* sp. and the ribbon worm *Cerebratulus fuscus* (Nemertea) Some mysid shrimps (*Gastrosaccus namibensis* and *G. psammodytes*) occur on the low shore. Due to the extreme exposure, the large mean particle diameter and the lack of fine particles in between, there is a complete absence of molluscs (e.g. *Donax serra* and *Bullia digitalis*) and paucity of polychaete worms (only *Sigambra parva* recorded) on the beaches to the north of the river mouth. The loose, highly penetrable sand does not provide sufficient anchorage, particularly for large molluscs, which bury themselves in the substratum. Clark and Nel (2002) described the meiofauna of the Mining Area 1 beaches was dominated by turbellarian flatworms, followed by polychaetes and oligochaetes. Nematodes and harpacticoid copepods, which are generally the most important groups on finer grained beaches (Brown and McLachlan 1994), collectively only represented 17% of the meiofauna on the Mining Area 1 beaches. Usually also associated with the intertidal zone, and extending into the surf-zone, are benthic microalgae and phytoplankton, which live attached to the sand grains or move actively between them. Both components are usually dominated by diatoms, but on exposed shores, wave action and the vertical mixing it causes, tends to limit their growth on the sand grains (Brown and McLachlan 1994).

The zonation described for intertidal beaches continues into the subtidal regions, where the structure and composition of benthic soft-bottom communities is primarily determined by water depth and sediment grain size. Other factors such as current velocity, organic content, and food abundance, however, also play a role (Snelgrove & Butman 1994; Flach & Thomsen 1998; Ellingsen 2002).

Inner turbulent zone: The inner turbulent zone extends from the Low Water Spring mark to about -2 m depth. The mysid *Gastrosaccus namibensis* (Crustacea: Mysidacea,), the ribbon worm *Cerebratulus fuscus* (Nemertea), and cumaceans are typical of this zone, generally extending partially into the midlittoral above.

Transition zone: The transition zone spans approximately 2 – 5 m depth beyond the inner turbulent zone. Extreme turbulence is experienced in this zone, and as a consequence this zone typically harbours the lowest diversity. Typical fauna of this zone include amphipods and burrowing polychaetes, although the extreme exposure, and large mean particle diameter characterising Mining Area 1 beaches suggest that diversity in this zone will likewise be limited.

Outer turbulent zone: Extends below 5 m depth, where turbulence is significantly decreased and species diversity is again much higher. The zone is typically inhabited by polychaetes, a host of amphipod species, sea pens and the three spot swimming crab *Ovalipes trimaculatus* (Brachyura, Crustacea), although none of the latter two have been recorded from MA1 beaches. A typical feature of the surf-zones of exposed shores is rich accumulations of diatoms, which accumulate at the water surface during the day, but are also dispersed throughout the water column and in the sediments. Under certain conditions these "blooms" can form a semi-stable foam (Brown & McLachlan 1994).

Nearshore zone: species diversity, abundance and biomass generally increase from the shore to 80 m depth, with communities being characterised equally by polychaetes, crustaceans and molluscs.

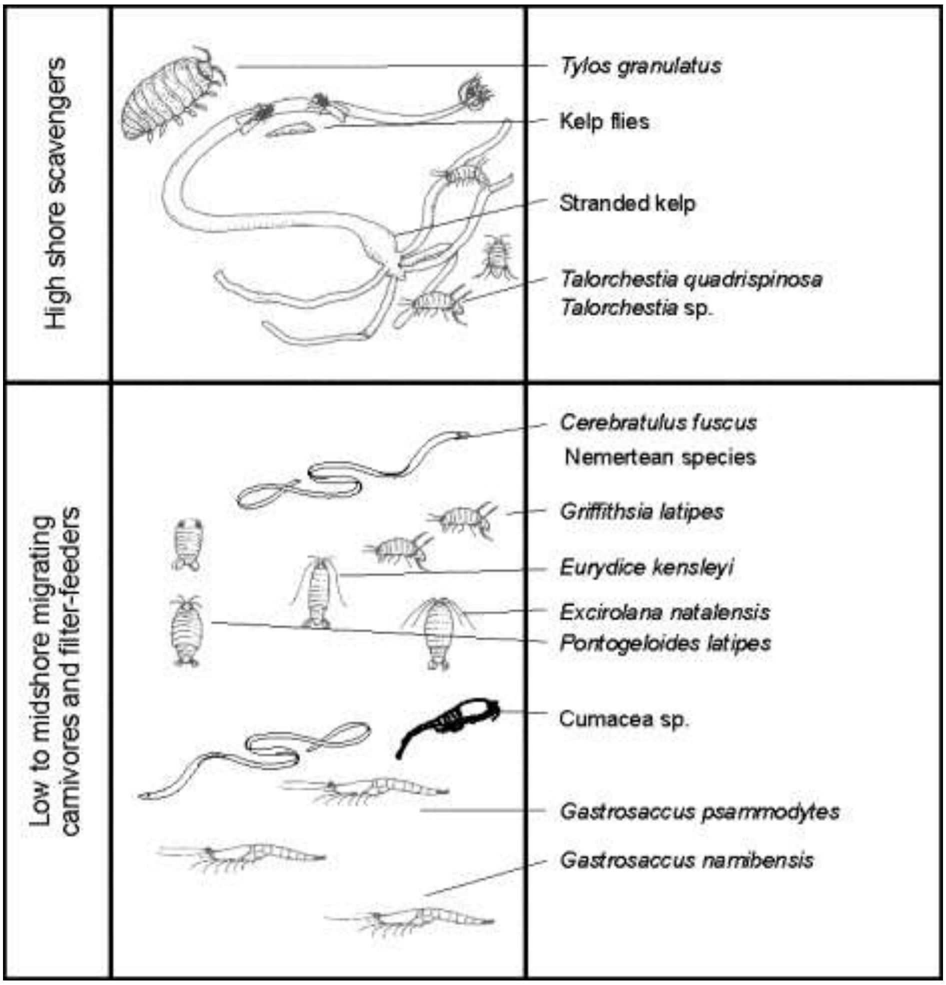


Figure 29. Schematic representation of the West Coast intertidal beach zonation (adapted from Branch and Branch 1981). Species commonly occurring on the southern Namibian beaches are listed.

A depauparate invertebrate macrofauna, both with regard to species diversity and biomass, is considered characteristic of high-energy west coast beaches (Nel et al., 1997; Meyer et al., 1998). The diversity, abundance and biomass of the macrofauna of the beaches in Mining Area 1 is, however, the lowest recorded anywhere in southern Africa (Clark & Nel, 2002). Clark & Nel (2002) reported that the meiofauna of the MA1 beaches similarly lacked diversity, as expected in coarse-grained beaches with a harsh wave environment. Specific information on the microphytobenthos from this area is lacking.

5.1.2 Surfzone fish

Most surf-zone fish on the west coast are common to sandy, rocky and mixed shore habitats but differ in abundance according to proportions of rock and sand and degree of exposure (Clark, 1997). Fish species diversity is greatest at intermediate levels of exposure but also increases with habitat heterogeneity from sandy to mixed shores.

Harders or southern mullet *Liza richardsonii* dominate surf-zone (and estuarine) fish assemblages across all habitat types throughout the west coast from Cape Point to Swakopmund. This species typically contributes 80% to the total fish biomass of sandy surf-zones such as that adjacent to the Orange Estuary mouth. This high biomass supports the gillnet and beach-seine fisheries in the region. Harders spawn close to shore and eggs larvae are entrained in the nearshore by surf-zone gyres. Both the Orange Estuary and adjacent sandy surf-zone serve a nursery function for juveniles of this species. Adults and juveniles optimise conditions by moving between estuary and sea and the health of both habitats is crucial in sustaining harder populations and the fisheries that depend upon them.

In terms of numbers, small-bodied Cape silverside *Atherina breviceps*, sandgoby *Psammogobius knysnaensis* and Cape sole *Heteromycterus capensis* also contribute significantly to the fish assemblage of sandy-beach surfzones adjacent to the Orange Estuary. Exploited fish species are represented mostly by silver kob *Argyrosomus inodorus*, elf *Pomatomus saltatrix* and west coast steenbras *Lithognathus aureti*. West coast dusky kob *Argyrosomus coronus* and white steenbras *Lithognathus lithognathus* are also important.

Species such as *Pomatomus saltatrix, Argyrosomus inodorus, Lithognathus lithognathus* and *Lithognathus aureti* tend to be distributed within the warmer-water areas along the west coast (Lamberth, 2003). These warm areas are limited and tend to be in shallow bays, estuaries or warm-water plumes in the vicinity of estuary mouths. Hypothetically, the southward distribution of Angolan dusky kob *Argyrosomus coronus* and west coast steenbras *L. aureti*, both non-estuarine marine species, to as far as Langebaan Lagoon, may depend on the availability of warm-water refugia offered by estuary mouths and plumes. Southward movement is most likely during anomalous years when the barrier presented by the Lüderitz upwelling cell breaks down or when there is a southwards intrusion of warm water during Benguela Niño years - the nett result being warmer coastal waters (Van der Lingen et al., 2006). Once upwelling resumes, populations of these species that have penetrated south will be confined to the limited warm-water areas provided by estuaries, their plumes in adjacent surf-zones and shallow bays. Consequently, a reduction in river flow may influence the distribution of these species by reducing the extent and availability of these refugia. A similar process could facilitate exchange between South African, Namibian and Angolan stocks of *Argyrosomus inodorus, Pomatomus saltatrix* and *Lichia amia*. All three of these species as well as *Lithognathus lithognathus* and *L. aureti*, are important commercial and/or recreational fish in the region.

5.2 Nearshore marine environment

5.2.1 Nearshore soft sediments invertebrates

Whilst many empirical studies related community structure to sediment composition (e.g. Christie 1974, 1976; Warwick et al., 1991; Yates et al., 1993; Desprez, 2000; van Dalfsen 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 Dalfsen et al., 2000; Zajac et al., 2000; Parry et al., 2003), with evidence of mass mortalities and substantial recruitments (Steffani & Pulfrich, 2004). 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, although Rogers and Rau (2006) considered oxygen levels in the sediments of the Orange River Delta (> 2 mg/ℓ O_2) to be sufficient to support benthic fauna, 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). There is a poor understanding of the responses of local continental shelf macrofauna to low oxygen conditions, but 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.

Numerous studies have been conducted on southern African West Coast continental shelf benthos, mostly focused on mining or pollution impacts (Christie & Moldan, 1977; Moldan, 1978; Jackson & McGibbon, 1991; Environmental Evaluation Unit, 1996; Parkins & Field, 1997; 1998; Pulfrich & Penney, 1999b; Goosen et al., 2000; Savage et al., 2001; Steffani & Pulfrich, 2004a,b). The description below is drawn from the various baseline and monitoring surveys conducted by diamond mining companies (Bickerton & Carter, 1995; Steffani & Pulfrich, 2007; Steffani, 2007a,b,; 2009a,b; 2010a,b; 2010c, 2012), supplemented by the work of Christie (1974), who undertook a systematic study investigating macrobenthic community distributions across the continental shelf off Lamberts Bay (South Africa).

In general, species diversity, abundance and biomass increase from the shore to 80 m depth, with communities being characterised equally by polychaetes, crustaceans and molluscs and biomass ranging from 3,62 g/m^2 dry weight to 16,2 g/m^2. This comparatively low biomass reflects the high depositional environment (estimated at 3,70 mm/year (Meadows et al., 1997)) on the inner continental shelf, with sediments emanating from the Orange River, or re-mobilised and transported in seabed turbulence within depths affected by swells, constantly smothering the area in freshly deposited sediment. Fine sands almost exclusively dominate sediment texture at these depths. Levels of bioturbation in these sediments are very low, with laminations in the sediment providing a relatively undisturbed record of historical flood events (Mabote et al., 1997). High sedimentation rates in the Orange River Delta also contribute to low organic matter content in this area (Rogers, 1977, Mabote et al., 1997). Faecal pellets are rare in sediments on the Orange River Delta, but they increase southwards in the mudbelt to off the Olifants River (200 km to the south) where over 90% of the sediment is composed of faecal pellets (Rogers and Bremner 1991).

In contrast, the mid-shelf mudbelt (70 – 120 m depth) is a particularly rich benthic habitat where biomass can attain 60 g/m² dry weight (Christie, 1974; see also Steffani, 2007b). Clays, silts and very fine sands dominate the sediment texture in the mudbelt, and scavenging and carnivorous polychaete worms, together with cnidarians, dominate the fauna. The infaunal burrowing activity associated with the higher biomass is sufficient to destroy laminations in the sediment (Mabote et al., 1997; Meadows et al., 1997; Meadows et al., 2002; Rogers and Rau, 2006). The comparatively high benthic biomass in this region also represents an important food source to carnivores such as the mantis shrimp, cephalopods and demersal fish species (Lane & Carter, 1999). Below this mid-depth zone, very fine sands dominate the sediment texture, and biomass declines to 4,9 g/m² at 200 m depth and remains consistently low (<3 g/m²) on the outer shelf, from 200 – 500 m depth (Christie, 1974). Crustaceans increase in relative importance in the biota, with amphipods comprising the major component at these deeper depths.

Typical species occurring at depths on the southern African West Coast included the snail *Nassarius* spp., the polychaetes *Orbinia angrapequensis*, *Nepthys sphaerocirrata*, several members of the spionid genera *Prionospio*, and the amphipods *Urothoe grimaldi* and *Ampelisca brevicornis*. The bivalves *Tellina gilchristi* and *Dosinia lupinus orbignyi* are also common in certain areas (Figure 30).

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,b; 2007; Steffani 2007a,b).

Figure 30. Benthic macrofaunal genera commonly found in nearshore sediments include: (top: left to right) Ampelisca, Prionospio, Nassarius; (middle: left to right) Callichirus (Callianassa), Orbinia, Tellina; (bottom: left to right) Nephtys, hermit crab, Bathyporeia

5.2.2 Demersal fish

Available data on offshore soft-sediment fish assemblage and communities off the Orange River is almost exclusively restricted to that collected by the annual demersal trawl surveys of the South African Fisheries Branch since 1985. Changes in the fish assemblage that occurred before then can only be inferred from anecdotal accounts and sparse data of commercial trawl fisheries that used to occur there. Nevertheless, there were a number of coincidental events such as floods, major dam construction and fishery collapse that suggest a number of relationships, short and long-term between catchment flow and fish and fisheries in the marine environment.

Dominant contributors to the biomass of soft sediment fish off the Orange- Senqu, each contributing approximately 20% are Cape gurnard *Chelidonichthys capensis*, sharks and rays e.g. smooth-houndshark *Mustelus mustelus* and horse-mackerel *Trachurus trachurus*. The latter fish is more of a midwater species. Also important are St Joseph shark *Callorhynchus capensis*, rat-tail *Caelorinchus simorhynchus*, spiny dogfish *Squalus* spp., monkfish *Lophius vomerinus* and jacopever *Helicolenus dactylopterus* each contributing 5% to demersal fish biomass. Species that were historically more important such as the commercially exploited west coast sole *Austroglossus microlepis* and kingklip *Genypterus capensis* now contribute less than 1% of the fish biomass in this soft sediment habitat.

5.2.3 *Planktic Communities*

The pelagic invertebrate communities of the Benguela are highly variable both spatially and temporally, their abundances largely being determined by the upwelling regime. The following brief description of the planktic communities is summarised from Andrews and Hutchings (1980); Chapman and Shannon (1985) and Shannon and Pillar (1986).

Phytoplankton, as the principle primary producer is dominated by large-celled organisms, 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. In the surf-zone, diatoms and dinoflagellates are nearly equally important members of the phytoplankton, and some silicoflagellates are also present.

Red tides (dinoflagellate and/or ciliate blooms) are ubiquitous features of the Benguela system. 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.

The mesozooplankton (≥200 μm) is dominated by copepods, which are overall the most dominant and diverse group in southern African zooplankton and typically occur in the phytoplankton-rich upper mixed layer of the water column. The macrozooplankton (≥1,600 μm) are dominated by euphausiids of which 18 species occur in the area.

Zooplankton biomass varies with phytoplankton abundance and, accordingly, seasonal minima will exist during non-upwelling periods when primary production is lower, and during winter when predation by recruiting anchovy is high. More intense variation will occur in relation to the upwelling cycle; newly upwelled water supporting low zooplankton biomass due to paucity of food, whilst high biomasses develop in aged upwelled water subsequent to significant development of phytoplankton. Irregular pulsing of the upwelling system, combined with seasonal recruitment of pelagic fish species into West Coast shelf waters during winter, thus results in a highly variable and dynamic balance between plankton replenishment and food availability for pelagic fish species.

Although ichthyoplankton (fish eggs and larvae) comprise a minor component of the overall plankton, it remains significant due to the commercial importance of the overall fishery in the region. Various pelagic and demersal fish species are known to spawn in the central Benguela, (including pilchard, anchovy, round herring, chub mackerel, lanternfish and hakes), and their eggs and larvae form an important contribution to the ichthyoplankton in the region. High densities of larval and juvenile anchovy *Engraulis encrasicolus* are associated with the turbidity plume off the Orange and other estuaries on the west coast. The use of river plumes as refugia or juvenile nursery areas is characteristic of many small pelagic fish populations globally. The Orange River plume might also represent a productivity front but this is likely to be insignificant to that generated by upwelling in the region. Nevertheless, anchovy are one of the main targets of the small-pelagic industry, the most important commercial fishery in terms of landed biomass and second to hake in catch value for both Namibia and South Africa.

5.3 Rocky intertidal shores

The coastline to the south of the Orange Estuary is dominated by rocky intertidal habitats (Figure 31). Along the beach-dominated coast to the north of the river mouth, rocky intertidal habitats are sparse, however, being represented only by occasional small rocky outcrops and wave-cut platforms north of Mittag (28°21'S; 16°07'E).

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 benthic communities of rocky shores throughout the region show a high degree of similarity, being in essence ubiquitous throughout the biogeographic province, differing primarily in response to substratum (sediment presence and reef structure and profile), and with exposure to wave action. 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 (Bustamante et al., 1997). Sheltered shores support lower biomass, and algae form a large portion of this biomass (McQuaid & Branch, 1984; McQuaid et al., 1985).

Figure 31. Typical rocky intertidal zonation on the southern African west coast

The following descriptions of typical communities found in different rocky shore zones on the South African West Coast are summarised from Field et al., (1980), Branch & Branch, (1981), Branch & Griffiths, (1988) and Field & Griffiths, (1991). In southern Namibia, intertidal fauna and flora have been studied on rocky shores in Mining Area 1 (Pulfrich & Atkinson, 2007), adjacent to some of the southern Namibian pocket-beaches (Meyer et al., 1998; Clark et al., 2004, 2005, 2006; Pulfrich et al., 2007), and in the Lüderitz and Elizabeth Bay region (Penrith & Kensley, 1970;

Lawson et al., 1990; Bustamante et al., 1993; Engeldow & Bolton, 1994; Bustamante & Branch, 1996a; Parkins & Branch 1995, 1996, 1997; Pulfrich 1998a, 2004b, 2005, 2012).

Supra-littoral fringe – Littorina zone: The supra-littoral fringe is the uppermost part of the shore most exposed to air, and so has more in common with the terrestrial environment. The supra-littoral is characterised by low species diversity, with the tiny gastropod *Afrolittorina knysnaensis*, and the red alga *Porphyra capensis* constituting the most common macroscopic life.

Upper Mid-littoral – Upper Balanoid zone: The upper mid-littoral is characterised by the limpets *Scutellastra granularis* and *Siphonaria capensis*, which are present on almost all shores. The gastropods *Oxystele variegata*, *Nucella dubia* and *Helcion pectunculus* are variably present, as are low densities of the barnacles *Chthalamus dentatus*, *Tetraclita serrata* and *Octomeris angulosa*. Flora are mainly represented by the leafy green alga *Ulva* spp., with the encrusting red alga *Hildenbrandia rubra* present in damp depressions.

Lower Mid-littoral – Lower Balanoid zone: Toward the lower shore, biological communities are determined largely by exposure to wave action. Sheltered shores are dominated by grazers, principally the limpets *Scutellastra granularis*, *Cymbula granatina* and a diversity of foliose algae. Algal diversity is typically high, with a variable representation of the green algae *Codium* and *Cladophora* spp., the brown algae *Splachnidium rugosum*, *Chordariopsis capensis*, and the red algae *Nothogenia erinacea*, *Aeodes orbitosa*, *Iridea capensis*, *Gigartina radula*, *G. stiriata*, *Champia lumbricalis* (often epiphytised by *Aristothamnion collabens*), with some *Polysiphonia* and articulated and crustose corallines. The gastropods *Burnupena* spp. and the starfish *Patriella exigua* are also common. However, filter-feeder biomass is low on sheltered shores, whereas densities of *C. granatina* can reach 180 individuals/m², partially feeding on drift kelp that accumulates there, but also directly restricting algal biomass by grazing on algal sporelings that might settle in this zone (Bustamante et al., 1995).

On exposed and semi-exposed shores throughout the region, where there are typically high levels of suspended particulate organic detritus in the water, the limpet *C. granatina* is virtually absent, and almost all of the available substratum space may be occupied by filter-feeders, principally the invasive Mediterranean mussel *Mytilus galloprovincialis*, the black mussel *Choromytilus meridionalis*, and/or the reef building polychaete *Gunnarea capensis* (Bustamante & Branch 1996a). The higher wave action on these exposed shores favours filter-feeders (McQuaid & Branch, 1985) by increasing the concentration and turnover of particulate detritus (Engledow & Bolton, 1994, Bustamante & Branch, 1996b), resulting in higher overall biomass (Bustamante et al., 1995). Several algal species are typically associated with the *Gunnarea* reefs, notably *Ceramium* sp., *Leathesia difformis*, *Caulacanthus ustulatus* and *Cladophora*.

Sub-littoral fringe – Argenvillei zone: Along the visually obvious sub-littoral fringe on semi-exposed and exposed shores, the limpet *Scutellastra argenvillei* dominates, except where it has been displaced by the invasive mussel *M. galloprovincialis*. Densities of *S. argenvillei* can exceed 300/m², often forming a mono-specific belt excluding all other species (Bustamente et al., 1995; Steffani & Branch, 2003a, 2003b). The kelps *Laminaria pallida* and *Ecklonia maxima* dominate the algal diversity in this zone, and where limpet densities are lower, there is variable representation of the flora and fauna described for the lower midlittoral above. This includes the anemone *Aulactinia reynaudi*, other patellid limpets, numerous whelk species and the sea urchin *Parechinus angulosus*. On more exposed shores, the mussels *Aulacomya ater*, *C. meridionalis* and *M. galloprovincialis*, or the tunicate *Pyura stolonifera* (red-bait) may dominate. Most of these species also extend into the subtidal zone below.

Rocky intertidal fish: As alluded to previously, most surf-zone fish on the west coast are common to sandy, rocky and mixed shore habitats but differ in abundance according to proportions of rock and sand and degree of exposure (Clark, 1997). The same applies to the intertidal as many fish of nearshore and surfzone habitats forage in the intertidal at high tide or are resident or find refuge in rock pools at low tide.

Important exploited fish species include the ubiquitous harders, both adult and juveniles that forage in the intertidal as well as galjoen *Dichistuis capensis*, the most sought after angling fish on the Namibian and South African coasts. In comparison, other exploited species are in low numbers. Small- bodied cryptic fish such as the klipvis *Clinus* spp, gobies *Caffrogobius* spp. and pipefish *Syngnathus* spp. are also typical of the rocky intertidal. The links between the rocky intertidal fish assemblage and the Orange Catchment, both positive and negative, are more tenuous than in other nearshore habitats. The 1988 Orange River floods diluted coastal waters causing mass mortalities of shallow-water invertebrates and kelps (Branch et al., 1990). No mortalities of intertidal fish were recorded, probably due to their escape to deeper more saline waters, whereas freshwater fish washed out of the river eventually succumbed to osmotic shock (Morant and O'Callaghan, 1990). In contrast, a low-oxygen, hydrogen sulphide event in St Helena Bay to the south caused mass mortality of most intertitidal and nearshore fish (Lamberth et al., 2010).

5.4 Mixed shores

The semi-exposed to exposed rocky shores occurring along the beach-dominated coastline north of the river mouth are strongly influenced by sediments, and may include considerable amounts of sand intermixed with the benthic biota. This intertidal mixture of rock and sand is referred to as a mixed shore. Substantial fluctuations in the degree of sand coverage are common (often in response to seasonal cycles in wave energy), and the fauna and flora of mixed shores can be impoverished compared to more homogenous shores. However, mixed shores can provide important habitat for opportunistic species capable of sequestering within sand, but susceptible to elimination by competition in more uniform rocky intertidal environments. The rocky/mixed shores in MA1 have recently received attention (Pulfrich & Atkinson, 2007; Pulfrich 2008, 2009, 2010, 2011, 2012), and the description below is based on these reports (Figure 32).

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 characterised 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 composition of the intertidal and subtidal macrophytes on mixed shores is dominated by sand-tolerant and opportunistic foliose genera, such as *Cladophora*, *Enteromorpha*, *Chaetomorpha*, and *Chondria* spp. Lower on the shore, the communities are typically dominated by foliose algae, particularly by the red foliose algae *Gigartina scutellata*, *G. stiriata* and *Gymnogongrus glomeratus*, and red filamentous ephemeral species such as *Ceramium capense* and *Polysiphonia virgata*, which often occurred in mixed stands with other unidentified turf-algae. Many of the psammophytic (sand-tolerant) algal species have mechanisms of growth, reproduction and perennation that contribute to their persistence on sand-influenced shores. Specific adaptations include peak growth and reproduction just prior to seasonal burial, abbreviated life cycles, regeneration of fronds from basal parts, or rhizomatous growth (Daly & Matheison, 1977; Airoldi et al., 1995). The mixed-shore habitat also provides important refuges for opportunistic species capable of sequestering vacant space, but susceptible to elimination by competition in less disturbed intertidal environments.

Figure 32. Typical sand-influenced rocky shores in Mining Area 1, southern Namibia (Photo: Andrea Pulfrich).

Invertebrate species common on the low-shore include the alien invasive mussel *Mytilus galloprovincialis* and the sand-tolerant anemone *Aulactinia reynaudi*. Interspersed in this low shore band were numerous limpets (*Scutellastra argenvillei*, *S. granularis* and *Cymbula granatina*) and whelks (*Burnupena* spp., *Nucella cingulata* and *N. dubia*). The predatory gastropod *Burnupena* spp., although common on rocky shores, also occur on mixed shores due to its adaptive ability of both moving over sand, as well as burrowing into it. Likewise various species of sea cucumbers (*Roweia frauenfeldii* and *Thyone aurea*) common in rock crevices and between mussels can tolerate sand burial (Branch et al., 1994; Brown, 1996). Further up in the mid-shore zone, *M. galloprovincialis*, the reef-building polychaete *Gunnarea capensis* and *A. reynaudi* were the dominant space occupiers, together with *Scutellastra granularis* and the Cape false limpet *Siphonaria capensis*, the latter in particular extending its distribution into regions where sand deposition is a regular occurrence (Marshall & McQuaid, 1989). Foliose algae characterising the mid-shore were dominated by *Iridea capensis* and *Ulva* sp. The mostly barren high-shore was characterized by the tiny snail *Afrolittorina* (=*Littorina*) *africana*, the limpets *S. granularis* and *S. capensis* and the opportunistic algae *Porphyra capensis*, *Ulva* sp. and *Phyllymenia belangeri*. Of interest is the dominance by the indigenous mussel *Choromytilus meriodionalis* on mixed shores in the south of the study area. Although more tolerant to sand-cover (Brown et al., 1991; Marshall & McQuaid, 1993), this species has largely been replaced on many rocky shores on the southern African West Coast by the thriving invader *M. galloprovincialis* (Steffani & Branch 2003a).

To reiterate, most surf-zone fish on the west coast are common to sandy, rocky and mixed shore habitats but differ in abundance according to proportions of rock and sand and degree of exposure (Clark, 1997). In contrast to the low diversity of invertebrate communities on mixed shores in the region, fish species diversity and abundance is greatest at intermediate levels of exposure but also increases with habitat heterogeneity from sandy to mixed shores. These shores are characterised by extensive sand movement and the repeated scouring or burial of algal and invertebrate communities. Much of the fish diversity can be attributed to grazers such as galjoen *Dichistius capensis* and harders *Liza richardsonii* feeding on new algal and invertebrate growth on recently scoured rock on the high tide. Piscivorous predators such as silver kob *Argyrosomus inodorus* and cowsharks *Notorynchus cepedianus* are also abundant in these mixed shore habitats.

5.5 Subtidal reefs and kelp beds

The biological communities of the sublittoral reef habitat can be broadly grouped into an inshore zone (from the supralittoral fringe to a depth of ~10 m), and an offshore zone (below 10 m depth). The shift in communities from the flora-dominated inshore zone to the fauna-dominated offshore zone is diffuse, however, with a gradation in abundance across species distributions by depth. As wave exposure is moderated with depth, wave action is less significant in structuring the communities than in the intertidal, with prevailing currents, and the vertical distribution of oxygen, temperature, turbidity, suspended sediments, particulate organic matter and nutrients playing more important roles.

Research on subtidal organisms along the southern Namibian and northern Namaqualand coastlines has been limited. Current knowledge is primarily restricted to macrobenthic reef communities in depths of less than 30 m in the area around Lüderitz (Tomalin, 1993; Parkins & Branch, 1996, 1997; Pulfrich, 1998a,b; 2007a,b; Pulfrich & Penney, 1998, 1999a, 2001) and Port Nolloth (Pollock, 1982; Hutchings & Clark, 2008). The following descriptions are summarised from these studies. To the north of the river mouth, rocky subtidal habitats are limited to a series of coast-parallel reefs running northwards from approximately latitude 28°25' S and extending ~4 km offshore. In contrast, to the south of the river mouth, reefs are primarily located in shallow subtidal regions adjacent to the rocky coastline, with offshore reefs being limited to a few isolated outcrops within 1 km from the shore that rise to levels slightly shallower than 15 m depth (Pollock, 1982).

In the nearshore areas these reefs are dominated by kelp beds (*Laminaria pallida* and *Ecklonia maxima*) (Figure 33). As wave exposure in the region is very high, kelp beds play a major role in absorbing and dissipating much of the wave energy reaching the shore, thereby providing important semi-exposed habitats for a wide diversity of both marine flora and fauna. In the inshore zone, the benthos is largely dominated by algae, in particular two species of kelp. The canopy-forming kelp *Ecklonia maxima* extends seawards from the shallow sub-tidal to a depth of about 10 m. The smaller

Laminaria pallida forms a sub-canopy to a height of about 2 m underneath *Ecklonia*, and in clearer water continues seaward to about 30 m depth. Increasing turbidity around the Orange River mouth and further north limits growth to shallower waters (15 m) (Velimirov et al., 1977; Jarman & Carter, 1981; Pollock, 1982). *Ecklonia maxima* is the dominant species along the South African coastline, but is poorly represented in southern Namibia, extending to just north of Lüderitz. *Laminaria* becomes the dominant kelp northwards, extending as far north as Rocky Point in northern Namibia (Stegenga et al., 1997). The sand-dominated, submarine Orange delta that extends seawards of the mouth plays and important role in determining the distribution of nearshore kelp in the region (Figures 33 and 34).

Through a combination of shelter and provision of food, kelp beds support recruitment and complex trophic food webs of numerous species, including the commercially important rock lobster. Growing beneath the kelp canopy and epiphytically on the kelps themselves are a diversity of under-storey algae which provide both food and shelter for predators, grazers and filter-feeders associated with the kelp bed ecosystem (Figure 35). These plants and animals all have specialised habitat and niche requirements, and together form complex communities with highly inter-related food webs. Representative under-storey algae include *Botryocarpa prolifera*, *Neuroglossum binderianum*, *Botryoglossum platycarpum*, *Hymenena venosa* and *Epymenia obtusa*, various coralline algae, as well as subtidal extensions of some algae occurring primarily in the intertidal zones (Bolton, 1986). Epiphytic species include *Carradoria virgata*, *Suhria vittata* and *Carpoblepharis flaccida*.

The sublittoral invertebrate fauna is dominated by suspension and filter feeders, such as the ribbed mussel *Aulacomya ater* and Cape Reef worm *Gunnarea capensis*, a variety of sponges, and the sea cucumbers *Pentacta doliolum* and *Thyone aurea* (Holothuroidea, Echinodermata) (Figure 36). Grazers are less common with most herbivory being restricted to grazing of juvenile algae or debris feeding of detached macrophytes. The dominant grazer is the sea urchin *Parechinus angulosus*, with lesser pressure from limpets, the isopod *Paridotea reticulata* and the amphipod *Ampithoe humeralis*. Key predators in the sublittoral include the commercially important rock lobster *Jasus lalandii* (Macrura, Crustacea) and the isopod *Cirolana imposita*. Of lesser importance although numerically significant is the starfish *Henricia ornata*, various feather and brittle stars (Crinoidea and Ophiuroidea, Echinodermata), and gastropods (*Nucella* spp. and *Burnupena* spp.).

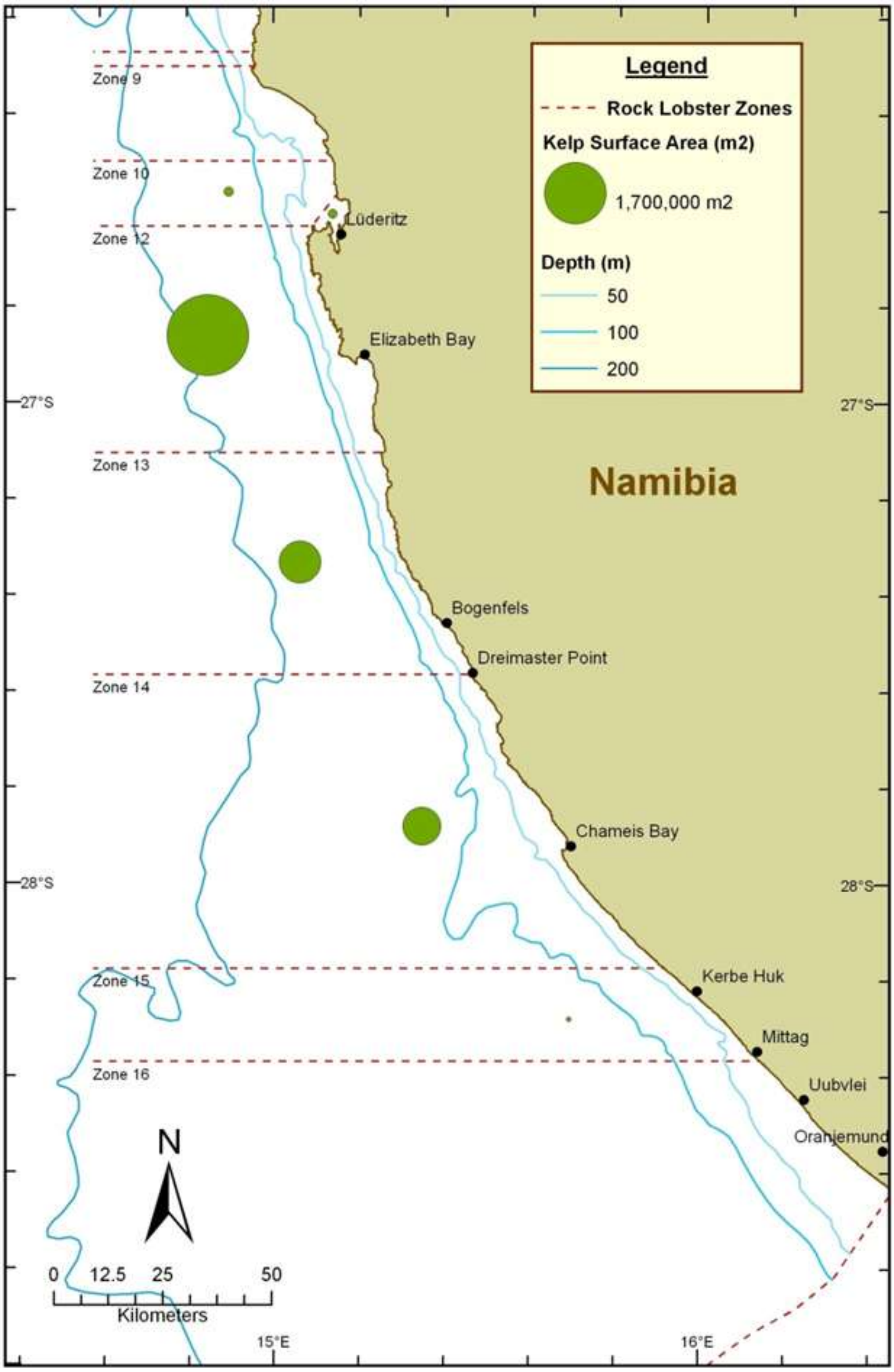


Figure 33. Estimated kelp bed area in the rock lobster fishing zones south of Lüderitz (adapted from Pulfrich and Penney, 2006)

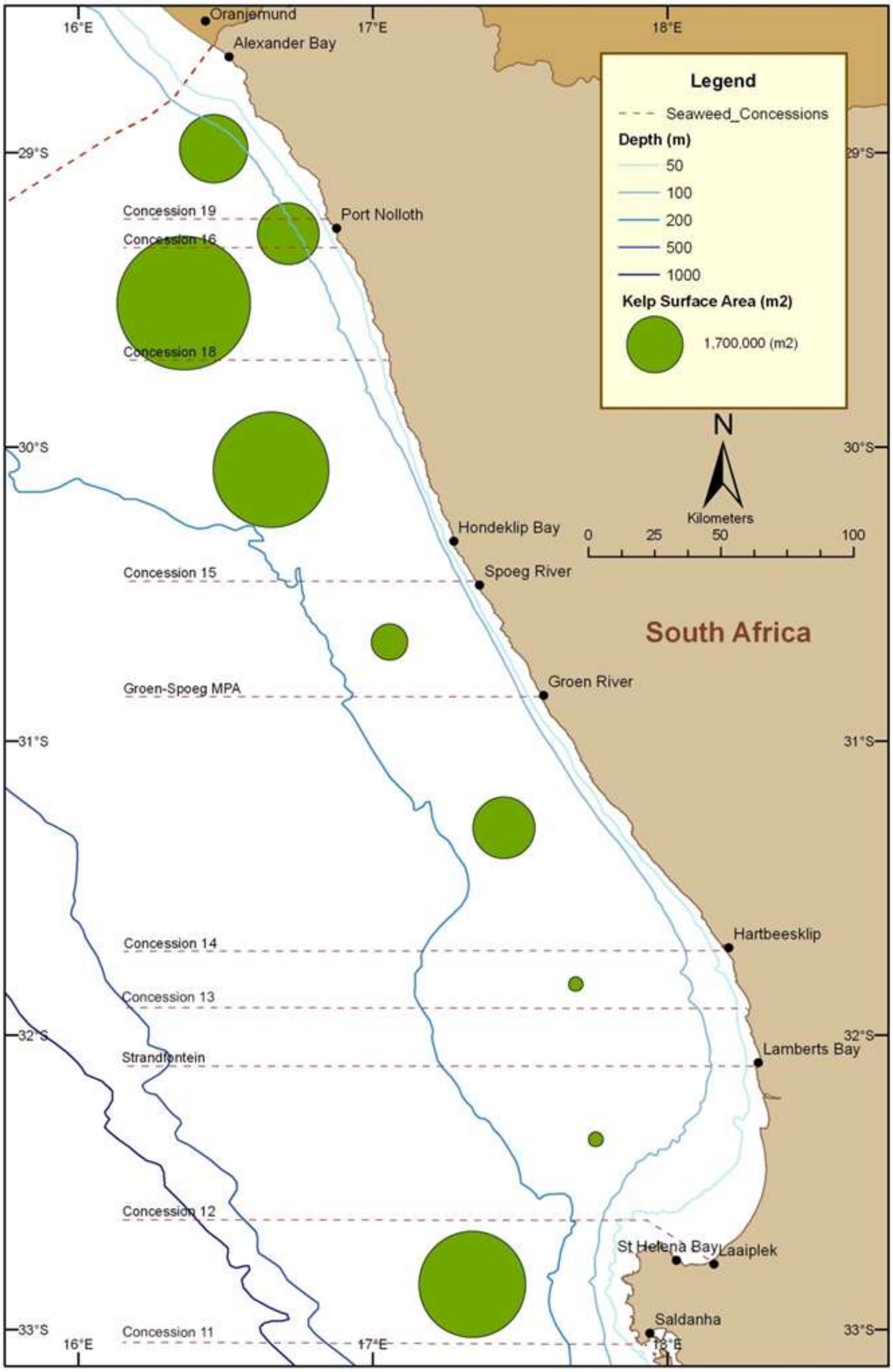


Figure 34. Estimated kelp bed area in the South African kelp concessions between the Orange River mouth and Cape Columbine (from Pisces, 2007)

Figure 35. Typical kelp bed dominated by Laminaria pallida occurring on shallow platform reefs off Mining Area 1 (left) (Photo: Pisces), and its diverse understorey community (right) (Photo: Kolette Grobler, MFMR).

The fish assemblages of kelp forests and subtidal reef in the region are represented by important commercial linefish species such as hottentot *Pachymetopon blochii* and jacopever *Helicolenus dactylopterus* as well as by cryptic fish and elasmobranchs such as twotone fingerfin *Chirodactylus brachydactylus* and dark shyshark *Haploblepharus pictus* respectively. Large predators such as cowsharks *Notorynchus cepedianus* are common as are aggregations of silver kob *Argyrosomus inodorus* and *Lithognathus* spp. These aggregations tend to be seasonal and predictable and succeptible to targeting by commercial and recreational fishers. The relationship between these fish assemblages and flow from the Orange is likely to be indirect and according to the influence of catchment flow and sediment dynamics on the distribution of kelp and the burying or exposure of subtidal reefs.

Figure 36. Nearshore reef communities off Lüderitz dominated by a diversity of encrusting sponges, encrusting coralline algae, soft corals, echinoderms and ribbed mussels (left), and providing optimal habitat for rock lobsters (right) (Photos: Pisces).

5.6 Birds

Colonies of breeding seabirds in the vicinity of the Orange River mouth, located mostly in Namibia, are currently in decline. African Penguins, for example, have reportedly dropped from over 40,000 breeding pairs in 1956 to fewer than 1,000 pairs in 2003, while Cape Gannets have dropped from 0,47 birds/ha in 1956 to 0,02 birds/ha in 1996 (BCLME, 2004). This was thought to be due to food scarcity, which resulted in poor recruitment to colonies, although some young birds have emigrated to other colonies to the north and south. The food shortage was reportedly caused by the collapse of Namibian sardine and anchovy stocks, and a decreased abundance of gobies in Namibia, caused by the Benguela Nino of 1994/95 (BCLME, 2004). These changes most likely also contributed significantly to reduction in abundance of piscivorous bird populations on the Orange Estuary (cormorants and terns), and are not related to the condition of the river or estuary itself at all.

6 Resource utilisation

6.1 Exploited fish and fisheries

6.1.1 Demersal fisheries

The Benguela Current supports a number of major offshore commercial fisheries including demersal (bottom) trawl and longline fisheries that focus primarily on hake, midwater trawl fisheries that focus on horse mackerel, purse seine fisheries focussing on sardine and anchovy, pelagic longline fisheries focussing on tuna, swordfish and sharks (Crawford et al., 1987, Griffiths et al., 2004). Closer inshore there is also an important commercial fishery for rock lobster, and smaller operations targeting sole and linefish (snoek).

The area off the Orange River mouth is not a particularly important fishing ground for any of these fisheries, except the west coast sole (*Austroglossis microlepis*), presumably owing to low biomass of the other target species in this area. The area off the Orange Mouth may, however, be important as a nursery and/or spawning ground for some species. Two stocks of *A. microlepis* exist in South African and Namibia waters, a southern population centred on the Orange mouth and a northern population opposite the Skeleton Coast (Crawford et al., 1987). The southern population collapsed in the mid 1970s and there has been no directed fishing for this species in South African waters since then. The fishery in Namibian waters to the north of the Orange has shown evidence of recovery but it's not clear whether this is a reflection of the improved status of the southern population or an expansion of the northern population southward. As is the case with many other sole species that are distributed according to sediment type, this one seems to favour areas of fine muddy sediment such as is found of the mouth of the Orange River. Monkfish (*Lophius* spp.), one of the most valuable bycatch species in the bottom trawl fisheries in the Benguela Current, also reportedly spawns off the Orange River mouth (Hampton et al., 1999, Hampton, 2003). These and other demersal species usually have sediment-specific habitat requirements. Presumably, changes in sediment discharge from the river must have had some impact on these two species, both positive and negative and which will be difficult to isolate from the effects of many decades of intense exploitation. This situation is exacerbated by the added difficulty of having to distinguish between environmental versus economic and political drivers of fleet behaviour.

The distribution of deep water hake (*Merluccius paradoxus*) the dominant species in the demersal trawl fishery in both South Africa and Namibia, spans the whole of Namibia and the South African west coast. The bulk (65 – 75%) of the stock is located in South African waters. Spawning seems to be confined to the area south of Cape Town. Eggs, larvae and juveniles are carried northwards up the west coast but remain south of the Orange River until they reach at least 10 cm in length. Thereafter, the small fish begin to move off the shelf into deeper water, spreading north and south, with a considerable portion moving northwards across or along the edge of the Orange shelf up into Namibian waters. Sediment originating from the Orange River that is distributed on the shelf presumably influences the distribution or movement of fish in the nursery area south of the river to some extent. It is not clear how important this is though. Hydrographic features on the Orange shelf areas are reportedly highly dynamic, with varying origin of the water masses, and may temporarily form a barrier to the movement of fish on the shelf (Strømme et al., 2004). The role of the Orange River in this is also not clear, but probably minimal.

One of the major nursery grounds for pelagic fish in the Benguela (sardine *Sardinops sagax*, anchovy *Engraulis encrasicolus* and redeye *Etrumeus whiteheadi*) is located on the continental shelf south of the Orange Mouth (Hutchings et al., 2009). These species spawn on the southern part of the west coast and on the Agulhas Bank (south of the subcontinent). Eggs and larvae are transported in a strong shelf-edge jet up the west coast at which point the pre-recruits move inshore towards the nursery grounds. This said, high densities of larval and juvenile anchovy are associated with the turbidity plume off the Orange and other estuaries on the west coast with that of the former extending more than 50 km seawards. The use of river plumes as refugia or juvenile nursery areas is characteristic of many small pelagic fish populations globally. The Orange River plume might also represent a productivity front but this is likely to be insignificant to that generated by upwelling in the region. Nevertheless, anchovy are one of the main targets of the small-pelagic industry, the most important commercial fishery in terms of landed biomass and second to hake in catch value for both Namibia and South Africa. Significant stocks of adult sardine and anchovy are reported to occur within 30 nautical miles off the Orange Mouth, but these are not exploited for logistic reasons (CSIR 1994, BCLME, 2004).

Nearshore coastal species important to the linefishery such as *Pomatomus saltatrix, Argyrosomus inodorus, Lithognathus lithognathus* and *Lithognathus aureti* tend to be distributed within the warmer-water areas along the west coast (Lamberth, 2003). These warm areas are limited and tend to be in shallow bays, estuaries or warm-water plumes in the vicinity of estuary mouths. Hypothetically, the southward distribution of Angolan dusky kob *Argyrosomus coronus* and west coast steenbras *L. aureti*, both non-estuarine marine species, to as far as Langebaan Lagoon, may depend on the availability of warm-water refugia offered by estuary mouths and plumes. Southward movement is most likely during anomalous years when the barrier presented by the Lüderitz upwelling cell breaks down or when there is a southwards intrusion of warm water during Benguela Niño years - the nett result being warmer coastal waters (Van der Lingen et al., 2006). Once upwelling resumes, populations of these species that have penetrated south will be confined to the limited warm-water areas provided by estuaries, their plumes in adjacent surf-zones and shallow bays. Consequently, a reduction in river flow may influence the distribution of these species by reducing the extent and availability of these refugia. A similar process could facilitate exchange between South African, Namibian and Angolan stocks of *Argyrosomus inodorus, Pomatomus saltatrix* and *Lichia amia*. All three of these species as well as *Lithognathus lithognathus* and *L. aureti*, are important commercial and/or recreational fish in the region

6.1.2 Rock lobster

Rock lobster occurs in commercially exploitable densities along a 900 km length of coastline either side of the Orange River, from about 25°S in Namibia to Cape Town in the south (Crawford et al., 1987). The area immediately south of the river is not considered a good fishing ground for this species, and supports only a small portion of the South African stock. Less than 1.1% of the Total Allowable Catch (TAC) for South Africa has been allocated in this area in recent years (BCLME, 2004). By contrast, the main commercial fishing area for rock lobster in Namibia is located just north of the Orange River (BCLME, 2004). Rock lobster stocks in both countries are severely depressed at the moment, reportedly due to overfishing (Griffiths et al., 2004). This most likely has little nothing to do with the Orange River.

7 Environmental Objectives

There are no explicit rules guiding the setting of the environmental objectives for the nearshore marine environment, but based on the international conventions, Namibian and South African national legislation and current marine resource utilisation, the following objectives were derived (Table 6).

Table 5. Objectives for the nearshore marine environment

Aspect	***Environmental objectives***
Biodiversity	The nearshore environment in the vicinity of the Orange Estuary has been declared a " Biodiversity area of interest" as it shows significant differences in abiotic and biotic characteristics from the adjacent coast. This diversity should be maintained through the regular supply of fluvial sediments to the coast. The strong link between benthic biodiversity and sediment type suggests that recovery of some biota, especially exploited populations, could be facilitated by environmental flow releases re-establishing the reference fluvial sediment budget to the sea. The Orange Estuary is a Ramsar site. The biota in the nearshore marine environment should be maintained at its present health and productivity levels to maintain links between populations as well as connectivity between estuaries and sea and biogeographical regions.
Fisheries	Existing fisheries may not decline in health and productivity. As the region is subjected to significant natural variability this was defined as within 25% of current annual catches.
Beaches	Beaches should be maintained in the present configuration. There is already evidence of coastal erosion along the beaches in the vicinity of the Orange Estuary. Fluvial sediment supply should be maintained at present levels to ensure coastal stability and prevent further erosion of coastline.
Mining	Mining is set to intensify along the Namibian coast. While this activity is off-setting the loss of sediment to the beaches in the vicinity of the Orange Estuary, fluvial sediment is required to allow for a range of sediment grains sizes and the ultimate formation of a more diverse, natural coastal habitat. This function should be maintained.

8 Hydrological scenario assessment

8.1 Hydrological scenarios

All EFR methods followed are in essence scenario based i.e. different flow regimes are evaluated and the ecological states predicted. A realistic set of scenarios (Sc) were therefore developed to allow for the testing of different flow regimes and for recommendations to be made. Recommendations made during this study can be used to optimise scenarios during further studies and attempt to minimise negative impacts on all users.

Scenarios consist of combinations of different drivers. The drivers were combined within the likely time-frame that these developments could take place so as to derive plausible scenarios. The combination of drivers that result in scenarios are illustrated in Table 6. A flow regime for each scenario is produced at the EFR site and then evaluated to predict the consequences on the ecological state.

Table 6. Time lines, scenario and driver combinations

Time frame	***Scenario***	***Orange River drivers***	***Fish River drivers***
Present day	Sc OF 1	Modelled present state current releases and use included.	
2013 – 2020	Sc OF 2	Metolong Dam, Tandjieskoppe, acid mine drainage (AMD) treated.	Neckartal Dam. Increase in Naute Dam irrigation.
	Sc OF 3	Metolong Dam, Tandjieskoppe, AMD treated.	Neckartal Dam with EFR release. Increase in Naute Dam irrigation.
	Sc OF 4	Metolong Dam, Tandjieskoppe, AMD treated, 2010 EFR flows released. Optimised releases from dams.	Neckartal Dam with EFR release. Increase in Naute Dam irrigation.
2020 – 2040	Sc OF 5	Metolong Dam, Tandjieskoppe, AMD treated, 2010 EFR flows released, Polihali Dam, Vioolsdrift Balancing Dam (small). Optimised releases from dams.	Neckartal Dam with EFR release. Increase in Naute Dam irrigation.
Post 2040 – maximum foreseeable development	Sc OF 6	Metolong Dam, Tandjieskoppe, AMD treated, Polihali Dam, Large Vioolsdrift Dam (no EFR), Boskraai Dam. Optimised releases from dams.	Neckartal Dam. Increase in Naute Dam irrigation.
	Sc OF 7	Metolong Dam, Tandjieskoppe, AMD treated, Polihali Dam, Large Vioolsdrift Dam (no EFR), Boskraai Dam. Optimised releases from dams.	Neckartal Dam with EFR release. Increase in Naute Dam irrigation.
	Sc OF 8	Metolong Dam, Tandjieskoppe, AMD treated, Polihali Dam, Large Vioolsdrift Dam (EFR O4 released), Boskraai Dam. Optimised releases from dams.	Neckartal Dam with EFR release. Increase in Naute Dam irrigation.

8.2 Floods

No detailed flood analysis were conducted for this study, but monthly flow volumes higher than 5,000 x 10^6 m^3 were seen as indicate of resetting events that influences the physical processes in the estuary. Under reference conditions this type of event occurred 25 times in the simulated flow period of 66-years (Table 7). Under the present state and Sc 2 their occurrence have been reduced to 8 times, under Sc 3 and 4 they occurs 9 times, while under the worst case Sc 6 and 7 they only occur twice.

Table 7. Number of occurrences of flood and freshwater pulses in a 66-year period in the Orange River as simulated in the offshore modelling study

Flood type	***Flood volume (m^3/s)***	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Inside Estuary	< 150	238	571	585	583	471	545	668	668
Pulse	150 – 1000	350	139	122	126	251	183	72	72
Small floods	1000 – 5000	179	74	77	74	62	58	50	50
Large floods	5000 – 8000	24	7	7	8	7	5	2	2
Very large floods	>8000	1	1	1	1	1	1	0	0

It is important to take cognisance of the changes in frequency of the defined flood events under the various scenarios being proposed. In the change from reference conditions to the present state there has been a dramatic decrease in occurrence of all but the very largest flood events (Table 7). This is particularly true for the smaller flood and high flow pulse events.

9 Implications of flow alteration on abiotic drivers and primary production

The hydrological simulation and numerical modelling results need to be presented in a manner that facilitates the assessment of the likely ecological changes associated with the various proposed scenarios. The expected changes in the offshore environment can be characterised largely in terms of changes in:

- shoreline, intertidal benthic and/or habitats;
- potential changes in pelagic habitats.

9.1 Shoreline, intertidal and benthic habitats

Changes in **shoreline and intertidal habitats** have not been explicitly modelled, but were assessed based on the input of sediments (mainly sand) close inshore and the subsequent alongshore transport of these sediments. These need to be assessed using expert opinion based on a characterisation of the expected change in the quantity of coarser sediments (fine sand, sand, etc.) being discharged into the marine environment. A change in the quantities of these sediments entering the near-shore environment will result in changes in nearshore, intertidal and coastal habitats, including potential changes in shoreline.

The changes in **benthic habitats** are best characterised through the use of a metric that quantifies changes in the extent of deposition of sediments in the nearshore region and further offshore. The metric used is a simple measure of the extent of sediments deposited on the seabed during each event with a thickness that exceeds 0,01 m (the sediment thickness at the seabed is calculated assuming a porosity of approximately 40% for the fine sands and 90% for the silt and two clay fractions). The quantities reported are the median, 80%tile and maximum extent of sediments deposited on the seabed with a thickness exceeding 0.01 m. The extent of sediments deposited on the seabed with a thickness exceeding 0,1 m was also determined however these results are not reported here.

The median extent of sediment deposition (exceeding 0,01 m) provides a measure of where there is a moderate to high degree of persistence of deposited sediments, while the 80%tile distributions represent a more ephemeral sediment distribution (i.e. the sediments persist at the reported thickness for at least 20% of the model simulation period, i.e. exceed the reported sediment thicknesses for a cumulative duration of approximately two weeks during the just over two month duration model simulations.

Concern was expressed on the quality of these benthic habitats. These concerns related to the fact that if there were excessive fines in the deposited sediments, for various reasons (burial effort, burrowing effort, etc.) this could represent sub-optimal benthic habitats. Based on inputs from the ecological experts on this study, two further measures where considered. The first was the extent of benthic habitat exceeding a deposition depth of 0,01 m and where the clay content remained below 40%. The second was the extent of benthic habitat exceeding a deposition depth of 0,01 m, where the clay content remained below 40% and where the sand content exceeded 20%. The motivation behind these metrics is that evidence exists of benthic habitats becoming sub-optimal should the clay content of the sediments be too high and/or the sand content in the sediments too low.

9.2 Pelagic habitats

The changes in pelagic habitats have been characterised using the distribution of water quality parameters such as salinity, total suspended solids (TSS) concentrations/turbidity and dissolved nutrients, particularly dissolved reactive silicate (DRS).

9.2.1 Salinity

Two measures of low salinity habitats were used. The first was the **area** of surface waters where the salinity remained below specified threshold(s), the second being the **volume** of marine waters where the salinity remained below the specified threshold(s). Given the nature of the plume dynamics, it is felt that the measure based on the volume of water not exceeding the specified salinity threshold(s) would be a better measure of change in the extent of pelagic habitats than would be a measure based solely on the area of surface waters not exceeding the specified salinity threshold(s).

Two salinity thresholds were considered, namely a large change (acute) in salinity represented by waters with a salinity <28 psu and a more subtle change (chronic) in salinity represented by waters with a salinity <33 psu. Similar to the measures of changes in the extent of benthic habitats, these areas and volumes of low salinity waters are reported in terms of the median, 80%tile and minimums observed. For salinity, the maximum effect (i.e. extent of changes in pelagic habitats) is represented by the observed **minimum salinities**. Conversely, for the other water quality variables under consideration (e.g. TSS and DRS), the maximum effects (i.e. extent of changes in pelagic habitats) are represented by the **maxima of the particular water quality variable** under consideration.

9.2.2 Nutrients

Compared with concentrations typically measured in seawater in the area (<1,000 μg/ℓ), river inflow can be a significant source of DRS (average 4,800 μg/ℓ) to the adjacent marine environment. DRS is an important nutrient in upwelling systems such as those occurring offshore of the Orange Estuary where diatoms blooms occur. Silicon, which is used in the cell structure of diatoms and silico-flagellates is typically not limiting however during diatom blooms the bioavailability of silicon may be severely lowered (Dawes, 1998). In extreme cases where other nutrients are not limiting, there may be significant shifts from diatom-dominated ecosystems to flagellate-dominated ecosystems that may affect significantly food web dynamics (Doering et al., 1989) or increase the risk of unwanted phytoplankton bloom conditions (e.g. Smayda, 1989).

The spatial extent of surface waters as well as the volume of marine waters exceeding the typical ambient concentrations of DRS in the adjacent marine environment by 20% and 75% (i.e. where the resulting DRS concentrations are 1,2 and 1,75[1] times greater than the ambient DRS concentrations in the ocean) have been calculated using the outputs from the model simulations. These areas and volumes are considered to represent the extent to which river inflows create and enhanced nutrient environment with respect to DRS in the adjacent marine environment. Similar to the measures of changes in the extent of benthic habitats, these spatial extents and volumes are reported in terms of the median, 80%tile and maximums observed.

9.2.3 Total suspended solids

The extent of turbid waters also is considered to be a measure of the quality of pelagic habitats. The effects of elevated turbidity may be both positive and negative, however excessively high water column turbidity (e.g. TSS >100 mg/ℓ) is considered to constitute a degraded environment for most species in the region (Probyn, 2000; EMBECON, 2004). More subtle changes are expected at lower TSS thresholds (e.g. 20 mg/ℓ). Consequently the area of surface waters as well as the volume of marine waters exceeding the TSS concentrations of 100 mg/ℓ and 20 mg/ℓ have been calculated. The total suspended sediment concentrations reported comprise all four sediment fractions modelled, namely fine sand, silt, and two (coarse and fine) clay fractions. These areas and volumes are considered to represent the extent to which river inflows have discernible turbidity effects in the marine environment. Similar to the measures of change in the extent of benthic habitats, these spatial extents and volumes are reported in terms of the median, 80%tile and maximums observed.

It should be noted that variability in TSS concentrations can be complex and long lasting due to the continual re-suspension and redistribution of sediments that occurs during and after storm events.

9.3 Indices of change for the proposed scenarios

The changes in the abiotic environment under the various proposed scenarios have been assessed relative to both reference conditions (that prevailed before any modifications in the catchment) and relative to present state (presently existing conditions). For all of the proposed scenarios each of the metrics is normalised relative to the magnitude of the metric under either reference conditions (i.e. is presented as a percentage change relative to reference conditions).

Significance ratings of the assessed changes

The final step is to translate these percentage changes in offshore habitats (under the various developments scenarios) into significance ratings. This has been done both for predicted changes

1 Based on dilution calculations, the DRS concentration contours of 1,200 µg/ℓ and 1,750 µg/ℓ are roughly represented in the model results by the 33 psu and 28 psu salinity contours, respectively.

relative to reference conditions. It is acknowledged that a degree of subjectivity exists in the selection of the mapping values used and that they are unlikely to fully represent all ecological sensitivities and changes in the marine environment, however the resolution provided by these mapping values are considered to be appropriate given other uncertainties in the overall modelling study (e.g. flood event categorisations, modelling durations, etc.). The mapping between percentage changes in the relevant metric and the significance of these changes under the various scenarios is summarised in Table 8 below.

While the mapping used is the same for estimated percentage changes with respect to the reference and, the interpretation of the significance and nature of the changes differ. For example, the significance of the predicted changes relative to reference conditions typically range between 0 for reference conditions (i.e. minimal change) to -2 to -3 for present state conditions as well as expected conditions under proposed scenarios (i.e. moderately significant to highly significant decreases).

Table 8. Mapping used to translate changes (%) into significance ratings/indices of predicted change

Significance rating	***Metric reported as a % of the magnitude of the metric under reference conditions***	***Comment***
3	175% to 200%	Highly significant increase.
2	150% to 175%	Moderately significant increase.
1	125% to 150%	Discernible increase.
0	75% to 125%	Minimal change.
-1	50% to 75%	Discernible decrease.
-2	25% to 50%	Moderately significant decrease.
-3	0% to 25%	Highly significant decrease.

9.4 Changes in offshore benthic habitats

Changes in the offshore benthic habitats have been characterised in terms of the thickness of sediments deposited in the marine environment as well as the sediment grain size distributions considered to constitute appropriate benthic habitats for the species utilising these habitats.

Contours of the maximum thickness of deposited sediments (> 0,01 m) for the characteristic flood sizes (very large flood, large flood, small flood and high inflow pulse of freshwater) simulated clearly indicate that there is a significant difference in the extent of these deposited sediments on the seabed for the various flood sizes (Figure 37 to 39).

Particularly notable is that the sands remain close to the coast as they are less easily re-suspended than the silts and clays. However the silts and clays do not persist in great quantities in these shallower waters as these fines (silts and clays) are relatively easily re-suspended by the wind and wave turbulence that prevails in these shallow waters and consequently are transported into deeper, more quiescent waters further offshore where they are deposited.

In an attempt to map quality of benthic habitats the extent of benthic habitats meeting two further criteria have been mapped. These two criteria are:

- the extent of benthic habitats where the sediment thickness exceeds 0,01 m and the percentage of clays <40% (Figure 40);
- the extent of benthic habitats where the sediment thickness exceeds 0,01 m and the percentage of clays <40% and the percentage of sand > 20% (Figure 41).

The changes in the extent and quality of benthic habitats have been assessed based on the significance ratings described in Table 9. Appedix A provides detail on the extent of deposited sediments and the extent of benthic habitats meeting the specified benthic habitat quality criteria are reported for the various scenarios under consideration as a percentage (indicated with a significance rating) of the extent of habitats expected to occur under the reference conditions. In terms of these significance ratings (with respect to the extent of benthic habitats) it is not possible to discern between present state and Sc 2 to 5. However Sc 6 and 7, at a rating of -3, compared to the rating of -2 for Sc 2 to 5, are significantly worse than the other proposed scenarios.

Table 9. Freshwater, sediment and DRS inflows to the marine environment under the various proposed scenarios. The inflows for the various scenarios are expressed in terms of the significance ratings specified in Table 8. All ratings are relative to reference conditions.

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume[1]	0	-2	-2	-2	-2	-2	-3	-3
Total discharge of sediments[1]								
All sediments	0	-2	-2	-2	-2	-2	-3	-3
Sand	0	-2	-2	-2	-2	-2	-3	-3
Silt	0	-2	-2	-2	-2	-2	-3	-3
Coarse clays	0	-2	-2	-2	-2	-2	-3	-3
Fine clays	0	-2	-2	-2	-2	-2	-3	-3
Total discharge of DRS[1]	0	-2	-2	-2	-2	-2	-3	-3

1 Annual average of 66-year period.

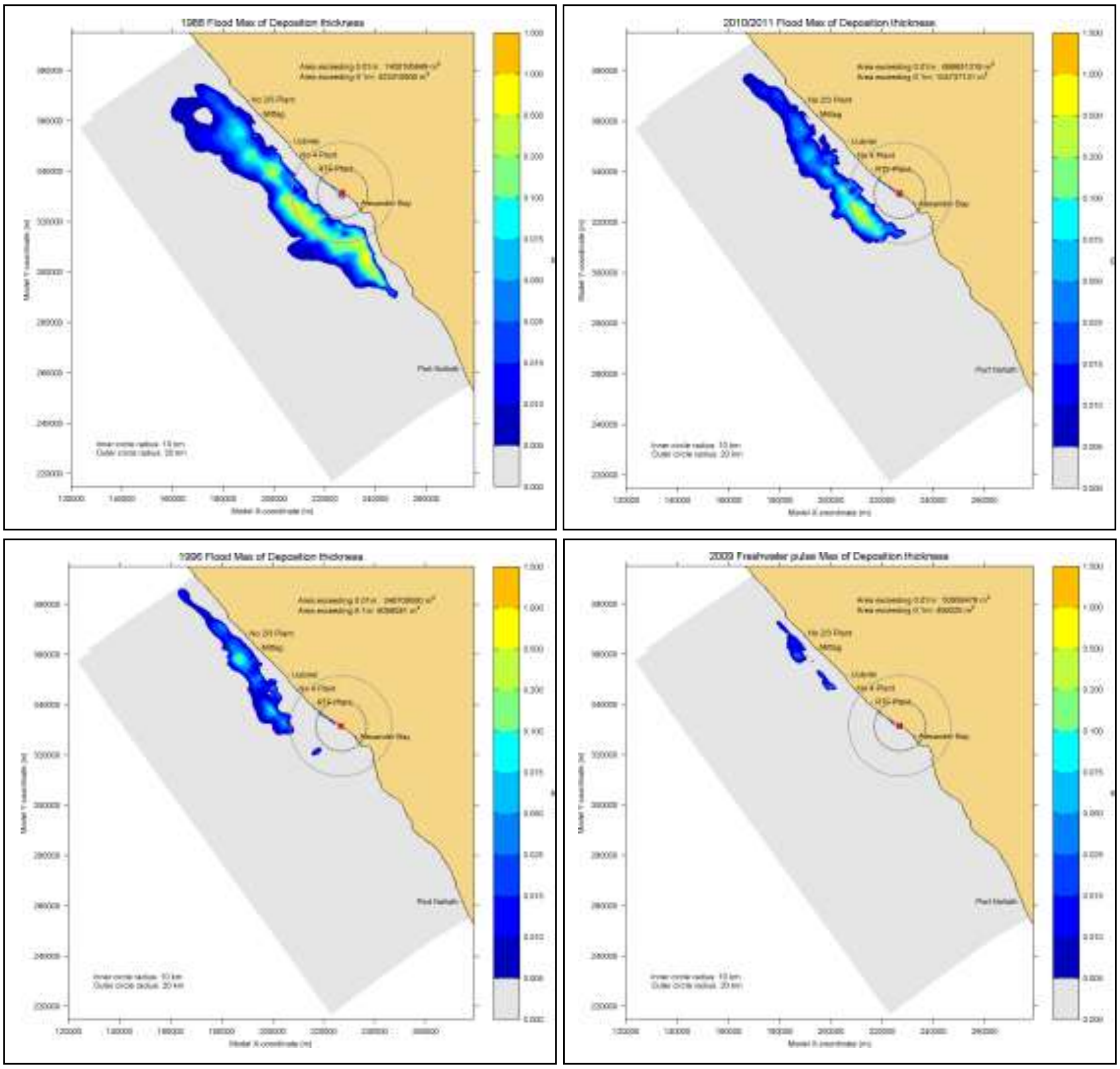


Figure 37. Maximum thickness of river sediments deposited on the seabed for i) a very large flood, ii) a large flood, iii) a small flood and iv) a high flow pulse of freshwater from the Orange River

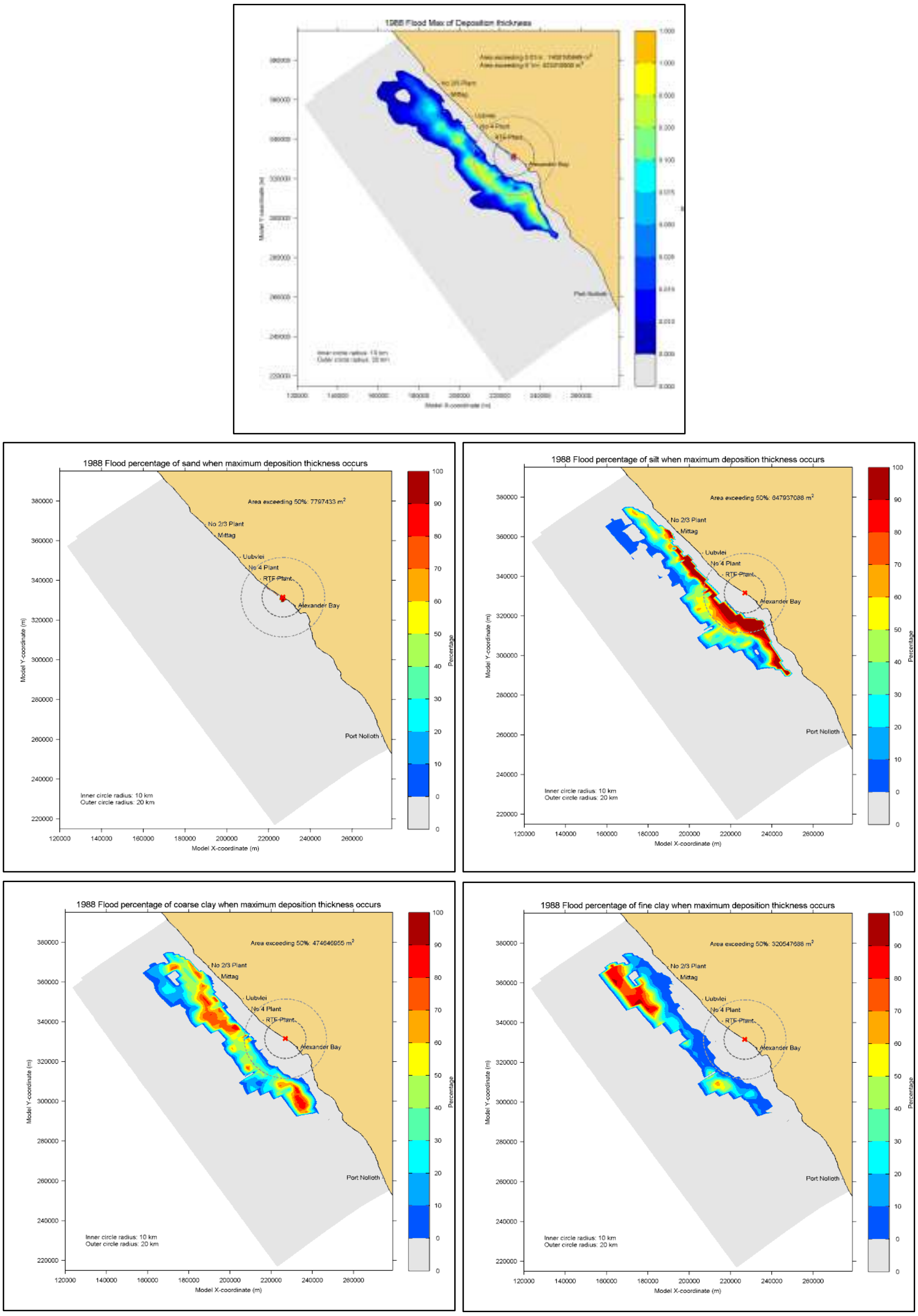


Figure 38. Plots of the maximum sediment thickness on the seabed (upper panel) and the percentage composition of sand (middle left panel), silt (middle right panel), coarse silt (lower left panel) and fine silt (lower right panel) constituting the sediment deposited on the seabed. The results are for a 'Very Large' flood (1988 flood).

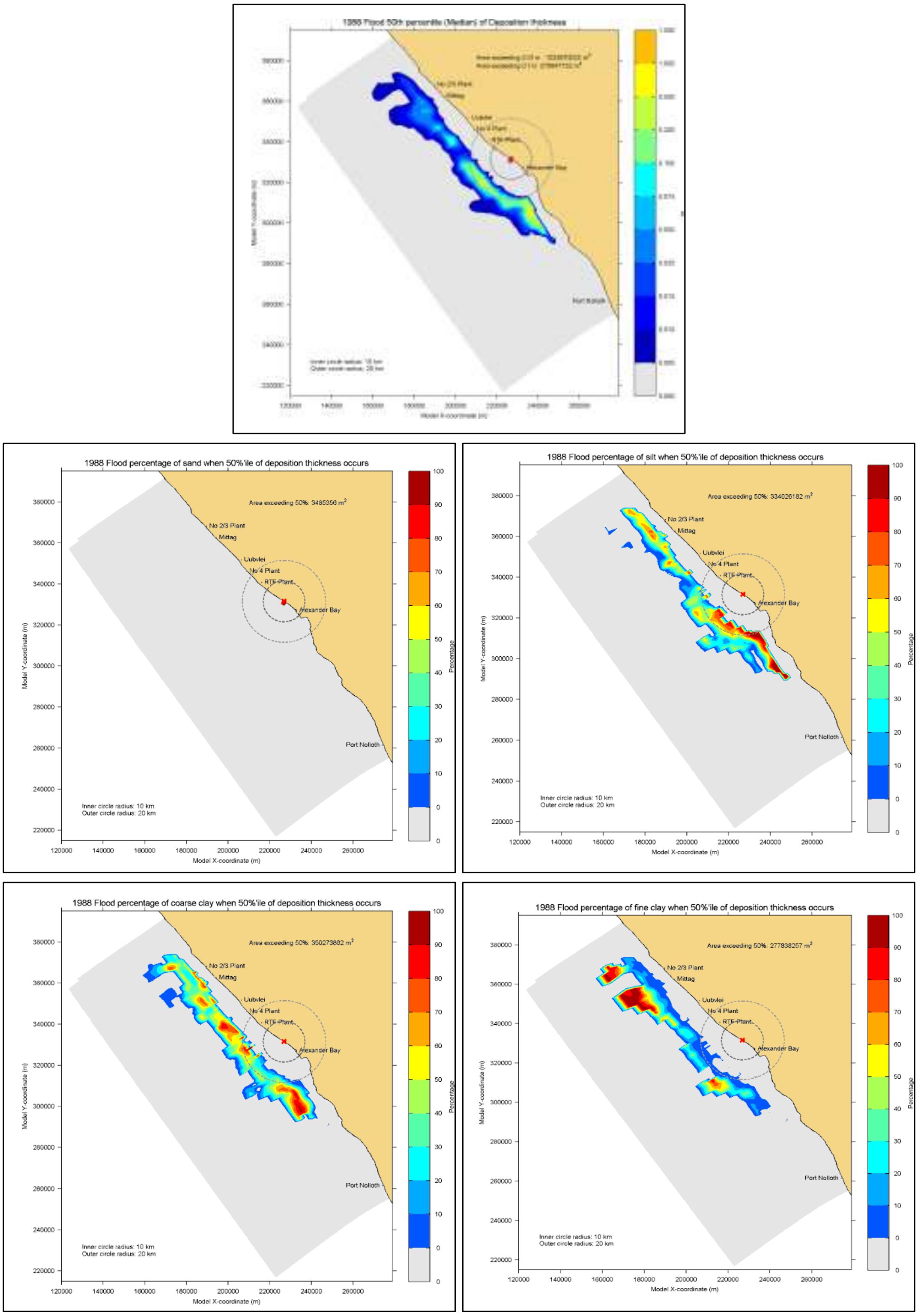


Figure 39. Plots of the median sediment thickness on the seabed (upper panel) and the percentage composition of sand (middle left panel), silt (middle right panel), coarse silt (lower left panel) and fine silt (lower right panel) constituting the sediment deposited on the seabed. The results are for a 'Very large' flood (1988 flood).

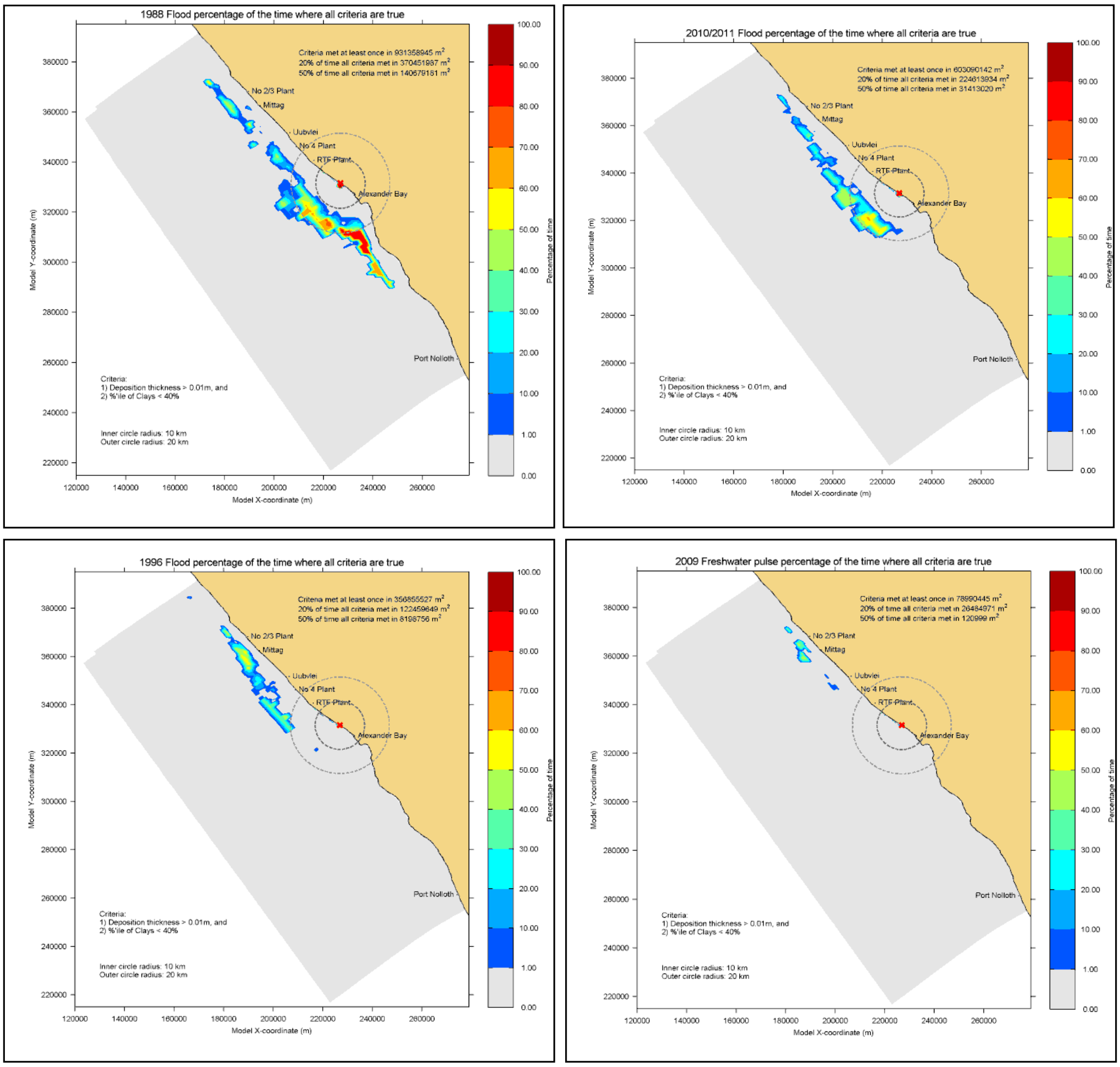


Figure 40. Maximum extent of benthic habitats where the sediment thickness exceeds 0.01 m and the percentage of clays <40% for simulations of i) a very large flood, ii) a large flood, iii) a small flood and iv) a high flow pulse of freshwater from the Orange River.

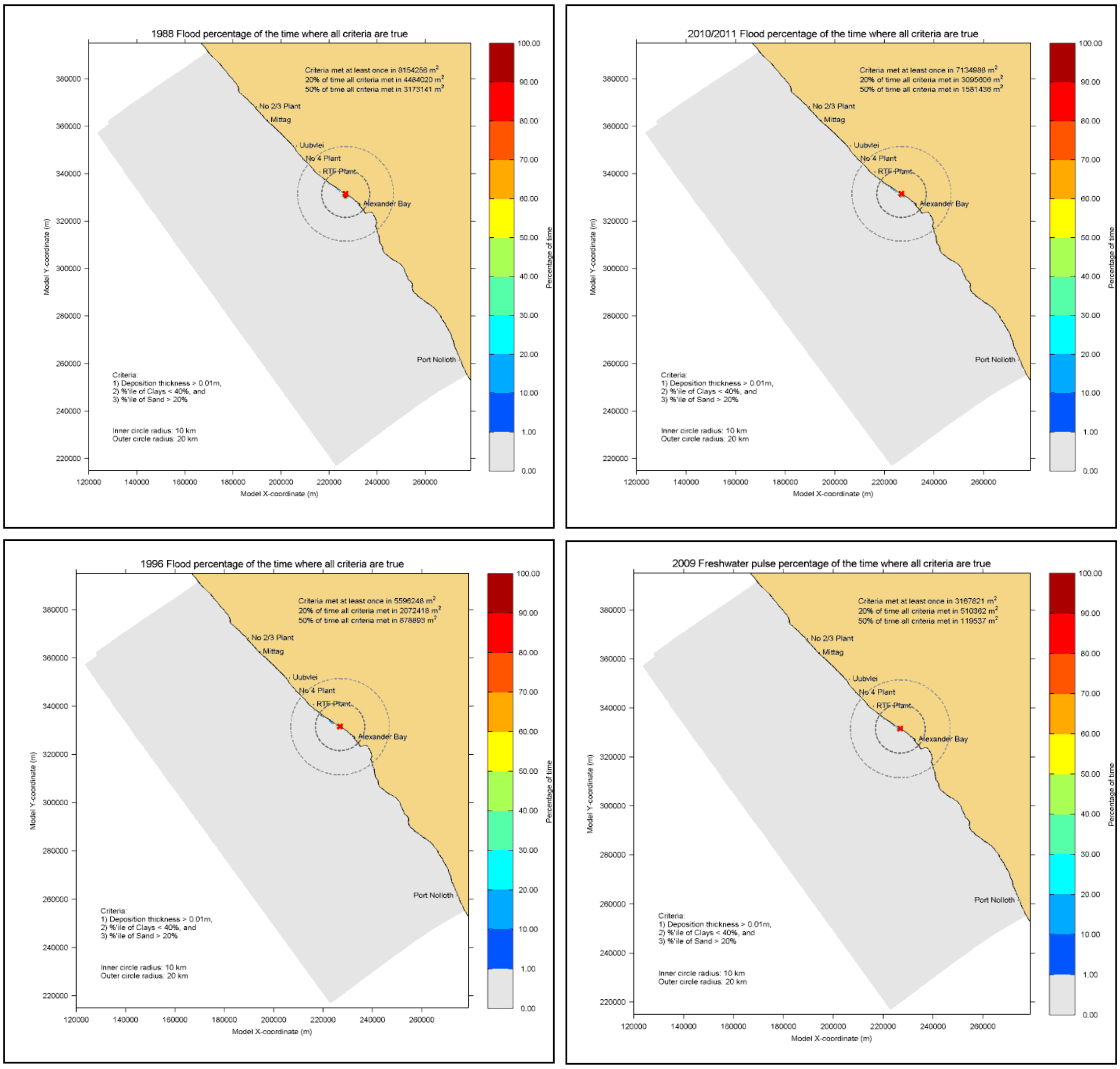


Figure 41. Maximum extent of benthic habitats where the sediment thickness exceeds 0.01 m, the percentage of clays <40% and the percentage of sand >20% for simulations of i) a very large flood, ii) a large flood, iii) a small flood and iv) a high flow pulse of freshwater from the Orange River.

9.5 Changes in pelagic habitats in the offshore marine environment

The changes in offshore pelagic habitats have been characterised using the distribution of water quality parameters such as salinity, TSS concentrations/turbidity and dissolved nutrients, particularly DRS.

9.5.1 Salinity

Two measures of low salinity habitat have been used to characterise the effect of freshwater inflows on the salinity of marine pelagic environments. The first is the **spatial** extent of surface waters where the salinity remains below specified threshold(s), the second being the **volume** of marine waters where the salinity remains below the specified threshold(s). Given the nature of the plume dynamics, i.e. that the low salinity waters in the flood water plumes in the marine environment are confined primarily to the surface waters (compare Figures 42 and 43), it is felt that the measure based on the volume of water not exceeding the specified thresholds would be a better measure of change in the extent of pelagic habitats associated with the various flood conditions. Two thresholds were considered, namely a large change (acute) in salinity represented by waters with a salinity <28 psu and a more subtle change (chronic) in salinity represented by waters with a salinity <33 psu. Similar to the measures of changes in the extent of benthic habitats, these spatial extents and volumes are reported in terms of the minimum (Figure 44) and medians (Figure 45) observed. For salinity the maximum change in pelagic habitat is represented by the minimum salinity measured as opposed to other water quality variables (e.g. TSS and DRS) where the maxima of the particular water quality variable under consideration represents the maximum extent of the changes in habitat.

The maximum extent of low salinity surface waters seems to decrease more or less linearly with flood size results while the extent of low salinities near the seabed are of limited extent for all but very large floods. The extent of fairly persistent low salinity surface waters (i.e. the 80%tile value where the salinity is lower than the values indicated in Figure 44 for 20% of the duration of the simulations), seems to decrease linearly with a decrease in flood size. However for small floods the area in which there is fairly persistent low salinity surface waters, is very limited. Contours of the median salinity values observed in the surface layer in the marine environment due to freshwater inflows from the Orange River (Figure 45) seem also to indicate a roughly linear decrease in extent with decreasing flood size. The median extent of low salinity surface waters in the marine environment is very limited for small floods and negligible for high flow pulses of freshwater from the Orange River.

The extent of changes in pelagic habitats (surface areas and volumes of low salinity waters) is reported in Appedix A (for the events simulated in the model simulations) and the significance ratings of these changes are reported relative to reference conditions in Table 10. In terms of these significance ratings (with respect to the extent of changes in pelagic habitats) it is not possible to discern between present state and Sc 2 to 5. However Sc 6 and 7, at a rating of -3, compared to the rating of -2 for Sc 2 to 5, are significantly worse than the other proposed scenario.

Table 10. Surface area of sediment deposition meeting the criteria listed in the table for the various high flow/flood scenarios under the various proposed future scenarios reported as a significance rating relative to reference conditions

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc4***	***Sc 5***	***Sc 6***	***Sc 7***
Total discharge of sediments[1] (M tonnes)	53.111	21.246	21.139	21.1	23.281	19.386	11.9	11.922
Surface area of sediment deposition >0.01 m								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max.	0	-2	-2	-2	-2	-2	-3	-3

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc4***	***Sc 5***	***Sc 6***	***Sc 7***
Surface area of sediment deposition >0.01 m and <40% clay								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max.	0	-2	-2	-2	-2	-2	-3	-3
Surface area of sediment deposition >0.01 m and <40% clay and >20% sand								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max.	0	-2	-2	-2	-1	-2	-3	-3

1 Annual average of 66-year period.

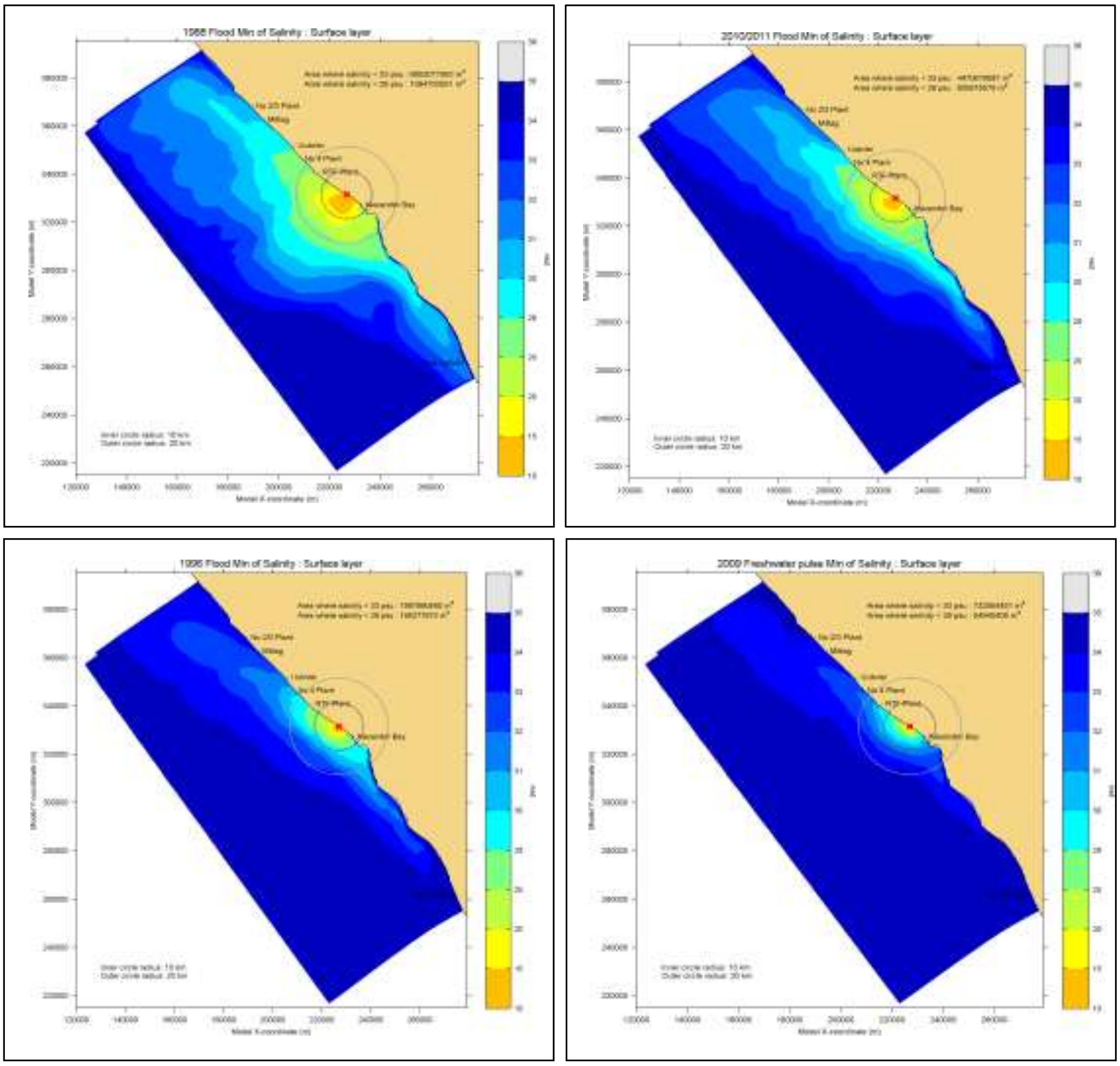


Figure 42. Contours of the minimum salinity observed in the surface layer in the marine environment due to freshwater inflows from the Orange River

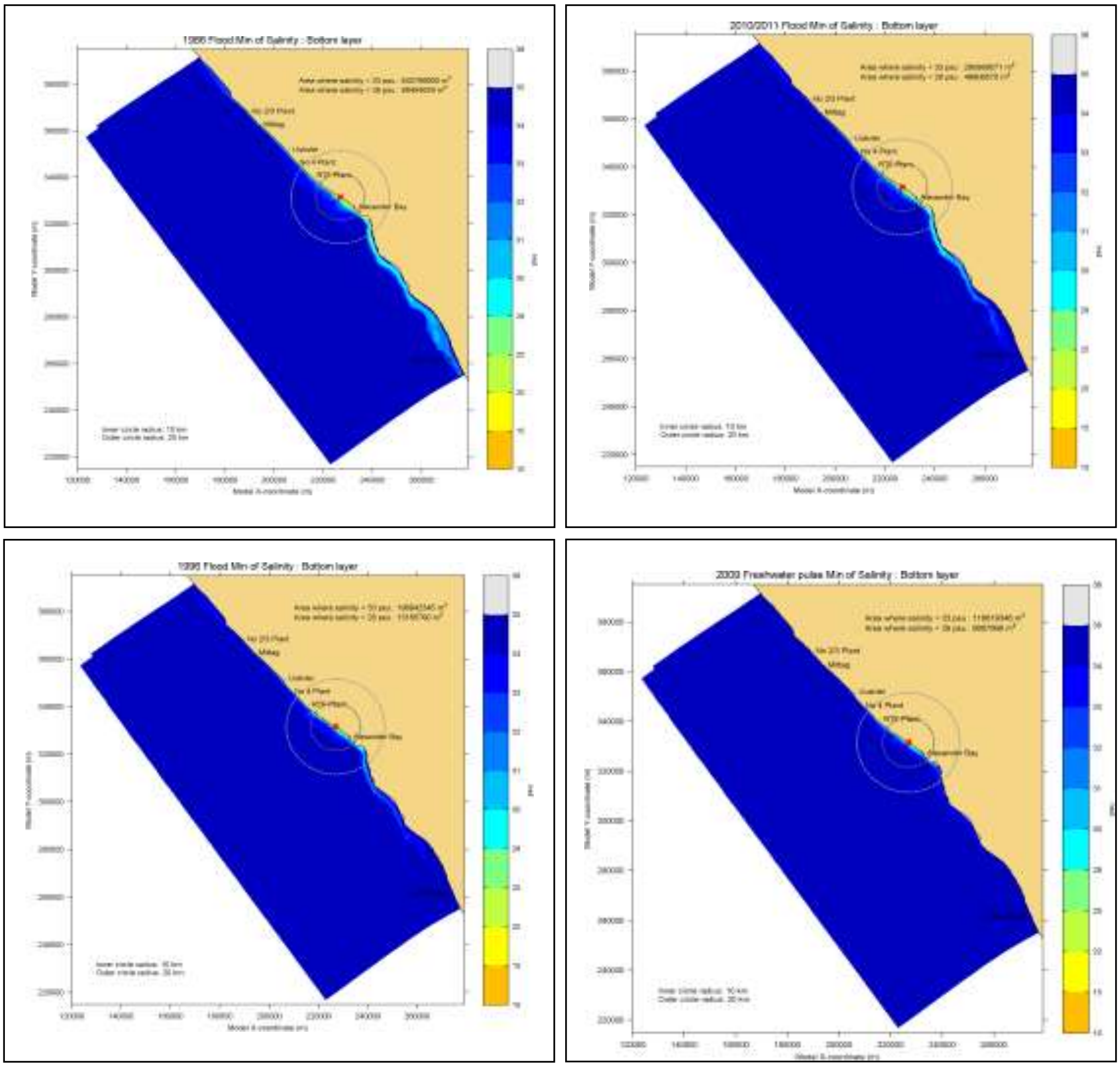


Figure 43. Contours of the minimum salinity observed in the near bottom layer in the marine environment due to freshwater inflows from the Orange River

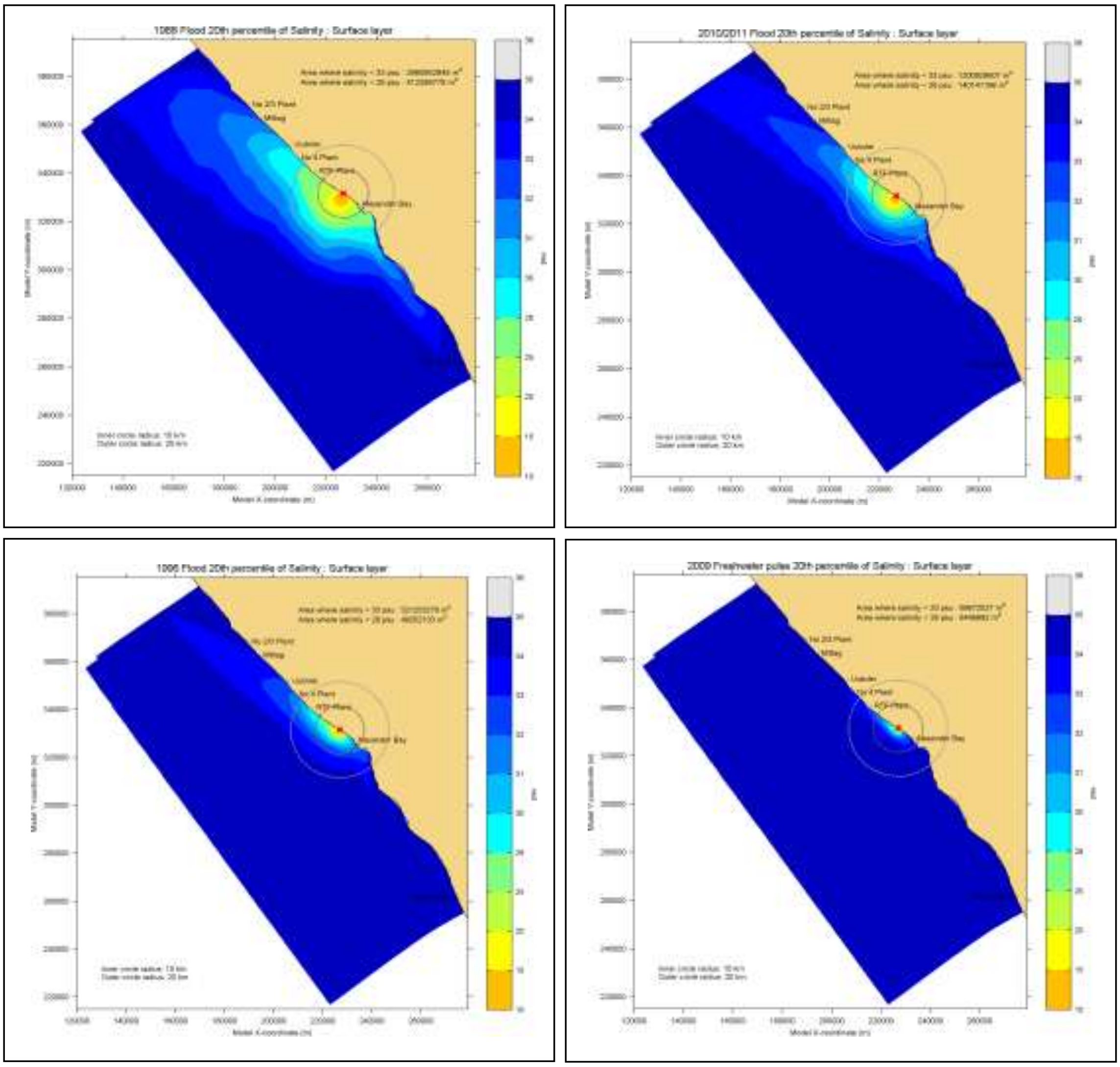


Figure 44. Contours of the 20 percentile salinity values observed in the surface layer in the marine environment due to freshwater inflows from the Orange River

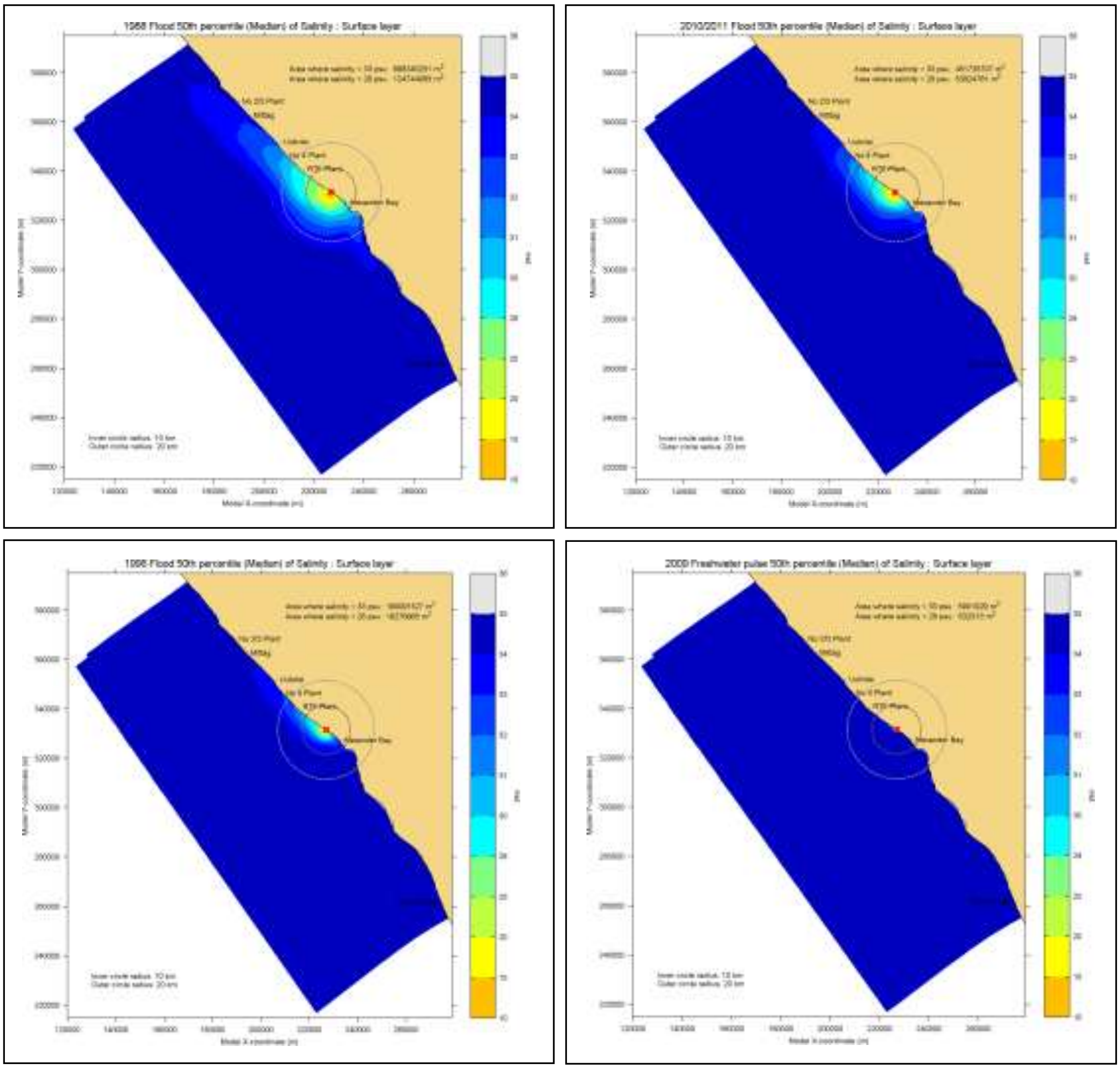


Figure 45. Contours of the median salinity values observed in the surface layer in the marine environment due to freshwater inflows from the Orange River

9.5.2 Nutrients

Compared with concentrations typically measured in seawater in the area (<1,00 μg/ℓ), river inflow can be a significant source of DRS (average 4,800 μg/ℓ) to the adjacent marine environment. Consequently the spatial extent of surface waters as well as the volume of marine waters exceeding the typical ambient concentrations of DRS in the marine environment by 20% and 75% have been calculated (i.e. where the resulting DRS concentrations are 1,2 and 1,75 times greater than the ambient DRS concentrations in the ocean). These areas and volumes are considered to represent the extent to which river inflows create and enhanced nutrient environment with respect to DRS in the adjacent marine environment. Similar to the measures of changes in the extent of benthic habitats, these spatial extents and volumes are reported in terms of the median, 80 percentile and maximums observed.

Nutrient distributions was not been explicitly simulated in the model. However based on dilution arguments, the contours for a salinity of 28 psu represents DRS concentration contours of 1,75 times ambient DRS concentrations in the marine environment (i.e. a 75% elevation in DRS compared to ambient due to the freshwater inflows from the Orange River). Similarly salinity contours of 33 psu roughly approximate DRS elevations of 25% above ambient. For this reason figures of all the relevant information can be gleaned from the distributions and spatial extents and volumes of salinities reported in Table 10. All that has to be remembered is that area/volumes associated with salinity that remains below 28 psu represent areas and volumes where DRS in the marine environment are enhanced by more than 75% compared to ambient DRS concentrations in the marine environment adjacent to the Orange River mouth. Similarly, the area/volumes associated with salinity that remains below 33 psu represent areas and volumes where DRS in the marine environment are enhanced by more than 25% compared to ambient DRS concentrations in the marine environment adjacent to the Orange River mouth.

As a consequence of the above relationships, the conclusions for elevated DRS in the marine environment are the same as those for reductions in salinity. The corresponding significance ratings of the changes in DRS against reference are provided in Table 11. Based on the results it is not possible to discern between present state and proposed Sc 2 to 5. However Sc 6 and 7, at a rating of -3, compared to the rating of -2 for Sc 2 to 5, are significantly worse than the other proposed scenarios.

Table 11. Changes in the extent of low salinity waters (<28 psu and <33psu) expressed as both surfaces areas of low salinity surface waters and volumes of lows salinity waters (i.e. includes a depth component) for the various high flow/flood scenarios under the various proposed future scenarios, reported as a significance rating relative to reference conditions

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume[1] (Mm³)	0	-2	-2	-2	-2	-2	-3	-3
Area of sea surface where salinity < 33 psu[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-2	-2	-3	-3
Area of sea surface where salinity < 33 psu[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-2	-2	-3	-3
Area of sea surface where salinity < 33 psu[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3

Scenario	*Reference*	*Present*	*Sc 2*	*Sc 3*	*Sc 4*	*Sc 5*	*Sc 6*	*Sc 7*
Max	0	-2	-2	-2	-2	-2	-3	-3
Area of sea surface where salinity < 33 psu[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-2	-2	-3	-3

1 Annual average of 66-year period.

9.5.3 Total suspended solids

The extent of waters with high TSS concentrations or highly turbid waters also is considered to be a measure of pelagic habitat quality. The effects of elevated TSS concentrations (and turbidity) may be both positive and negative, however excessively high water column turbidity (e.g. TSS >100 mg/ℓ) is considered to constitute a degraded environment for most species in the region (EMBECON, 2004). More subtle changes are expected at lower TSS thresholds (e.g. 20 mg/ℓ). Consequently the spatial extent of surface waters as well as the volume of marine waters exceeding the TSS concentrations of 100 mg/ℓ and 20 mg/ℓ have been calculated. These areas and volumes are considered to represent the extent to which river inflows have discernible (positive or negative) TSS or turbidity effects in the marine environment. Similar to the measures of changes in the extent of benthic habitats, these spatial extents and volumes are reported in terms of the median, 80%tile and maximum TSS concentrations observed. In the model all TSS concentrations observed are related to the river inflows as the background or ambient TSS concentrations of the marine waters have not been simulated in the model.

The spatial extents and volumes of elevated TSS concentrations are reported in terms of the median, 80%tile and maximum TSS concentrations observed. The maximum extent of changes in pelagic habitats due to sediment inputs from the Orange River is represented by the extent of the maximum TSS concentrations (and turbidity) in the water column. As expected the maximum extent of elevated TSS concentrations (and turbidity) occurs near the seabed rather than in the surface waters. It should be noted that variability in TSS concentrations can be complex and long lasting due to the continual re-suspension and redistribution of accumulated sediments that occurs during and after storm events.

Elevated TSS concentrations occur mainly to the north of the Orange River mouth. However there is also elevated TSS concentrations observed to the south of the Orange River mouth that most probably are due to the initial plume dynamics occurring during the peak inflows of freshwater into the marine environment. The theoretical behaviour for a large freshwater plume discharged into a relatively quiescent offshore marine environment is that the plume should move southwards (Shillington et al., 1990). The bias of high TSS concentrations towards the north is associated with the strong south to southeasterly winds that prevailed during the model simulation period. The elevated TSS concentrations extend further offshore in the bottom waters that in the surface waters. This is related to the re-suspension, advection and re-distribution of the bottom sediments (mostly into deeper waters) over time. The elevated TSS concentrations in the surface waters are related primarily to the initial plume dynamics which result in fairly extensive distribution of low salinity but high turbidity surface waters. Small floods and high freshwater inflow pulses result in significantly less extensive elevations in TSS concentrations in the surface waters, however the extent of elevated turbidity in the bottom waters is fairly extensive for all but the smallest flows simulated (i.e. high freshwater inflow pulses) where elevated turbidity is restricted mainly to the near-shore areas.

The model simulations were of limited duration therefore the model outputs do not necessarily fully reflect the persistence of this elevated turbidity. It is expected that elevated turbidity will persist near the seabed over a fairly extensive area for relatively long periods after flood events (as can be seen from the satellite imagery). Persistent elevation of turbidity in the surface waters is expected only in the near-shore and even then will only persist until the (predominantly fine) sediments have been re-distributed into deeper waters (water depths deeper than -25 to -30 m chart datum) by wind and wave-driven turbulence. The elevated turbidity for small floods and freshwater pulses is indicated by the modelling results to extend only northwards. This is expected as nearshore flows are predominantly northwards, however the model results may be biased in that the offshore conditions simulated in the model are summer conditions when predominantly south to southeasterly flows predominate, i.e. the model does not fully take into account potential south-easterly wind-driven flows driven north-westerly winds associated with larger coastal lows and storms during the winter months.

Based on the findings in EMBECON (2004), the higher the elevation of TSS concentrations (and therefore turbidity) and the more extensive the elevated TSS concentrations; the greater the impacts in the marine environment. However it should be noted that changes in turbidity can be both detrimental and beneficial, depending on the specific ecological effects being considered. In keeping with the principle that changes from the status quo are deemed to be detrimental, the indices used to assess the effects of changes in TSS concentrations on the marine environment can be interpreted as being highly detrimental for highly significant increases or decreases in the indices measuring change in TSS concentrations, moderately detrimental for moderately significant increases or decreases in these indices and of more limited detriment for changes that are considered to be discernable but not large enough to be of moderate to high significance.

As for the other metrics used in this study (e.g. low salinities and higher dissolved reactive silicates), the significance ratings of the changes in TSS concentrations, normalised to reference conditions (see Table 12, Figures 46 to 51), indicate that not possible to discern between present state and proposed Sc 2 to 5. However Sc 6 and 7, at a rating of -3, compared to the rating of -2 for Sc 2 to 5, are significantly worse than the other proposed scenarios.

Table 12. Extent of surface and bottom waters with elevated turbidity (> 100 mg/ℓ and >20 mg/ℓ) for the various high flow/flood scenarios under the various proposed future scenarios , reported as a significance rating relative to reference conditions.

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume[1] (M m^3)	0	-2	-2	-2	-2	-2	-3	-3
Area where TSS >100 mg/ℓ in the surface waters[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-1	-2	-3	-3
Area where TSS >100 mg/ℓ in the near bottom waters[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-1	-2	-3	-3
Area where TSS >20 mg/ℓ in the surface waters[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-1	-2	-3	-3
Area where TSS > 20 mg/ℓ in the near bottom waters1								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-1	-2	-3	-3
Max	0	-2	-2	-2	-1	-2	-3	-3

1 Annual average of 66-year period.

9.6 Conclusion

The changes in the freshwater, sediment and dissolved reactive silicates inputs into the marine environment have been assessed based on the significance ratings described in Table 9. In terms of these significance ratings (with respect to inflows to the marine environment) it is not possible to discern between present state and proposed Sc 2 to 5. However Sc 6 and 7, at a rating of -3, compared to the rating of -2 for Sc 2 to 5, are significantly worse than the other proposed scenarios.

The above results (indices of change) characterise only the changes of freshwater and associated fluxes (sediments, nutrients, etc.) into the marine environment. This information can be used to assess potential changes in nearshore and offshore marine environments based on expert opinion and/or model simulated changes in water quality and/or sediment-related marine habitats.

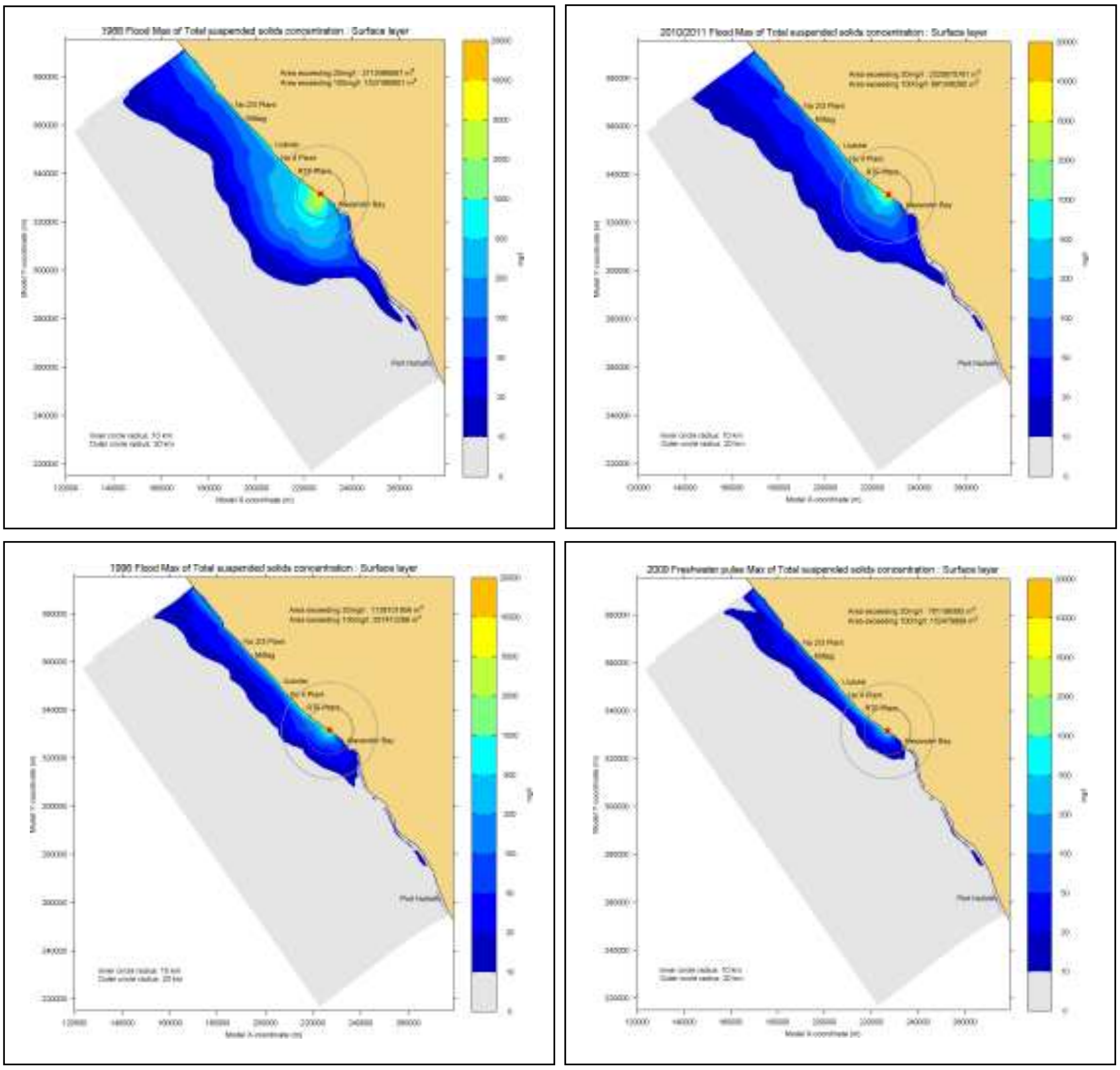


Figure 46. Contours of the maximum TSS concentrations observed in the surface layer in the marine environment due to freshwater inflows from the Orange River

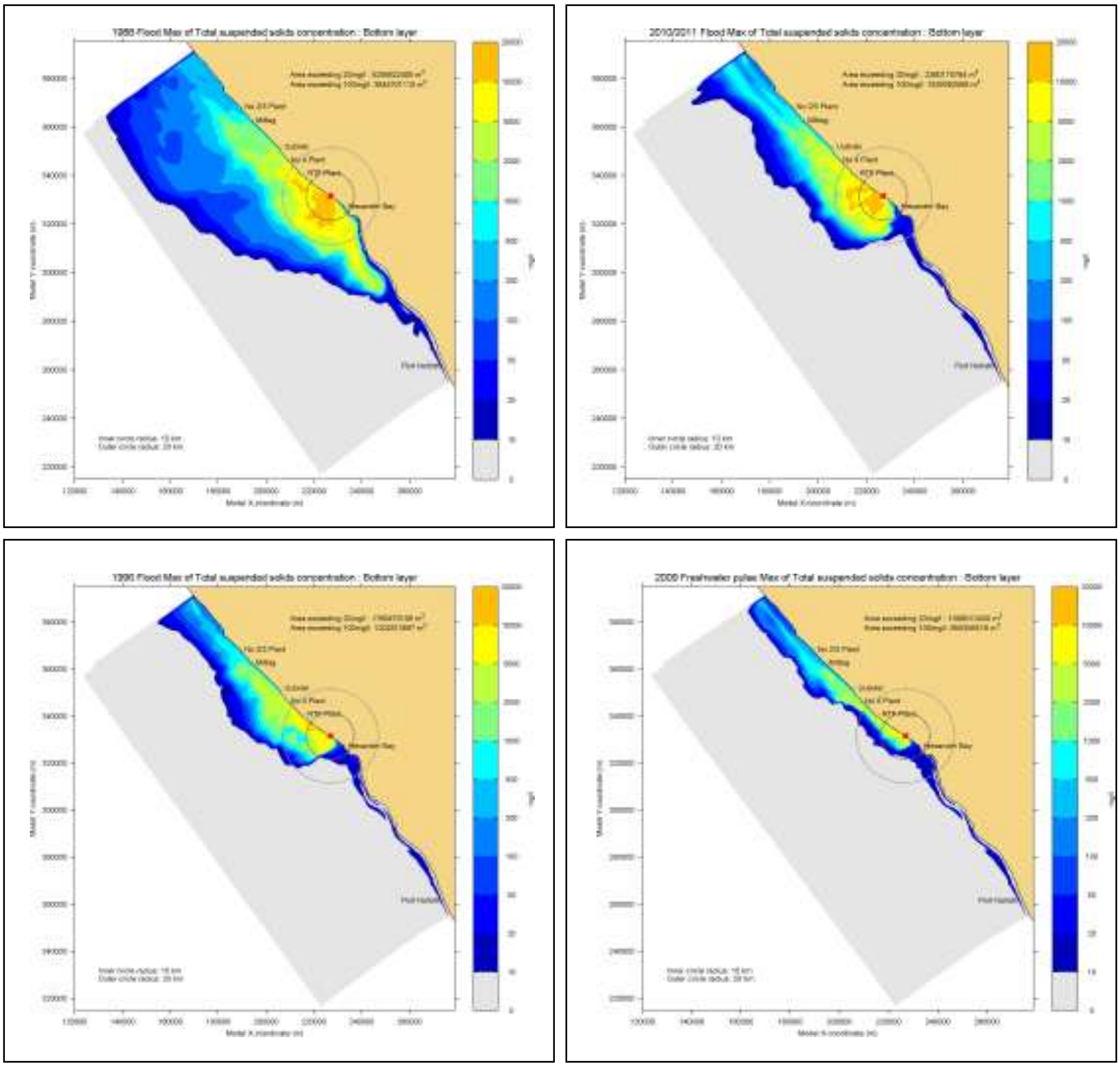


Figure 47. Contours of the maximum TSS concentrations observed in the near the seabed in the marine environment due to freshwater inflows from the Orange River

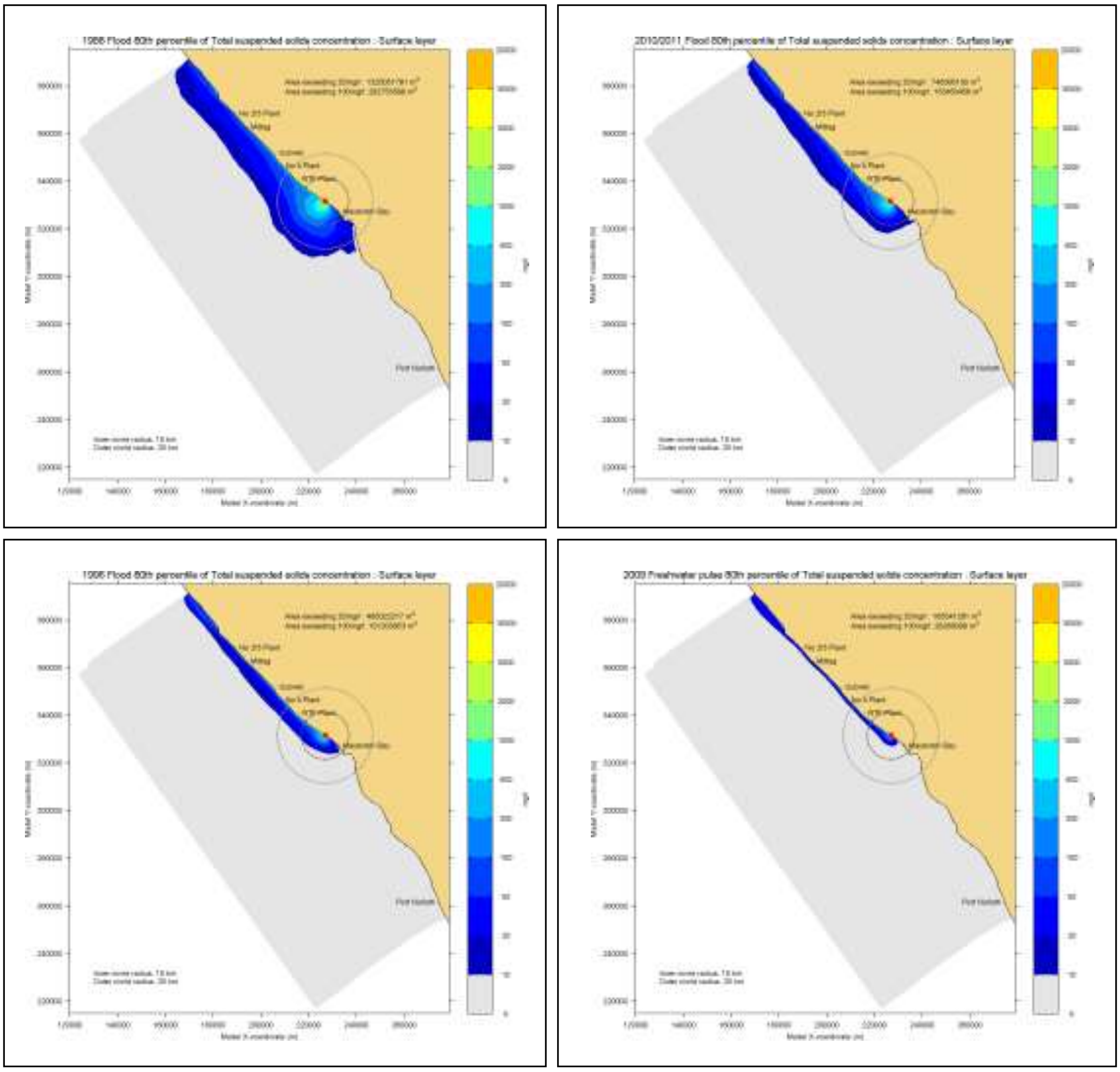


Figure 48. Contours of the 80 percentile TSS concentrations observed in the surface layer in the marine environment due to freshwater inflows from the Orange River

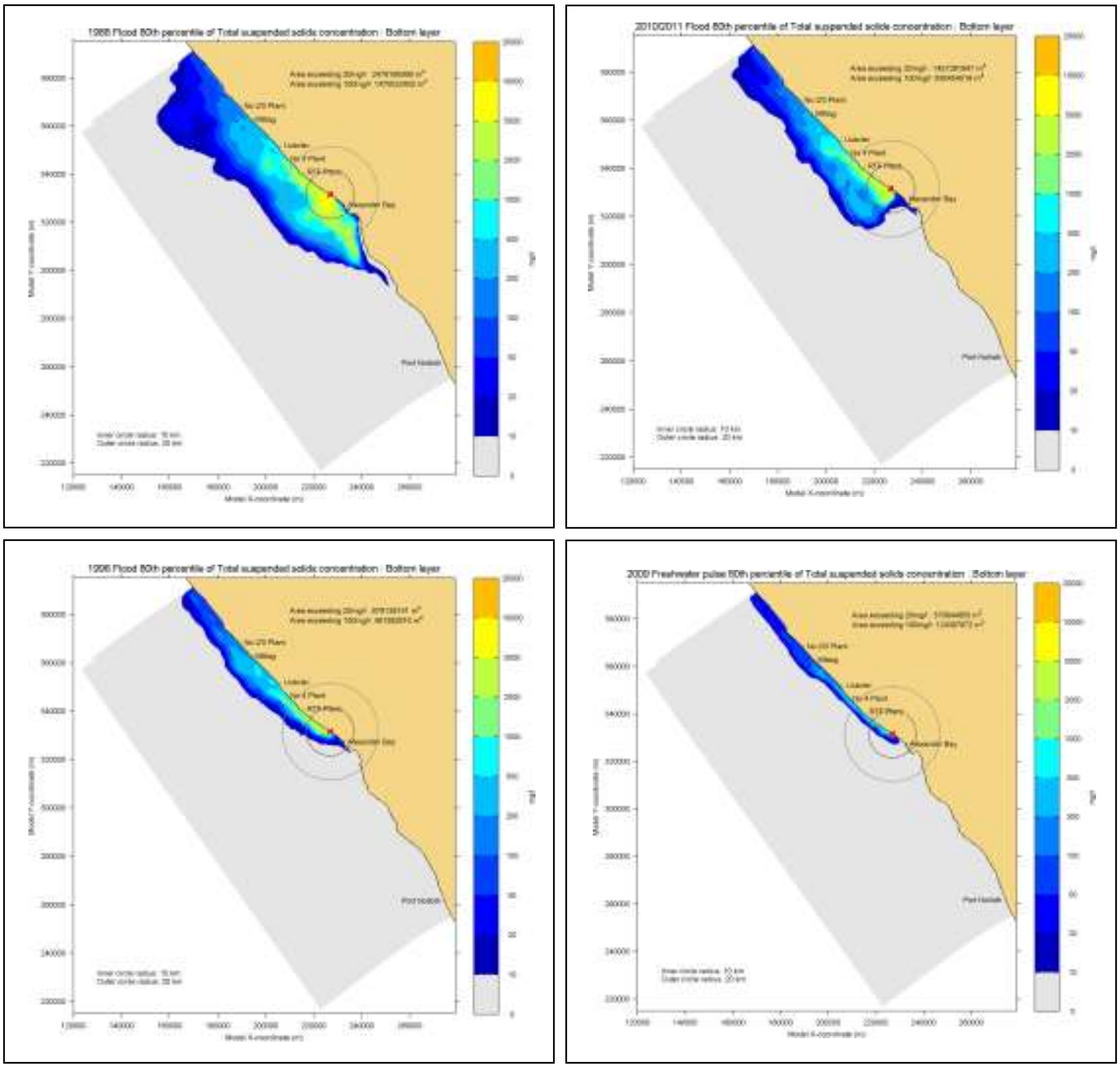


Figure 49. Contours of the 80 percentile TSS concentrations observed in the near the seabed in the marine environment due to freshwater inflows from the Orange River

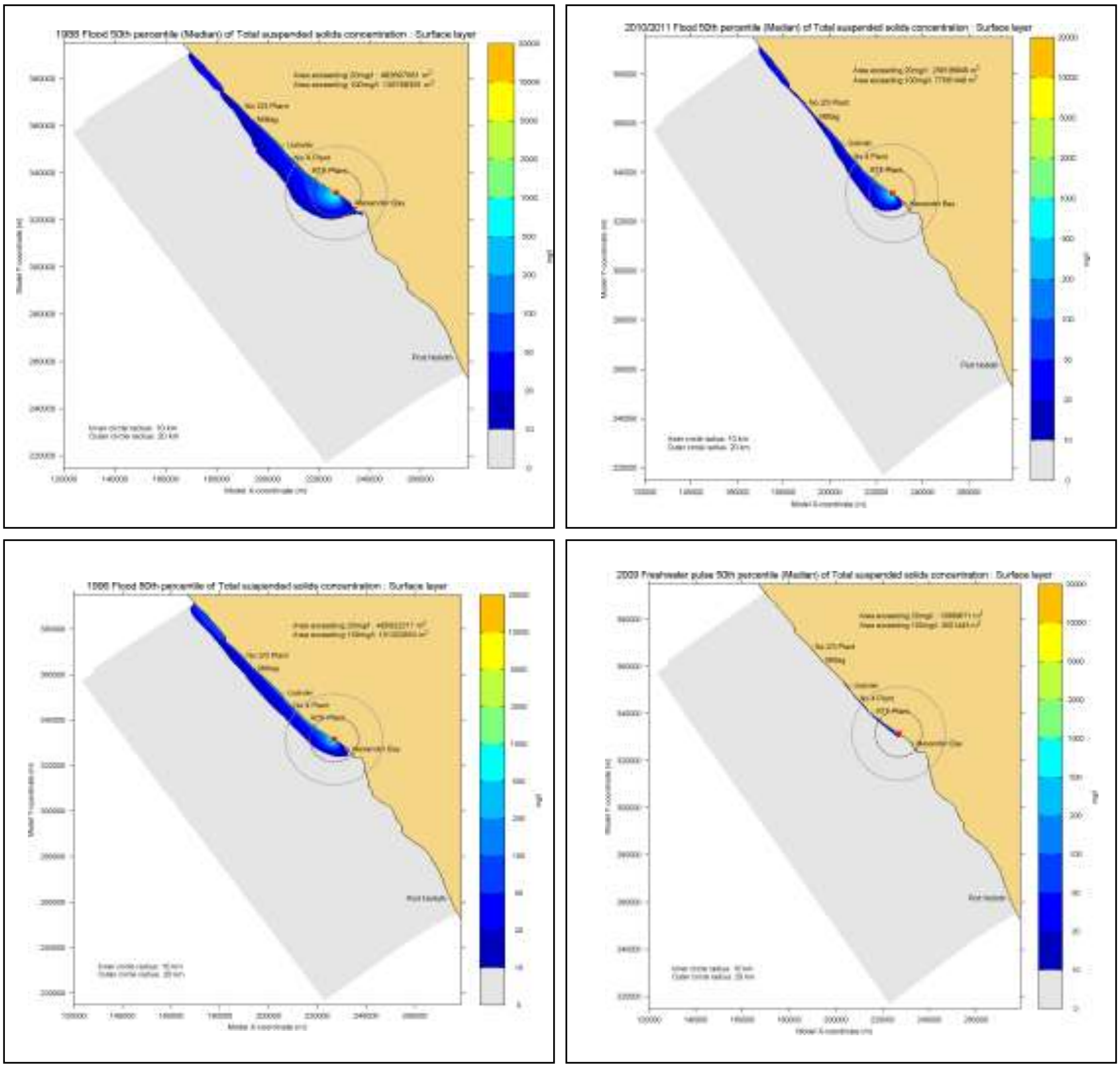


Figure 50. Contours of the median TSS concentration observed in the surface layer in the marine environment due to freshwater inflows from the Orange River

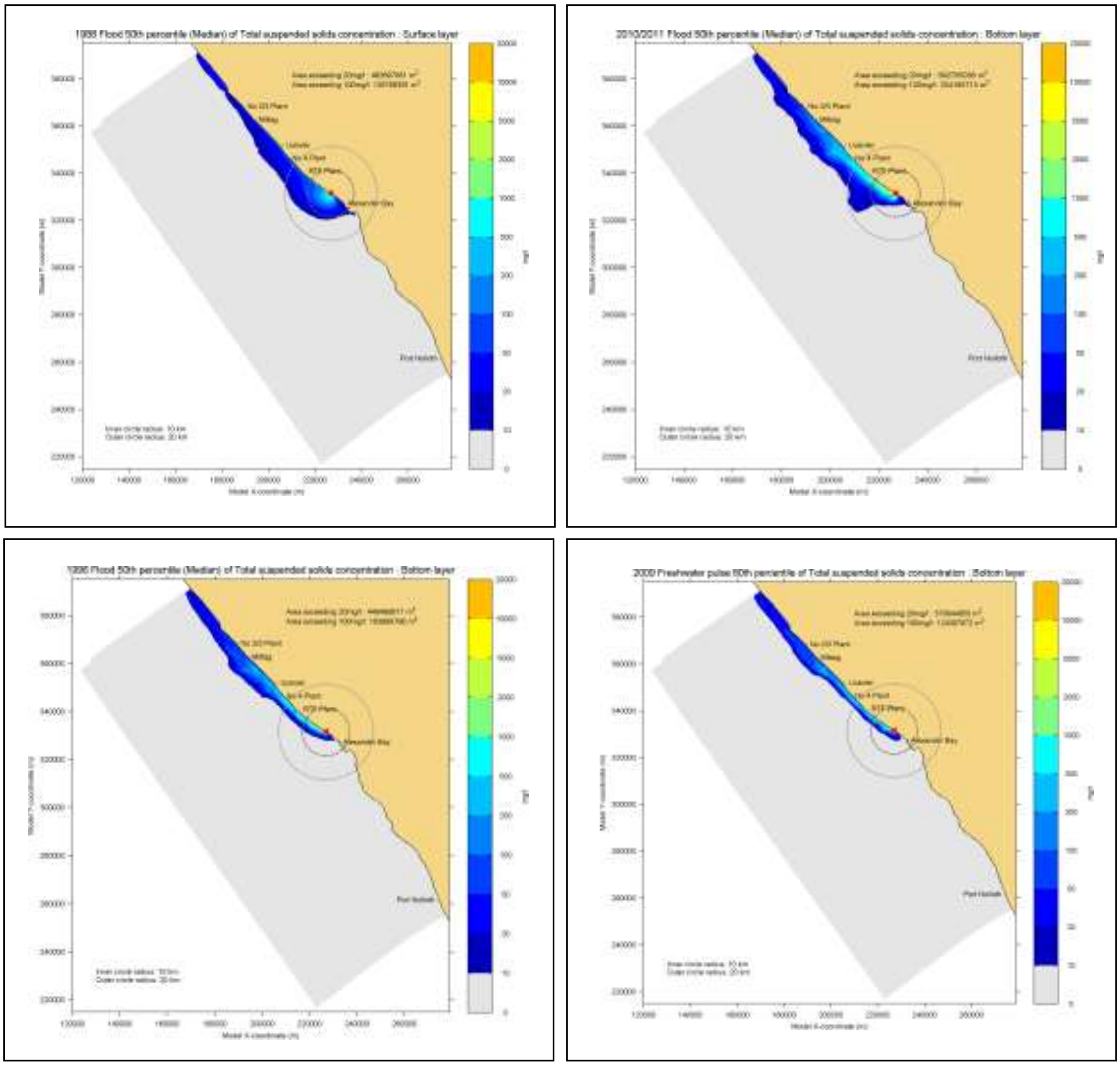


Figure 51. Contours of the median TSS concentration observed in the near the seabed in the marine environment due to freshwater inflows from the Orange River

10 Implications of flow alteration on selected biological components

Types of fish (and fisheries) response to changes in freshwater inflow to the marine environment fall into four broad categories (Lamberth et al., 2009):

1. Apparent negative responses to reduced flow, that are most likely due to rainfall/climate patterns throughout a biogeographical region rather than local flow rates.
2. Negative responses to local reduced flows that are real, e.g. reduced flow from the catchment will result in reductions in turbidity, preferred sediments, nutrient loads, phyto- and zooplankton production and ultimately reduced biomass and catches.
3. Cases of zero or negligible response, either positive or negative, to changes in flow. However, the lack of response may be an artefact of the fact that catches are low and reporting correspondingly poor. The proviso is that frequently-caught species providing the bulk of the catch are by default those providing the better data and exhibiting significant relationships with flow.
4. Situations where flow reduction has a positive effect on catches. Correlations like these are likely to be less due to ecological drivers than to various aspects of fleet behaviour (and fisheries management).

Sediment movement and deposition rates are closely linked to water depth and flow speed (Hiscock, 1983; Overnell & Young, 1995). Consequently, high levels of siltation but reduced motion of sediments may be expected in areas with low degrees of water movement (e.g. beyond the wave-base on the Orange submarine delta). In contrast, sediment movement, deposition and resuspension are often combined in wave-dominated intertidal and shallow subtidal habitats characterised by highly turbulent near-bottom flows. The effects of these physical variables on the benthos are different, however, and a distinction between them should thus be made. Below follows a review of the responses of marine biota to the key abiotic drivers associated with river discharges, namely 1) elevated suspended sediment concentrations; 2) sediment deposition; and 3) depressed salinities.

10.1 Effects of elevated suspended sediment concentrations

The tolerance or preference of marine fauna and flora to increases or decreases in suspended sediment concentrations is variable, with tolerance levels ranging from tens of mg/ℓ to several thousand mg/ℓ. Sediment suspension loads the water with inorganic suspended particles, which may affect plants and animals directly or indirectly, either lethally or sub-lethally. Furthermore, the probability of a detrimental effect is likely to increase with both increasing concentration and duration of exposure (Newcombe & MacDonald, 1991; Newcombe & Jensen, 1996; Clarke & Wilber, 2000). Certain organisms may display an ability to adapt to elevated suspended sediment over time, but there will be a concentration above which they are unable to respond in this manner. Despite the paucity of information on the specific physiological tolerance levels of southern African intertidal and subtidal species to increased and prolonged suspended sediment concentrations, it is safe to assume that most of the marine biota of the highly dynamic Benguela coastal region are well adapted to cope with short-term increases in suspended sediment concentrations, and many of those frequenting mixed shores can actually tolerate long-term burial by re-deposited sediments. This review is, however, based primarily on work done in other regions, and on species which do not occur on the southern African West Coast.

10.1.1 Phytoplankton

A reliable relationship between suspended matter and turbidity is often difficult to establish due to variations in the characteristics of suspended particles. Nonetheless, one of the more apparent effects of increased suspended sediment concentrations and consequent increase in turbidity, is a reduction in light penetration through the water column with potential adverse effects on the photosynthetic capability of phytoplankton and marine algae (Johnson, 1981; Poopetch, 1982; Kirk, 1985; Parsons et al., 1986a,b; Monteiro, 1998; O'Toole, 1997).

Parsons et al. (1986a,b) showed that addition of mine tailings at 30 mg/ℓ and 300 mg/ℓ to experimental mesocosms reduced the photic zone significantly and delayed the onset of phytoplankton blooms, with a shift in the phytoplankton biomass spectrum towards small microflagellates over the three week experimental period. Monteiro (1998), who modelled the effects of suspended sediments on the depth of the photic zone, deduced that an increase in suspended sediment concentrations from 50 mg/ℓ to 100 mg/ℓ reduced the photic depth from 2 m to 0,5 m. At 10 mg/ℓ the photic depth will be approximately 10 m. The non-linear relationship between suspended sediment concentrations and light extinction dictates that the effect on the photic depth of an increase in suspended sediment concentration is highly dependent on the background level of turbidity. In turbid systems (>100 mg/ℓ) any further increase in turbidity will be insignificant, whereas in non-turbid systems even moderate increases in suspended sediments result in a marked reduction in absolute photic depth. Along the southern African West Coast, where turbid water is a reference occurrence, inhibition of primary production in the near-shore environment is likely to be negligible.

10.1.2 Bivalve molluscs

Suspension-feeding bivalves from temperate shallow water habitats are typically exposed to a food supply that fluctuates substantially in concentration and nutritional composition. They not only feed on particles suspended in the water column, but (in the case of infaunal species) may also ingest surfacial deposits available at the water–sediment interface (Kamermans, 1994). Bivalve suspension-feeders thus show remarkable physiological adaptations in their levels of filtration, feeding and digestion, acting to maximize energy uptake under conditions of high turbidity and concomitant low food value. These adaptations include the capacity for adjusting the duration and intensity of filtration and pseudofaeces production[2] rates in response to changes in particle concentration and composition, the ability of the pallial organs to sort particles of different value, and achieving a positive net food absorption when ingesting matter of low organic content.

Many suspension feeders benefit from a diet supplemented with natural sediments. Studies have shown that the addition of sediment to seston through natural sediment transport or bioturbation can increase clearance rate, absorption efficiency and growth rates in some bivalve species (Winter, 1976, 1978; Kiørboe et al., 1980, 1981; Møhlenberg & Kiørboe, 1981; Bayne et al., 1987; Grant et al., 1990; Navarro et al., 1996; MacDonald et al., 1998). In contrast, however, many authors have demonstrated reduced feeding activity and/or growth in response to high suspended loads (Stuart, 1982; Robinson et al., 1984; Bricelj & Malouf, 1984a,b; Bricelj et al., 1984b; Grant et al., 1990). Suspended inorganic material can also enhance food availability to suspension-feeding organisms by providing an extensive surface for the adsorption of dissolved organic material and microorganism colonization (Grant et al., 1997). The efficiency with which bivalves can select organics from the matter retained on the gills, and subsequently assimilate them, varies between species. In general, however, the amount of organic matter ingested and assimilated declines with increasing sediment concentrations, and the amount of pseudofaeces produced increases with increased inorganic content of the seston.

If the concentration of suspended sediments is raised above a certain threshold, the particulate inorganic matter has the effect of diluting the available food with a consequent decline in energy available for production. Many suspension-feeding bivalves counteract this food dilution by selectively rejecting excess filtered material and particles of poor nutritive value as pseudofaeces (Bricelj & Malouf, 1984a,b; Bayne et al., 1989, 1993; Grant & Thorpe, 1991; Navarro et al., 1994; Iglesias et al., 1996; Hawkins et al., 1998; Wong & Cheung, 1999; Urrutia et al., 2001; amongst others). This prevents the animal's ingestive capacity from being exceeded, but incurs an energetic cost. Filtration rate and pseudofaeces production rate generally increase with increasing particle concentration up to a threshold level above which the filter-feeding mechanism becomes overloaded, and filtration rate again declines in order to maintain assimilation rates and minimize energy loss.

Numerous authors have postulate that species able to regulate ingestion primarily by producing pseudofaeces are better adapted to cope with high suspended sediment loads, than species that control ingestion mainly by reducing clearance rates (Bricelj & Malouf, 1984a,b; Bacon et al., 1997; MacDonald et al., 1998). The ability of a species to deal effectively with elevated concentrations of particulate inorganic matter (PIM) may thus determine its success in a particular environment. Many infaunal species, for example, appear to have evolved adaptive feeding behaviour to cope with high concentrations of poor quality particles including surficial deposits or sediment slurries, in contrast to epifaunal species living on or just above the bottom and feeding predominantly on particles suspended higher in the boundary layer.

2 Pseudofaeces production is a process of particle selection whereby less nutritious particles are rejected and the quality of the ingested material improved proportionately.

The effects of elevated suspended sediment loads on juvenile and adult bivalves thus occur mainly at the sublethal level with the predominant response being reduced filter-feeding efficiencies at concentrations of about 100 mg/ℓ. Lethal effects are only seen at much higher concentrations (>7,000 mg/ℓ) and after long (3 weeks) exposures (Clarke & Wilber, 2000).

10.1.3 Fish

As light impacts dramatically on the visual regime in the aquatic environment, turbidity limits the pelagic visual predator but enhances the opportunities for finding refuges for the prey. Even reasonably low suspended sediment concentrations can affect the foraging patterns and feeding success in adult fish (Hecht & van der Lingen, 1992), with avoidance reactions linked to reduced visibility being triggered at concentrations as low as 13,5 mg/ℓ (Wildish et al., 1977). Clark et al., (1998), who investigated the possible effects of mining-related suspended sediment plumes on nearshore fish fauna at Elizabeth Bay, near Lüderitz, concluded that surf zone fish seemed to benefit from the turbidity plume. Fish species richness and abundance in areas affected by the tailings plume were higher relative to control sites on the same beach, suggesting that the plume may provide a form of visual cover from predators. An increase in turbidity effectively reduces the visual range for detecting prey thereby reducing feeding success (Blaber & Blaber, 1980; Bruton, 1985; Cyrus & Blaber 1987a,b,c; Fox et al., 1999). The effect of turbidity on visual predators can have significant impacts on food web structure (Eiane et al., 1999), as well as plankton and fish behaviour (Kaartvedt et al., 1996). Asknes et al. (2004) reported a reduction in zooplanktivorous fish abundance in more light absorbing environments, with the subsequent decrease in predation resulting in an increase in zooplankton abundance. The size structure of the zooplankton community can also be affected, with larger species benefiting more than smaller species from a reduction in predation induced by poor visibility (Asknes et al., 2004). Prolonged periods of elevated turbidity, or reduced irradiance, can promote dramatic shifts in consumer community composition; such as shifts from visual (fish) to tactile (jellyfish) planktivores (Eiane et al., 1999; Sørnes and Aksnes, 2004).

Fish are also susceptible to physical damage from increased suspended sediment loads, particularly at the egg and larval stages. For example, hatching may be delayed for short term exposures to concentrations of 100 mg/ℓ, with mortality of larvae over the longer term at levels of 500 mg/ℓ. In contrast, concentrations as high as 7,000 mg/ℓ had no observable effect on the hatching success of Atlantic herring eggs (Messieh et al., 1981). It should be borne in mind that most of the data for fish eggs and larvae were obtained for freshwater conditions and thus may not be applicable to the marine environment (Clarke & Wilber, 2000).

In adult fish lethal responses are highly species specific with effects occurring at several hundred mg/ℓ for relatively short exposure periods to no effect at concentrations above 10,000 mg/ℓ after a week's exposure. Numerous fish species have been tested for concentration mortality responses (Sherk et al., 1975; O'Connor et al., 1976), and were classified as either tolerant (24 h LC_{10} >10,000 mg/ℓ), sensitive (LC_{10} <10,000 >1,000 mg/ℓ), or highly sensitive (LC_{10} <1,000 mg/ℓ). Bottom dwelling fish species appear most tolerant of elevated suspended sediment. For example, plaice

(*Pleuronectes platessa*) survived immersion in 3,000 mg/ℓ clay suspension for 14 days (Newton, 1973, cited in Moore, 1978).

In summary, short-term impacts may occur by reduced visibility of pelagic food, loss of potential food items (e.g. surf zone macrofauna) due to smothering, reduced egg and larval survival, and the clogging of gill rakers and gill filaments (Vinyard & O'Brien, 1976; Wilber, 1983; Benfield & Minello, 1996). In general, however, the concentrations of suspended sediments that stimulate avoidance responses or result in mortality are orders of magnitude higher than would be expected in naturally turbid coastal waters, or from mine discharges. Being mobile animals, fish are also able to move away from areas of elevated turbidity, and thus less likely to suffer long-term or lethal effects.

The above said, most nearshore and estuarine fish either prefer or are tolerant of turbid waters and only move away when conditions approach tolerance levels. Turbidity induced mass mortalities of estuarine fish *Mugil cephalus* and *Liza richardsonii* through suffocation have been recorded on two occasions from the small Buffels Estuary at Kleinsee when silt-laden floodwaters dammed up behind a mining road prior to breaching (S.J. Lamberth, personal observation). However, these two incidences were of fish trapped in the estuary and not those escaping to the sea. Mortalities of marine fish during the 1987 Orange River floods were mostly due to low salinities and osmotic shock rather than high turbidities (Morant and O'Callaghan, 1990). Higher fish densities than those in surrounding waters were associated with turbidity plumes from marine mining activity to the north of the Orange River Clark et al., (1998). In reality, high flood-induced turbidities in the Orange and other nearshore areas appear to attract many 'turbidity adapted' fish probably in response to potential refuge and/or more concentrated prey. Indeed, aggregations of 'turbidity adapted' fish most notably silver kob *Argyrosomus inodorus*, start occurring in the surf-zone adjacent to the Orange Estuary up to two weeks prior to a flood event, probably in response to the first physico-chemical signals from the catchment. Flow-driven changes in the magnitude and nature of sediment export to the marine environment will result in concomitant shifts in the diversity and abundance of fish that are distributed according to sediment preference or intensity of turbidity plumes off the Orange mouth. Examples of these are flatfish species such as sole and skates which are distributed according to sediment type and particle size or small pelagic fish such as anchovy which find refuge in the turbidity plumes of the Orange and other catchments off the Namibian and South African coastline.

10.1.4 West coast sole

West coast sole *Austroglossus microlepis* are targeted in South African and Namibian waters whereas east coast sole *Austroglossus pectoralis* are caught on South Africa's eastern seaboard. There are two recognized stocks of west coast sole a southern population centred on the Orange mouth and a northern population opposite the Skeleton Coast (Crawford et al., 1987). The trawl fishery for the southern population collapsed in the 1970s. The South African fishery has never recovered whereas there's been a resumption of the fishery in Namibian waters. It is not known whether this represents a recovery of the southern stock or a shift of the northern one southward. South Africa's east coast sole fishery has remained stable.

Fishers in the sole trawl industry here and elsewhere in the world have long used rainfall (terrestrial runoff) as a predictor of catches in the following season. From 1970 to 1980, dam storage capacity on the Orange rose from 10% to 90% of that in the present state. The west coast sole fishery collapsed in the mid 1970s. Demersal trawl survey data (DAFF 1984 – 2011) indicate a weak but positive relationship between Orange flow and biomass estimates. However, there are stronger but negative relationships between sole and their predators e.g. gurnard *Chelidonichthys capensis* and smooth-hound shark *Mustelus mustelus*. Damming saw sediment discharge into the sea change in composition from predominantly silt to cohesive clays. Hypothetically, this influenced the burying ability and crypsis of juvenile sole leaving them more exposed to predators on the sediment surface and abrupt stock collapse. Changes in nutrient and food availability may also have played a role.

10.1.5 *Small pelagic fish*

Anchovy, sardine and round herring are the backbone of the small pelagic purse-seine fishery on the South African and Namibian coast. At 400 – 600,000 tons per annum, catches by this fishery are the largest of all commercial fisheries off South Africa and Namibia, and only the trawl-fishery for hake and other demersal species is more valuable (DAFF, 2012). Small pelagic fish play a key role in regulating ecosystem function arising through their mid-level trophic position and influence on the abundance of both the plankton they feed on and the predators that feed on them (Van der Lingen et al., 2013, DAFF, 2012). High mobility, short life-spans, and feeding at lower trophic levels are life-history characteristics that make small pelagic fish sensitive to environmental influences and extremely variable in their abundance, distribution and recruitment. Hypotheses that may explain the shifts in distribution of anchovy and sardine over the past two decades have included climate change, shorter-term environmental variability, localised fishing pressure, changes in stock structure, natal homing arising as a result of shifts in major spawning locations or a combination of these. Up until this project, there have been no studies on the influence of catchment flows on the distribution of these pelagic fish in the Benguela Current Large Marine Ecosystem (BCLME).

South African populations of anchovy and sardine are monitored by means of hydro-acoustic surveys conducted annually since 1984 (DAFF 2012). Two main assessment surveys are conducted each year, including a summer spawner biomass survey which estimates the total size of the stock and a recruit survey in winter which estimates the number of fish that recruit to the population. In addition to the long-known association with upwelling cells, high densities of anchovy juveniles also appear to be associated with the river plumes of the larger catchments, specifically the Orange, Olifants and Berg on the west coast (Figure 52). This is probably due mainly to the refuge offered by these turbid plumes, productivity fronts being of lesser importance in this upwelling dominated region. Given the generally strong relationship between recruitment and end-of-the-year spawner biomass it could also be expected that river flow, plume size and its influence on juvenile fish density may be another useful predictor of spawner biomass. Initial data-mining of the relationships between river flow and pelagic fish biomass indicated a positive relationship between river-flow and the densities of the three small-pelagic species but only the relationship between anchovy and flow was significant (Figure 53). More robust analysis and numerical modelling of all variables should improve on this.

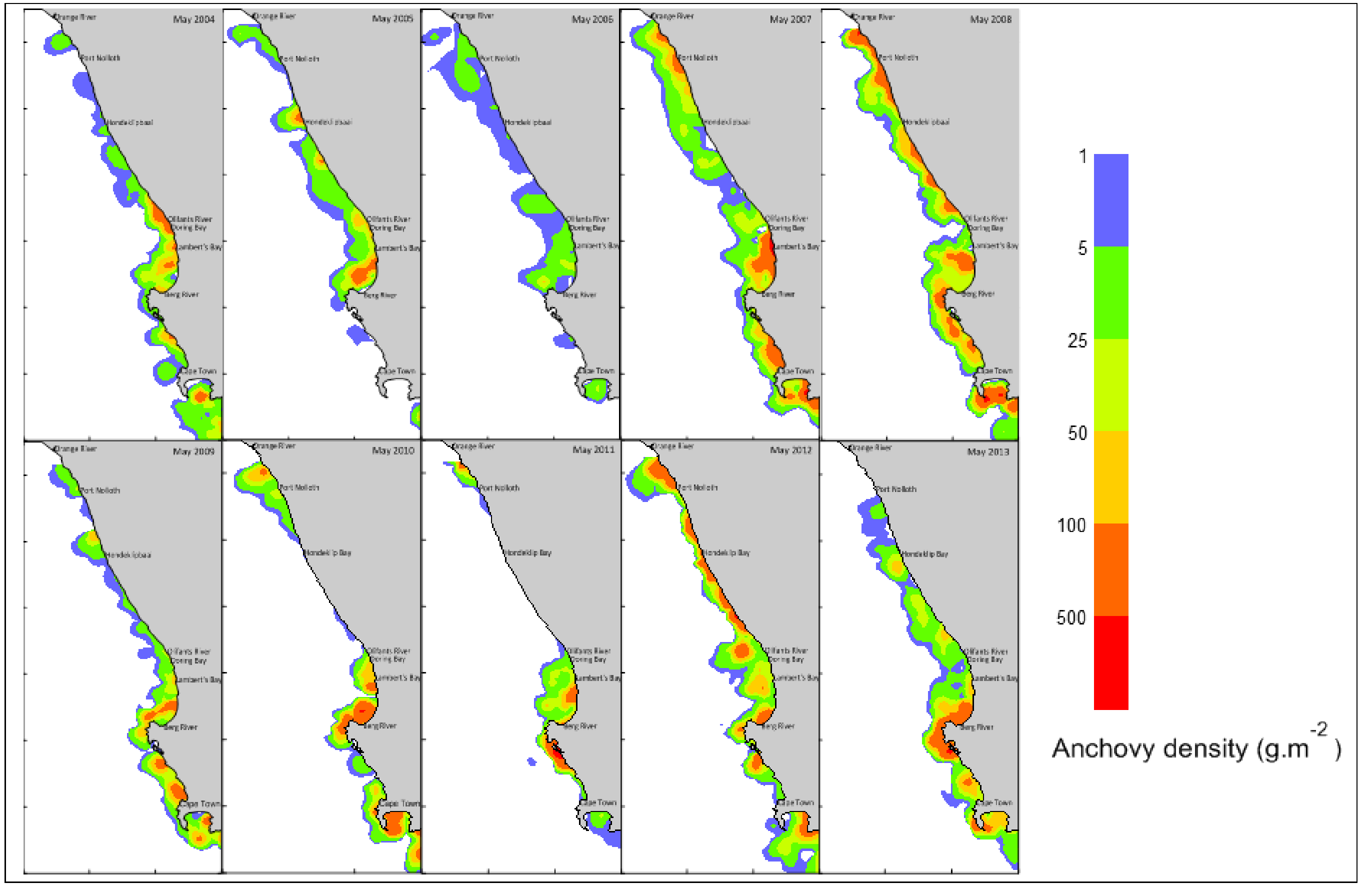


Figure 52. Juvenile anchovy densities g/m² on the west coast of South Africa during May of each year from 2004 to 2013 (Source: D Merkle, DAFF, small pelagic acoustic survey)

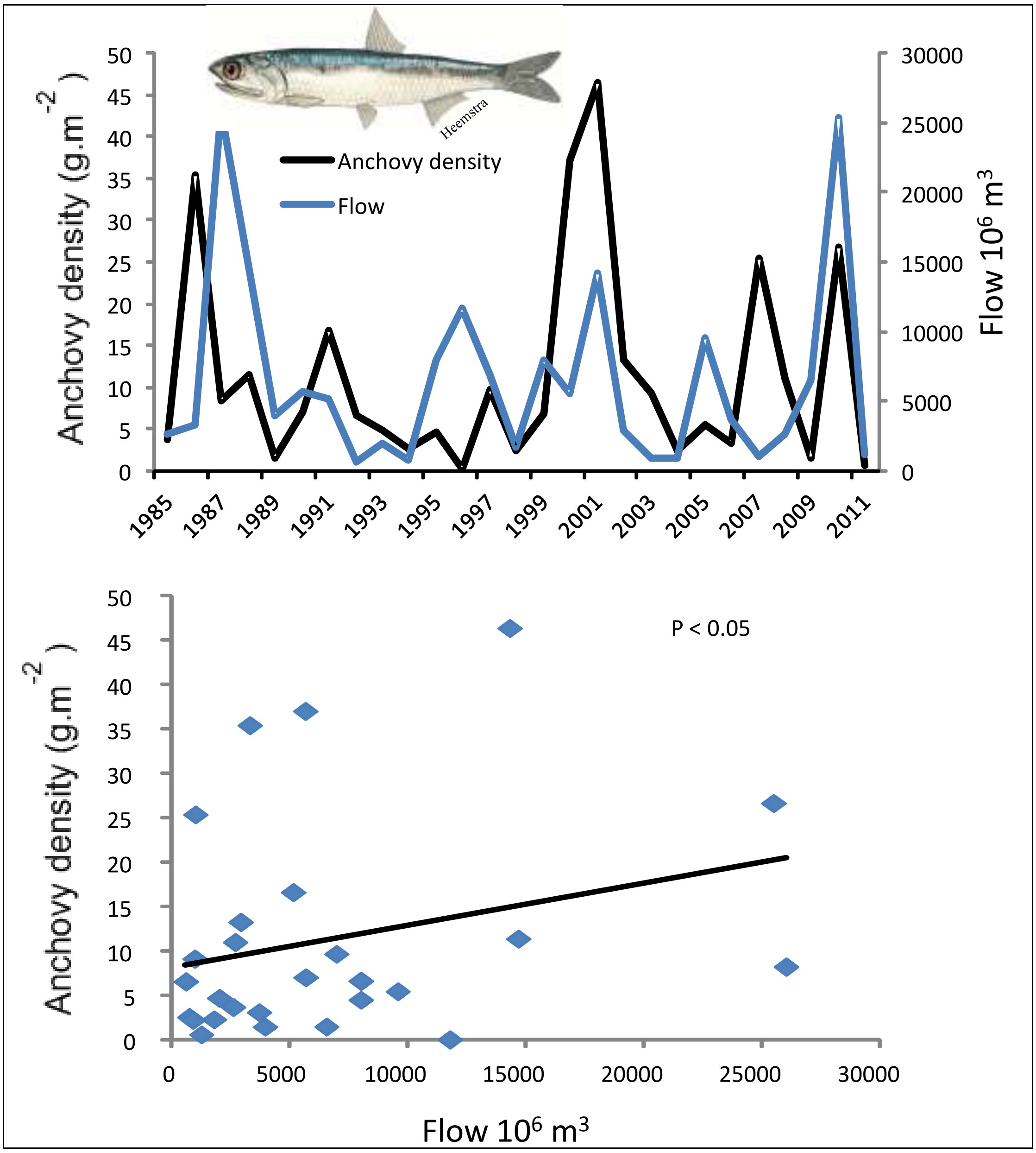


Figure 53. Juvenile anchovy Engraulis encrasicolus density vs. monthly flow volumes from the Orange-Senqu catchment (After DAFF small pelagic acoustic survey)

10.1.6 Other species

Information on the effects of suspended sediment on other benthic invertebrates and crustaceans is comparatively sparse. In the suspension feeding gastropod *Crepidula fornicata*, filtration rate was markedly reduced at suspended sediment concentrations >140 mg/ℓ (Johnson, 1972, cited in Moore, 1978). Genovese & Witman (1999) found that increased particle concentration reduced the growth rate; diversity and abundance of bryozoans due to saturation of the filtering apparatus, potentially leading to competitive disadvantage (see also Cook, 1977).

Sponge species prevalent in temperate coastal waters with high particulate load are characterized by a loose tissue architecture, in contrast with dense-tissue sponges common in cleaner waters. Certain sponges, such as *Tethya* spp. and *Mycale* spp., are adapted to high turbidity and are able to maintain pumping rates during periods of storms, increased water turbulence and increased suspended sediments, or even suspending pumping activity for as long as 5 days (Reiswig, 1971a,b, 1974). Dense tissue sponges, on the other hand, are more susceptible to high particle concentrations and suffer serious decreases in pumping rates (Gerrodette & Flechsig. 1979), which may ultimately have adverse effects on vital functions such as feeding and respiration (Butler et al. 1995). It is believed that deleterious environmental conditions such as heavy sedimentation, low salinities, or organic pollution activate a variety of infections that can lead to mass mortalities of sponges (Hummel et al., 1988; Vincente, 1989; Vacelet et al., 1994).

Ascidians are characterized mainly as non-selective suspension feeders that, instead of producing pseudofaeces possess physiological and behavioural mechanisms such as squirting that allow them to compensate for episodic events of sediment resuspension (Klumpp, 1984; Armsworthy et al., 2001). Squirting serves to alter the retentive capabilities of the mucus sheet, reduce the distance of mucus transport and the probability of clogging, reject unwanted material, and may possibly be involved in particle selection. In the redbait *Pyura stolonifera*, common on the southern African West Coast, concentrations of between 15 mg/ℓ and 50 mg/ℓ of PIM greater than 40 μm triggered a squirting response (Klumpp, 1984).

Crustaceans appear to be comparatively resistant to elevated suspended sediment concentrations. Anderson & Mackas (1986) reported reduced respiration in crustacean zooplankton in mine tailings at 100 mg/ℓ, while higher levels of 560 mg/ℓ affected survival. Concentrations in excess of 10,000 mg/ℓ are generally required for lethal responses over the long term, >240 hour exposures (Clarke & Wilber, 2000).

A wide range of birds forage in or just behind the surf zone. Seabirds are visual predators that forage by sight and therefore need clear water to locate their prey. Most pelagic fish species, which form the major component of seabird diets tend to avoid turbid waters. Suspended sediment plumes may thus affect feeding efficiency of seabirds either by obscuring their vision or by potentially reducing prey availability through avoidance responses of prey species to turbid water areas. However, for pelagic birds that exploit only the top-most layer of the water, turbidity would have to be considerable before affecting the catchability of prey (Moore, 1978). Deeper diving birds such as penguins and cormorants would be more affected by increases in water column turbidity. It is difficult to assess the significance of the potential impacts of flood-related turbidity on seabird populations, as it largely depend on the extent and duration of the sediment plumes. If the plumes are highly localised and disperse quickly, then the consequences are likely to be negligible. If, however, plumes are dispersed over a wider area by winds and currents and persist for more than a few days, this may have a significant local impact on the feeding behaviour and efficiency of seabirds, particularly those species that target specific foraging areas (Simmons, 2005; Braby, 2009).

What must be kept in mind, however, is that marine communities in the Benguela are frequently exposed to naturally elevated suspended-sediment levels. They can thus be expected to have behavioural and physiological mechanisms for coping with this feature of their habitat, and are unlikely to be significantly affected by suspended sediment plumes generated by river discharges.

10.2 Effects of depositing sediments

The larger bedload fraction of the sediments discharged by the river settles out close to the river mouth in a submarine delta, contributing substantially to nearshore sediment transport. The coarser sediments exported from the river thus serve to replenish those sediments continuously eroded from nearshore habitats by wave action and longshore currents.

The effects of sediment deposition on intertidal and shallow subtidal benthic communities in temperate coastal waters are notoriously temporally and spatially variable. Long-term changes occur with the seasons and between years (e.g. in response to episodic flooding of major rivers or berg-wind events). Superimposed on these longer-term trends are high-frequency fluctuations resulting from changes with sea conditions, spring / neap tidal cycles as well as the semi-diurnal tidal cycle. Spatial variability on the other hand results from differences in wave exposure, height on the shore and water depth. Consequently, the intensity of the perturbation to benthic communities caused by suspended sediments is largely determined by the rates of sediment movement and deposition, the quality / quantity of the sedimenting material, and the degree and duration of burial.

In wave-dominated nearshore habitats characterised by highly turbulent near-bottom flows, sediment movement, deposition and resuspension are usually combined. The abrasive action of particles in suspension can cause scouring which may result in the removal of whole organisms or their parts. The force of water movement, for example, increases the danger of dislodgement of benthic invertebrates (Denny, 1985; Denny et al., 1985, Denny, 1987; Denny & Shibata, 1989; amongst others), whilst the abrasive action of suspended particles may result in wearing down or erosion of algal thalli or shells, which subsequently need to be replaced (van Tamelen, 1996; Day et al., 2000). Deposition of sediments under conditions of reduced water movement, on the other hand, can lead to smothering which involves a reduction in light, nutrients and oxygen, clogging of feeding apparatus (Eggleston, 1972), as well as affecting initial recruitment / choice of settlement site (Hiscock, 1983; Rodrýguez et al., 1993), and post-settlement survival (Hunt & Shebling, 1997).

The effects of sediment deposition have been well studied, and are known to have a marked effect in determining the composition and ecology of benthic reef communities (Zoutendyk & Duvenage, 1989). Below follows a review of the effects of sediment deposition on marine macrophytes and invertebrates.

10.2.1 Macrophytes

It has been postulated by many authors that the relative abundance of species of temperate reef algal communities is coupled to fluctuations in sediment levels (Littler et al., 1983; Stewart, 1983; D'Antonio, 1986; Schiel & Foster, 1986; Santos, 1993 amongst others). Together with other environmental variables such as depth (light), substrate topography, wave exposure, consumption by grazers and interspecific competition, sedimentation is believed to influence the distribution and diversity of seaweeds in a variety of ways.

Saiz-Salinas & Isasi Urdangarin (1994) found that the disappearance of algae, and progressive replacement by an invertebrate assemblage of sediment tolerant and opportunistic suspension feeders characterised subtidal community changes along a suspended sediment gradient. Variable patterns of sediment-induced disturbance and recovery, however, may lead either to impoverishment of the existing benthic communities resulting from the extensive dominance of a few species that define the structure of the community (Daly & Mathieson, 1977; Devinny & Volse, 1978; Airoldi et al., 1995; Airoldi & Cinelli, 1997; Airoldi & Virgilio, 1998) or, alternatively, through maintaining habitat heterogeneity (patchiness), sediment disturbance may promote diversity by preventing monopolization of space by competitively dominant species (Littler, 1980; Littler & Littler, 1981; Seapy & Littler, 1982; Littler et al., 1983; McQuaid & Dower, 1990). Consequently, in sediment influenced areas, species richness appears to be controlled by the frequency, nature and scale of disturbance of the system through sedimentation (McQuaid & Dower, 1990; Dethier, 1984).

Persistent sedimentation tends to favour the establishment and subsequent perseverance of intertidal and subtidal algal turfs (Thom & Widdowson, 1978; Kendrick, 1991; Airoldi et al., 1995; Airoldi & Virgilio, 1998) leading to reduced diversity. Certain turf-forming algae demonstrate a remarkable resistance to siltation, and in fact trap and retain sediments, thereby transforming the habitat further (D'Antonio 1986; Stewart 1989; Kendrick 1991; Airoldi & Cinelli 1997; Airoldi & Virgilio, 1998). The thallus-bound sand has the effect of excluding epiphytes as well as many macroscopic invertebrates (gastropods and crustaceans) living in and feeding on algal clumps (Stewart, 1989). Sediment-influenced areas thus appear to serve as refuges for tolerant species from competitors, herbivores and epiphytes resulting in patterns of spatial dominance of sand tolerant and/or opportunistic species (Seapy & Littler, 1982; Littler et al., 1983; D'Antonio, 1986).

Engledow & Bolton (1994) ascertained that the diversity of intertidal seaweeds in Namibia is strongly affected by sedimentation when the level of deposition rises above 5,6 kg of sediment/m^2 (equivalent to a 5,6 mm layer of sand). Sand inundation was found to directly affect species diversity by favouring a few sand-tolerant species, thereby controlling dominance by a single species. Below this critical level of sediment deposition, however, the diversity was determined rather by wave exposure, increasing with increased exposure to wave action, up to a threshold level. Wave action and sand deposition are of course interlinked.

Although many studies have focused on the response of algal assemblages to increased sedimentation and scour, much research has also been conducted on the survival or life style of single species subjected to increased sediment deposition, and/or recurrent sand burial (D'Antonio, 1986; Pineda & Escofet, 1989; Trowbridge, 1996; Maughan, 2001; amongst others).

Crustose coralline algae are commonly found in areas of high wave action and sediment scour (Kendrink, 1991; Konar & Roberts, 1996; van Tamelen, 1996) and appear to require some form of disturbance reducing grazing or overgrowth by other algae and invertebrates, to persist (Johansen, 1981; Littler & Littler, 1984; Steneck ,1986; Kendrink, 1991; Dethier, 1984). However, whereas siltation may negatively affect the initial recruitment and subsequent development of an encrusting community (Maughan, 2001), established encrusting corallines may exhibit prolonged (2 – 3 months) survival when buried by sediment or overgrown by algae (Littler, 1973; Miles & Meslow, 1990; Kendrink, 1991; Trowbridge, 1996). Periodic cover by sand appears to maintain the coralline crusts by providing a cleaning effect, allowing the species to occupy much of the space in an area. Although erect corallines are less resistant to scouring (van Tamelen, 1996), basal crusts can survive for over a year under sand regenerating rapidly after sand removal (Stewart, 1989).

The abundance of many erect fleshy red algae declines when scouring becomes more intense (e.g. during winter storms) suggesting that the lower distributional limits on the shore may be set by the effects of sediment movement (van Tamelen, 1996). A number of southern African west coast species are, however, adapted to sand-influenced rocky intertidal habitats, and may dominate the midlittoral region. *Iridea capensis* and several species of *Ahnfeltiopsis* (previously *Gymnogongrus*) have crustose holdfasts and tetrasporophytes, and are able to survive sand burial for at least 6 months (Stegenga et al., 1997; but see also Dahl, 1969, 1971; D'Antonio, 1986; Trowbridge, 1996).

Even small amounts of sediment in combination with water movement can produced a scouring effect which prevents the recruitment and establishment of kelps and may raise species diversity (Foster, 1975; Kendrick 1991). Off the southern African West Coast, the kelps *L. pallida* and *E. maxima* can tolerate a degree of sand scour and even short-term deep sand burial (Marszalek, 1981; Rogers, 1990; Airoldi et al., 1996), but persistent sand inundation destabilizes the holdfast ultimately leading to detachment from the rocky substrate below (R. Anderson, Seaweed Unit: Marine & Coastal Management, pers. comm., see also Ebling et al., 1960; Umar et al., 1998) or thalli may be broken by abrasion or die (Daly & Mathieson, 1977). The consequence is a patchy loss of kelp plants by dislodgement of holdfasts. However, kelp recovers relatively quickly after each disturbance episode resulting in the competitive dominance of kelp over other species. Long-term changes in kelp forest communities in response to various disturbances was documented by Dayton et al. (1992), who determined that disturbances caused many lag-effects including the outbreak of understory algae (see also Foster, 1975), intraspecific competition, and changes in grazing patterns of herbivores. On the West Coast, where adult plants are removed, a highly diverse understorey algal community establishes and may predominate for at least 12 months before kelp sporelings can recruit (Simons & Jarman, 1981; Christie et al., 1998; Levitt et al., 2002). Increased interspecific competition resulting from this increased species diversity is, however, thought to negatively effect kelp recruitment (Levitt et al., 2002). Although kelp sporelings are tolerant of low light levels associated with elevated turbidity, kelp recruitment is thought to be affected by sediment scour and siltation, the former preventing spore settlement and the latter smothering established germlings (Devinny & Volse, 1978, but see also Norton, 1978).

In conclusion, it appears that prolonged sediment smothering may have more extensive impact than short-term deep sand burial. However, only in extreme conditions of sand movement may macrophytes be broken by abrasion or killed by prolonged burial. Nonetheless, sediment movement and deposition play important roles in maintaining habitat heterogeneity and thus determining the diversity of algal assemblages.

10.2.2 Invertebrates

The amount of suspended sediment in water is a major physical factor limiting the distribution and abundance of benthic invertebrates. Shifting sediments and frequent sand inundation have a significant effect on the species diversity and community structure of intertidal and subtidal macrobenthic assemblages by removing grazers (Littler et al., 1983; D'Antonio, 1986; Branch et al., 1990; Marshall & McQuaid, 1993), predators (Robles, 1982), and either promoting or inhibiting the establishment of competitively dominant species (Berry, 1982; Taylor & Littler, 1982).

Rocky-shore communities disturbed by sediments, or pollutants and other anthropogenic impacts are often typified by a dominance of foliose algae, as a result of reductions in grazer densities (Littler & Murray, 1975, 1978; Hawkins & Hartnoll, 1983; Littler et al., 1983; Hockey & Bosman, 1986; López Gappa et al., 1990; Eekhout et al., 1992). Patellid limpets are dominant grazers on rocky intertidal shores on the southern African West Coast but are intolerant of siltation and sand inundation, and their deletion in areas of inundation leads to a proliferation of algae (Marshall & McQuaid, 1993; Pulfrich et al. 2003a,b).

Sand inundation may, however, promote siphonariid limpets, because they are not only tolerant of sand but inferior competitors with patellids, having developed the capacity to withstand hypoxia and evolved reproductive strategies more suitable for a sand-influenced habitat (Marshall & McQuaid, 1989). Siphonariids are, however, incapable of controlling macroalgae in the manner that patellid limpets do.

Varying tolerances to sedimentation have also been reported for numerous other macrobenthic species inhabiting intertidal and immediate subtidal habitats (Chitons: Dayton, 1975; Dethier, 1982; Littler et al., 1983; Paine, 1984; D'Antonio, 1986; Bryozoans: Maturo, 1959; Cook, 1963; Eggleston, 1972; Cancino & Hughes, 1987; Keough, 1986; Hughes, 1989; Genovese & Witman, 1999; anemones: Taylor & Littler, 1982; Littler et al. 1983). For example, the removal of clonal assemblages of the anemone *Anthopleura* sp. resulted in the establishment and competitive dominance of a sand tube building polychaete *Phragmatopoma californica*, or of more sand tolerant algal species. *Anthopleura* frequently characterise shorelines influenced by deposited sediments and may alter the physical environment of the lower intertidal zone by contracting into a moisture retaining carpet of small cushions covered in shell and rock fragments and sand. These moist carpets facilitate recruitment and survival of invertebrate species living between the anemones, and prevent the establishment of opportunistic macroalgae species that competitively utilize space (Taylor & Littler, 1982)

An important invertebrate predator in algal bed habitats is the rock lobster. Although crustaceans are comparatively resistant to elevated suspended sediment concentrations, chronic siltation and/ or sand inundation may affect adult rock lobsters, as well as the post-larval settlement stage, through habitat degradation or loss. Herrnkind et al. (1988), showed that high siltation, both natural and man-induced, may be potentially deleterious to rock lobster recruitment, as settlement of post-larval pueruli of the spiny lobster *Palinurus argus* was significantly reduced in severely silted habitats, despite extensive benthic algal growth. Furthermore, as prey abundance was lower in silted algae, recruits subsequently left the affected habitat to obtain adequate food in unsilted algal stands. Movement by the juveniles, searching either for food or unsilted habitat, was predicted to result in increased mortality, thereby affecting recruitment success. On the southern African West Coast, nearshore reefs, islands and kelp beds play an important role as habitats for the West Coast rock lobster *Jasus lalandii*, as well as being significant for post-puerulus settlement and juvenile recruitment (Tomalin, 1996). Sedimentation or heavy siltation of nearshore reefs may reduce the carrying capacity of an otherwise suitable habitat, therefore potentially directly affecting rock-lobster populations, or reducing regional recruitment where sedimentation is widespread. This may consequently have important implications for the success of the commercial harvest of this resource in an area.

On the southern African West Coast, the loss of kelp in areas of high concentrations of suspended sediments can also lead to competitive dominance by the Cape Reef worm *Gunnarea capensis* in intertidal and shallow subtidal regions. This filter-feeding polychaete is capable of forming extensive communal reefs and often dominates areas influenced by mobile sediments thereby precluding the settlement and establishment of kelp sporelings (see also Wilson, 1971; Taylor & Littler, 1982).

In unconsolidated habitats, many benthic invertebrates are able to burrow or move through the sediment matrix, and some infaunal species are able to actively migrate vertically through overlying deposited sediment thereby significantly affecting the recolonisation and subsequent recovery of impacted areas (Maurer et al., 1979, 1981a,b, 1982, 1986; Lynch, 1994; Ellis, 2000; Schratzberger et al., 2000; but see Harvey et al., 1998; Blanchard & Feder, 2003). Migration rates and survival were generally higher in sandy sediments than in silty sediments, although the available food sources in the depositional sediment was also influential (Schratzberger et al., 2000). The frequency of deposition and amount of deposited material, however, also play an important role in the response of benthic invertebrate assemblages, with most intense changes in communities occurring after the deposition of a single large dose (as would be the case during a major flood event).

Vertical migration experiments with beach macrofauna, to determine their tolerance to sand overburdens, found that several species were capable of burrowing through sediments between 30 and 90 cm (Maurer et al., 1979; Lynch, 1994). In contrast, consistent reduction in benthic macrofaunal densities, biomass, and species richness were noted during deposition of mine tailings and dredge spoils when the thickness of deposited sediments exceeded 15 – 20 cm (Schaffner, 1993; Burd, 2002). Similarly, Roberts et al. (1998) and Smith & Rule (2001) found difference in species composition detectable only if the layer of instantaneous applied overburden exceeded 15 cm. In general therefore, mortality tends to increase with increasing depth of deposited sediments, and with speed and frequency of burial.

10.3 Effects of depressed salinities

Periodic floods issuing from the Orange River will extend further from the mouth than regular river discharges. Such episodic events will affect coastal systems by severely depressing salinities due to elevated freshwater input into the marine system, and cause large-scale and substantial increases in turbidity by increasing the sediment load. Depending on the temperature of the flood water, nearshore water temperatures may also be depressed or elevated. Furthermore, the deposition of silt and organic matter resulted in depletion of oxygen level in the water column (Branch et al., 1990). These physical affects can act alone or in synergy to have an impact on the marine biota. Whilst they can result in mass mortalities of the associated marine biota, the degree of impact depends largely on the severity and duration of the flood, as well as the physiological tolerances of the organisms in question.

In March 1988, the Orange River experienced its most severe flood of the 20th century, discharging some 15,400 Mm^3 of water, and 35,7 Mm^3 (64,2 M tonnes) of bedload and suspended sediment (Branch et al., 1990; Bremner et al., 1990). The most intense effects of the flood on intertidal and shallow subtidal biota occurred within 25 km of the mouth, but consequences were evident up to 140 km south of the Orange River.

Species resident above the midlittoral were relatively unaffected by the flood, probably as the result of higher physiological tolerances of the flora and fauna inhabiting this zone. In the sublittoral below about 5 m depth, the biota was also relatively unaffected as the floodwater rode above the saltwater, establishing a sharp halocline. From 5 m depth to the midlittoral the devastation of benthic organisms was almost 100%, with the rocks virtually denuded of all life. The devastation of the flood was thought to be principally attributable to depression of salinity (Branch et al., 1990), with lower than usual salinities being measured as far south as Hondeklip Bay, some 200 km from the mouth. The effects of the flood waters led to a near-absence of patellid limpets, producing a proliferation of opportunistic algae that dominated the substrate on low- to mid-shores, with a concurrent decline of encrusting algae. Subsequent to the flood, a succession of benthic communities established themselves in the intertidal, taking approximately 5 years for a full recovery (G.M. Branch, UCT, pers. comm.). Sublittoral fish are unlikely to be affected by flooding, finding refuge below the halocline, but intertidal fish trapped in rock pools would likely be severely affected. Marine mammals would be negligibly affected.

Reports of a lobster walkout at U50 in Mining Area 1 during 1974 as a result of a plug of sediment laden water moving northwards from the river mouth (CSIR, 1985), may likewise be attributable to the penetration of flood water into the shallow subtidal areas following the major Orange River flood of 1974.

10.4 Implication of flow modifications on nursery function

Historically the Orange Estuary was a temporarily open/closed system closing briefly under low flow periods during winter months, or the build-up of a sandbar across the mouth through storm conditions. However, the Orange is an extremely regulated river, with over fifteen major impoundments and numerous inter-basin transfer schemes. Hydro-power releases during the winter months have elevated flows to such an extent that mouth closure seldom, if ever, occurs and seawater intrusion is severely limited (Taljaard, 2005). The last documented closure events occurred in spring 1993, December 1994 and December 1995. Very few data are available for these events.

Furthermore, summer flood events that naturally reset the system and inundated saltmarsh and floodplain, have been greatly reduced due to increased water abstraction in combination with high evaporation. Most of the spring and early summer floods are now captured by the upstream dams with the result that mouth closure (previously a winter event) is now more likely to occur during summer. Historical changes in the discharge volumes, shifts in seasonal flow variations and shifts in mouth closure events will result in a seasonal reversal of the abiotic drivers with potential serious consequences for the estuary and ultimately the marine environment beyond (Taljaard, 2005). Such changes would almost certainly have influenced the community composition and abundance of fish and invertebrate communities surrounding the mouth of the estuary, and will have some impact on those species that rely on seasonal cues for entering or exiting the estuary.

10.5 Linking the key ecosystem services to abiotic drivers

Alterations to inflow of freshwater to the marine environments thus potentially have a number of effects on organisms and habitat-forming species in coastal and near-shore marine habitats. These effects are primarily due to alterations in flow volume, sediment loads, and hydrodynamics, as well as changes to water quality (i.e. salinity, temperature, and nutrients). Changes in response to increase flows will include reduction/elimination of less tolerant species due to the abiotic effects on recruitment, growth and survival. This in turn would likely result in changes in community structure to favour sediment-tolerant communities. Concurrent declines in biodiversity would also be expected. In the case of habitat-forming species such as kelps, response to increased flows will likely include reduction in cover through elimination of the dominant species with potential competitive dominance by a sediment tolerant species such as Cape Reef worm. Organisms dependent on the kelp-bed habitat (e.g. rock lobsters) would also be affected by changes to their preferred habitat, recruitment, growth, movement, mortality and fecundity.

In the case of soft-sediment macrobenthos, community composition and diversity is likely to be affected due to chronic changes to sediment structure, particularly in the ‘fallout’ plume of the river discharge, but also further downstream as deposited sediments become redistributed over the long-term. The change in texture of the suspended sediment load carried by the river from silt-dominated material discharged pre-1970, to clay dominated since this time (Bremner et al., 1990), is likely to have affected the structure of benthic macrofaunal communities on the delta front. It is possible that the habitat complexity on the delta front has been reduced over time due to dams constructed on the river trapping particular size fractions. This reduction in habitat complexity on an interstitial scale may have negatively affected species diversity. The absence of historic data on community structure in the area, however, makes it impossible to quantify the direction of this change.

Changes in response to decreased flow rate, and thus decreased stressors, are likely to result in the development of more complex community structures with higher biodiversity. Reduced sediment deposition and redistribution on rocky intertidal shores and submerged reefs would facilitate increased habitat complexity enabling the establishment of more diverse communities, which are likely to show greater resemblance to similar habitats not typically influenced by inflows.

The effect of reduced flows on soft-sediment macrobenthos is difficult to predict. Whereas a lower depositional environment may favour a more diverse community, this may be countered by a loss of habitat complexity, which is expected to result in reduced biodiversity.

The important ecosystem services provided by the nearshore marine environment in the vicinity of the Orange Estuary and the critical abiotic drivers that they respond to are summarised in Table 13. This table considers increased inflow during pulsed events or floods only.

Table 13. Critical abiotic drivers and the responses to these of important ecosystem services provided by the nearshore marine environment in the vicinity of the Orange-Senqu River mouth. Only increased inflows are considered.

Biotic component / ***Critical drivers***	***Comment***
Phytoplankton	
Suspended sediments	Reduced primary productivity with increasing turbidity.
Sediment deposition	No effect.
Salinity	Biomass is generally elevated near riverine plumes. Salinity gradients influence the structure of phytoplankton assemblages, with salinity levels determining which taxa dominate the community.
Macrophytes	
Suspended sediments	Reduced primary productivity with increasing turbidity.
Sediment deposition	Change in community structure to favour sediment adapted species.
Salinity	Typically adapted to a wide range of salinities. Degree of inflow will determine depth of adverse effects on individual species and assemblages.
Habitat-forming macrophytes (kelps)	
Suspended sediments	Kelp grows at shallower depths and beds become dominated by *Laminaria* which is adapted to lower light levels associated with increased turbidity. Reduction in kelp bed area.
Sediment deposition	Kelp is dependent on availability of hard substratum and susceptible to sediment scouring. Scouring and siltation prevents the recruitment and establishment of kelps and may raise species diversity of other macrophytes. Loss of kelp bed habitat may affect recruitment of species such as rock lobster.
Salinity	Sensitive to low salinities with lethal effects dependent on the volume of inflow and the depth of the low salinity wedge.
Soft-sediment macrofauna	
Suspended sediments	Reduced filter-feeding efficiencies of bivalve molluscs at ~100 mg/ℓ. Lethal effects only after long exposure to >7,000 mg/ℓ.
Sediment deposition	Abundance, biomass and diversity dependent on sediment properties. Most

Biotic component	***Critical drivers***	***Comment***
		species are adapted to some degree of smothering by moving through depositing sediments. Migration rates and survival generally higher in sandy sediments than in silty sediments whereas burrowing and crypsis easier in the later.
	Salinity	Mortalities of benthic invertebrates in the top 2 – 3 cm of sediment following strong pulses of freshwater inflow.
Reef-associated macrofauna		
	Suspended sediments	Change in community structure to favour suspension- and deposit-feeding species.
	Sediment deposition	Change in community structure to favour sediment tolerant species (e.g. Cape Reef worm). Scouring of shells/body parts resulting in loss of diversity. Lethal effects on prolonged smothering.
	Salinity	Heavy mortalities of benthic invertebrates following strong pulses of freshwater inflow.
Rock Lobster		
	Suspended sediments	Lethal responses only after long exposure to >10,000 mg/ℓ.
	Sediment deposition	Dependent on the availability of reef habitats for shelter and food and populations may decline during periods that reefs are covered by depositing sediments.
	Salinity	Avoid areas of depressed salinity by migrating away. Lethal response to rapid declines in salinity.
Fish		
	Suspended sediments	Delayed hatching at 100 mg/ℓ) and increased mortality at 500 mg/ℓ. Lethal effects to adults typically only after long exposure to >10,000 mg/ℓ. However, most fish associated with the plume are adapted to, or have a preference for elevated turbidity. One advantage would be the refuge provided by turbid waters.
	Sediment deposition	Benthic species distributed according to sediment particle size, directly due to burrowing ability and crypsis and indirectly via prey preference for s particular sediment type. Mobile and can move away. Indirect effects through loss of habitat or food source for reef-associated or demersal species.
	Salinity	Avoid/ attracted to areas of depressed salinity by migrating away/towards.

Using the significance ratings for predicted changes relative to reference conditions in Table 8 (section 9.3), the responses of the key ecosystem services/biotic component to predicted abiotic changes under the various scenarios in flow were assessed. The results are provided in Table 14.

Table 14. Assessment of the responses of the key ecosystem services/biotic component to predicted abiotic changes under the various flow scenarios

Key ecosystem services/ biotic component	**Natural**	**Present**	**Sc 2**	**Sc 3**	**Sc 4**	**Sc 5**	**Sc 6**	**Sc 7**
Phytoplankton	-1	+1	+1	+1	+1	+1	+2	+2
Macrophytes	-2	+1	+1	+1	0	+1	+3	+3
Habitat-forming macrophytes (kelps)	-2	+1	+1	+1	0	+1	+3	+3
Soft-sediment macrofauna	+3	-1	-1	-1	-1	-1	-2	-2
Reef-associated macrofauna	-3	+1	+1	+1	+1	+1	+3	+3
Rock Lobster	-3	+1	+1	+1	+1	+1	+3	+3
Benthic biodiversity	+3	-1	-1	-1	-2	-1	-3	-3
Nomadic coastal fish (e.g. kob)	+3	-2	-2	-2	-1	-2	-3	-3
Demersal soft sediment fish (e.g. sole)	+3	-3	-3	-3	-3	-3	-3	-3
Small-pelagic fish (e.g. anchovy)	+3	-2	-2	-2	-1	-2	-3	-3
Intertidal, subtidal, surf-zone fish (e.g. clinids, galjoen)	+3	-1	-1	-1	0	-1	-1	-1

Using the results of the model simulations in combination with a review on the responses of marine flora and fauna to the abiotic drivers expected from a river discharge, it has been possible to provide a comparison of the expected changes in selected marine biotic components to various proposed flow scenarios. From referenced condition to present state, it is not possible to discern changes in biotic components in response to Sc 2, 3 and 5. For Sc 5 slight changes may be expected, although these may be negated by the predicted decline in the number of occurrences of flood and freshwater pulses. However the results for development Sc 6 and 7 are significantly different to those for present state and Sc 2 to 5.

The changes in the biotic components for Sc 6 and 7 in general amount to an anticipated increase in marine benthic biodiversity in the vicinity of the river mouth, as the abiotic stressors decline or are removed entirely should the mouth remain closed for extended periods of time. Marine communities are expected to become more similar to those in West Coast habitats not influenced by river inflows.

11 Socio economic implications

In recent years, concerns about the potential impact of marine mining operations along the southern Namibian coastline have increasingly focussed on the potential smothering or scouring of nearshore reefs by sediments discharged or re-distributed by mining operations. However, the primary source of sediment input into the central Benguela region is, and always has been, the Orange River. The long-term historic average annual sediment input from the River (60 million tons) is four times the average annual mining-related sediment input.

Penney et al. (2008) investigated the relationship between rock lobster declines and major Orange flood events off Namibia (Figure 54) and South Africa (Figure 55). Declines off Namibia coincided with, or followed shortly after, flood events in 1955, 1958, 1967, 1974, 1976 and 1988. In contrast, the massive decline that coincided with the abolition of the Namibian minimum size limit between 1968 and 1970 does not coincide with an flood event, emphasizing the negative impact that this management decision had on the Namibian rock-lobster fishery. Off South Africa, declines coincided with the 1938, 1944, 1955, 1958, 1967 and 1988 flood events. Even the 1948 flood appears to coincide with a minor dip in catches during the period of otherwise rapid expansion of this fishery between 1945 and 1951. Only the 1976 flood did not seem to coincide with a catch decline in the South African fishery.

The recent collapse of both these fisheries, attributed by various authors to 'unexplained' environmental factors (Gammelsrod et al., 1998; Hampton, 2003), followed directly after the 1988 Orange River flood. Total sediment discharges over the three month period from February – April 1988 are estimated to have been almost five times the current average annual sediment input from the Orange River. The massive quantities of sediment deposited on the submarine delta would have been gradually re-distributed north and south of the river mouth by waves and currents for many years after the event. The resulting benthic community degradation is likely to have been an important cause of subsequent declines in rock-lobster growth observed over much of the area.

A number of factors other than relentless over-fishing of accumulated stocks with low productivity appear to have contributed to the decline of Namibian and South African rock lobster resources over the past 50 years. Recovery of the over-fished resources has been severely limited by highly variable recruitment which, to a large extent, results from the extreme environmental variability, and generally harsh conditions, of the central Benguela region. Coupled with declines in growth rate since the late 1980s, productivity was further reducing. Furthermore, since 1988, large-scale environmental changes have contributed to increased frequency and severity of low oxygen events, and occasional massive floods have deposited substantial quantities of mobile sediment into the nearshore ecosystems.

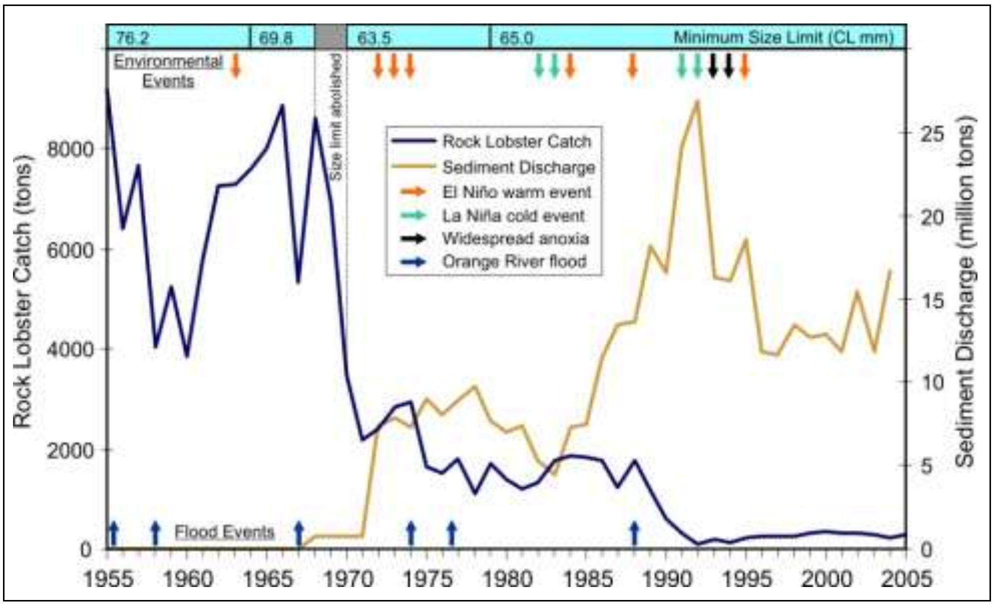


Figure 54. Total annual reported Namibian rock lobster catches per year (all areas, tons), compared with total estimated mining-related sediment discharges (millions of tons) along the Namibian coast south of Lüderitz, from 1955 to 2005 (from Smith et al., 2006). The top bar shows periods during which various minimum size limits (carapace length - mm) applied. Arrows show years of major warm events (Benguela Niños), cold events, low-oxygen water events and major Orange River floods.

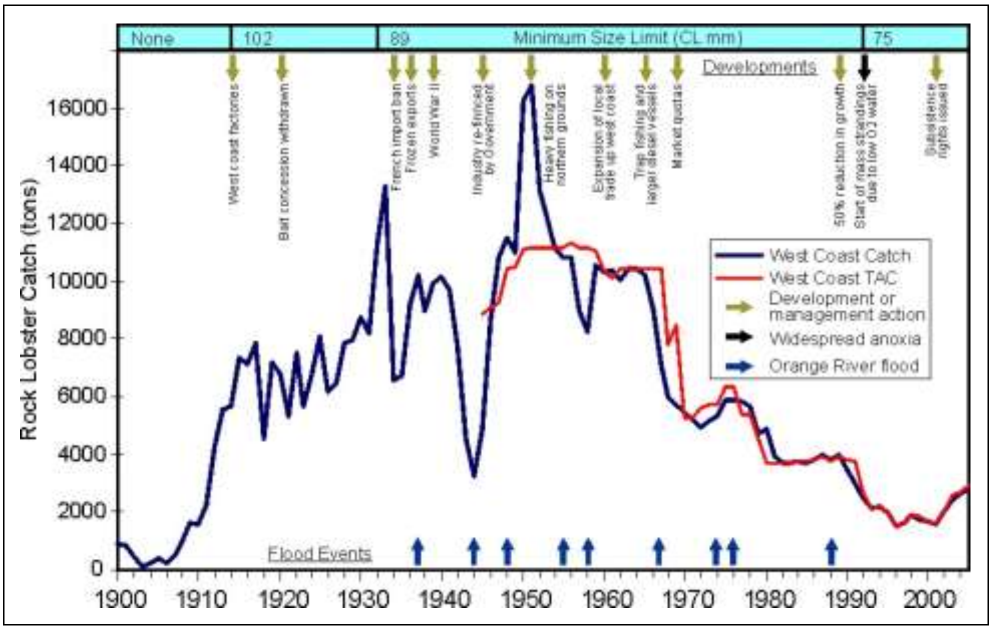


Figure 55. Total annual reported South African rock lobster catches per year (tons), compared with annual catch limits (TACs) along the South Africa west coast from 1900 to 2005 (data from Melville-Smith & van Sittert, 2005). The top bar shows periods when various minimum size limits (carapace length – mm) applied. Arrows show years of major fishery developments or management interventions and years of major Orange River floods.

11.1 Macroalgae

The South African West Coast is divided into numerous seaweed concession areas (see Figure 34). Access to a seaweed concession is granted by means of a permit from the Fisheries Branch of the DAFF to a single party for a period of five years. The seaweed industry was initially based on sun dried beach-cast seaweed, with harvesting of fresh seaweed occurring in small quantities only (Anderson et al., 1989). The actual level of beach-cast kelp collection varies substantially through the year, being dependent on storm action to loosen kelp from subtidal reefs. Permit holders collect beach casts of the both *Ecklonia maxima* and *Laminaria pallida* from the driftline of beaches. The kelp is initially dried just above the high water mark before being transported to drying beds in the foreland dune area. The dried product is ground before being exported for production of alginic acid (alginate). In the areas around abalone hatcheries fresh beach-cast kelp is also collected as food for cultured abalone, although quantities have not been reported to DAFF. Activity in the northern Namaqualand concessions in the vicinity of the river mouth is limited, mainly due to restricted access to the coastal diamond mining areas.

Further south, around Cape Columbine, permits also allow the harvesting of live kelp by hand from a boat. The Maximum Sustainable Yield (MSY) for the hand harvested product is set annually (Anderson et al., 2003) and is based on the estimated kelp biomass in the concession area determined from the total area of kelp beds and the mean biomass within them (Table 15).

Table 15. Maximum Sustainable Yield (MSY) (in kg wet weight) and beach-cast collections (in kg dry weight) for kelp concessions north of Lamberts Bay (Data source: Seaweed Section, DAFF)

Concession number	Concession holder	MSY	2005	2006	2007	2008	2009
13	Eckloweed Industries	60,000	65,898	94,914	122,095	61,949	102,925
14	Eckloweed Industries	216,000	165,179	145,670	79,771	204,365	117,136
15	Rekaofela Kelp	1,020,000	10,300	19,550	0	23,646	0
16	Rekaofela Kelp	220,000	35,920	28,600	84,445	16,804	0
18	FAMDA	835,000	0	0	0	0	0
19	Premier Fishing	220,000	0	0	0	0	0

11.2 Fish

The longshore littoral transport of large volumes of sands deposited during flood events could potentially have a negative impact on the standing stock biomass of nearshore kelpbeds. Sand scour and persistent inundation could lead to large-scale detachment from the rocky substrate below. Although this may initially be beneficial to kelp collectors through increased yields of beach cast kelp, it may result in the establishment of competitively dominant species such as Cape Reef worm (*Gunnarea capensis*) or sandy anemones (*Aulactinia reynaudi*) with concomitant cascade effects of higher order consumers who rely on kelpbeds as nursery areas or recruitment habitats (e.g. *Jasus lalandii*).

12 Recommended freshwater requirements

12.1 Environmental flow requirement

Based on model simulations (hydrology, sediment and hydrodymanics) and associated indices of predicted change, it is clear that the change from the reference condition to present state has been highly significant. The present state is substantially poorer than those prevailing under the reference conditions. Based on estimated river flows, the major changes occurred during the early 1970's when some major dam development occurred in the Orange River catchment.

In terms of the proposed new scenarios, the indices used to predict likely change suggest that there is no discernable difference between the present state and Sc 2 to 5. However the indices do indicate significant detrimental changes associated with development Sc 6 and 7. It may be the case that one or more of the proposed Sc 2 to 5 could be significantly different to the present state but that the precision of the indices is too low to discern such changes.

12.2 Monitoring requirements

The following additional surveys are needed to improve the baseline information on the Orange nearshore environment (summarised in the Table 16).

Table 16. Additional baseline surveys needed to improve baseline information

Component	***Baseline survey***	***Temporal scale***
Sediments	Sample suspended sediment load at Vioolsdrift.	Daily
Remote sensing	Observations on turbidity, salinity, temperature and chlorophyll-a	Daily
Fish	Small pelagic acoustic surveys in South African and Namibian coastal	2X annual (i.e. quarterly)
Invertebrates	Benthic and beach monitoring on both Namibian and South Africa side.	Annual (i.e. quarterly)

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Appendix A Modification of freshwater inflows and associated fluxes from the Orange River into the adjacent marine environment: Model predictions of indices of potential change

A.1 Overall approach to the study

The approach taken in the modelling study is to identify a number of representative inflow/'flood' scenarios and associated flow ranges for simulation in the model. Flood and high flow events representative of each of the identified flow ranges have been identified in existing measured flow data which are used as representative flood scenarios in the model. The resulting model outputs for each type of 'high flow/flood' event have been analysed with respect to a number of both abiotic and biotic metrics (e.g. extent of sediment deposition on the seabed and changes in salinities in the water column) that have been designed to provide a quantitative measure of the expected changes ecologically important 'habitat variables' in the relevant offshore habitats.

These quantitative measures (metrics) for each of the 'high flow/flood' events simulated are then used together with a characteristic distribution of the occurrence of these 'high flow/flood' events under the various proposed scenarios, to provide indices of likely overall change in offshore habitats under each of the various proposed scenarios.

These indices initially are provided as percentages relative to either the reference condition with respect to river inflows. The final step has been to translate these percentage changes in offshore habitats (under the various developments scenarios) into significance ratings with respect to the likely severity of the resultant habitat changes in the offshore marine environment.

A.2 Approach to the modelling component of the study

The modelling undertaken in this study follows on a previous modelling study undertaken by Smith et al. (2006) to assess the cumulative effects of sediment discharges from on-shore and near-shore diamond mining activities on the Benguela Large Marine Ecosystem (BCLME). In that study two large flood events, namely the 1988 and 1996 floods, were simulated to provide an insight into the natural variability in the BCLME associated with such high flows from the Orange River into the marine environment. In that study a coupled modelling study was undertaken using the Delft3D-Wave, Delft3D-Flow and Delft3D-WAQ software modules (Deltares, 2011a,b,c), however in this study only a coupled Delft3D-Wave and Delft3D-Flow modelling study has been undertaken. Previously the sediment dynamics were simulated in the Delft3D-WAQ software however here the sediment dynamics are simulated within the Delft3D-Flow module that now includes a sophisticated sediment dynamics modelling capability that was not fully developed at the time of the 2006 BCLME study.

Originally the intention was to simulate each of the selected 'high flow/flood' events using the offshore environmental conditions (winds, waves and currents) prevailing at the time of the specific 'high flow/flood' being simulated. It was realised that, due to the limited availability of such offshore data prior to 1996, such an approach would severely limit the time period from which 'representative' floods (in the various representative flow ranges) could be selected for this study. More importantly it was realised that it would not be possible to isolate and quantitatively describe the changes in the offshore environment for the various flow ranges to be simulated were both the offshore environmental conditions and the flows from the Orange River to be changed simultaneously. Therefore to isolate the effects of only the changes in river inflows to the marine environment, it was decided to run all of the selected 'high flow/flood' conditions using only one fixed set of offshore environmental conditions, namely those that had been selected in the BCLME study (Smith et al., 2006) to simulate the 1988 flood. It should however be noted that the offshore environmental conditions used in the 1988 flood simulations in the BCLME study were not those prevailing in 1988 but rather a set of offshore wave conditions for a similar period in 1998. The reason for this is that information on the offshore environmental conditions that prevailed during the 1988 floods is largely non-existent.

All model simulations have been set-up for a 3 month period using the offshore environmental conditions utilised to simulate the 1988 floods in the BCLME study (Smith et al., 2006)., however for expedience each model simulation has been undertaken for only a limited 64 day period (representative of the period of 27 February 1988 to 30 April 1988). In all cases (except for perhaps the 1988 flood) such a two month period was sufficient to capture the inflow of flood waters to the marine environment during the identified 'high flow/flood' event, as well as the initial deposition of the river sediments in the marine environment and varying degrees of re-suspension and re-distribution of these sediments. The smaller the 'high flow/flood' event simulated, the shorter is the typical duration of the inflow and consequently the greater is the relative time period during which the initially deposited sediments can be re-suspended and re-distributed in the marine environment during the model simulation.

A.3 Model set-up, calibration and validation

The model used in this study comprises a coupled wave and flow model. For the wave modelling Delft3D-WAVE model is used that has the SWAN wave modelling system at its core. The flow and sediment modelling has been undertaken using Delft3D-FLOW that includes a sophisticated sediment dynamics modelling capability.

A.3.1 Computational grid and model bathymetry

The wave modelling was undertaken on an extended computational grid (Figure A1). The flow and sediment dynamic grid (Figure A2) was nested within this wave grid. The computational grid comprises orthogonal curvilinear co-ordinates in the horizontal and in the case of the Delft3D-

FLOW model sigma co-ordinates are used in the vertical (Deltares 2011b). The set-up of the models used is described in greater detail in below.

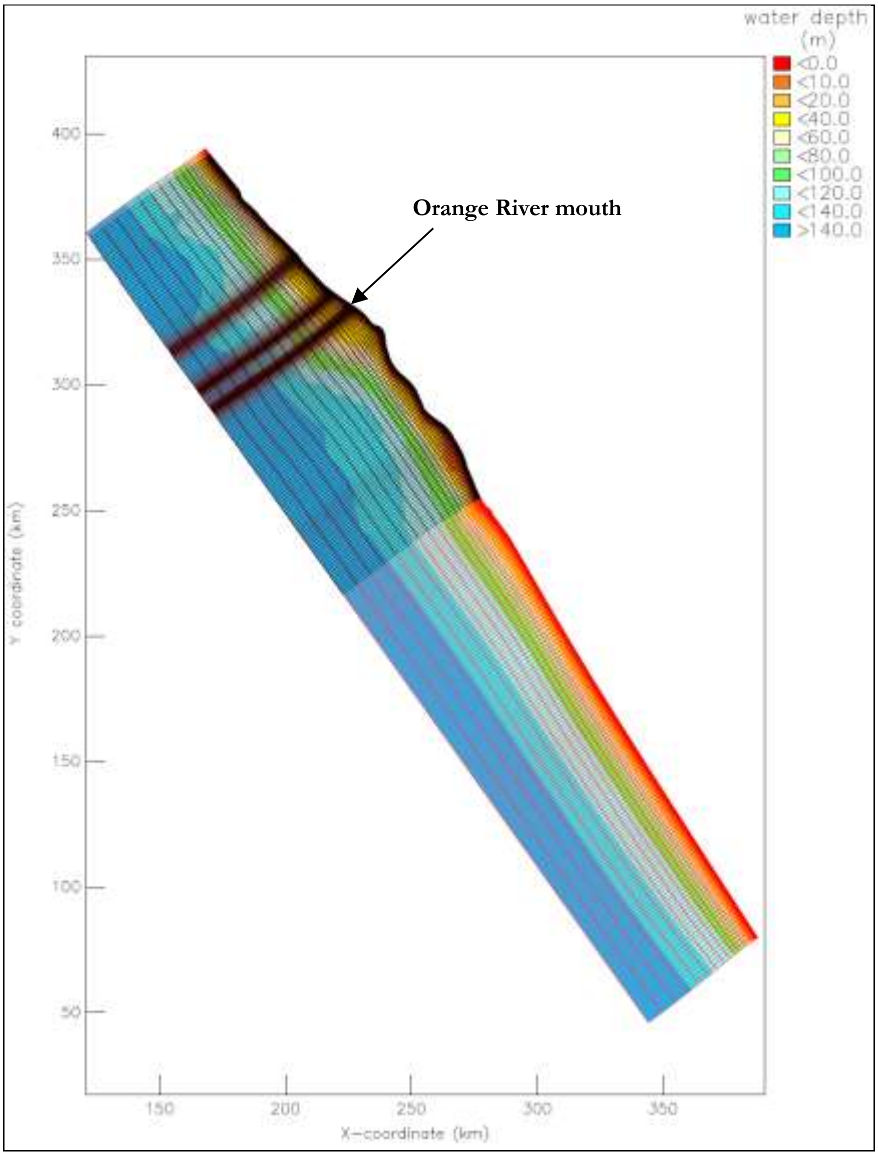

Figure A1. The large computational grids (red) and nested computational grid (black) and bathymetry used in the wave model

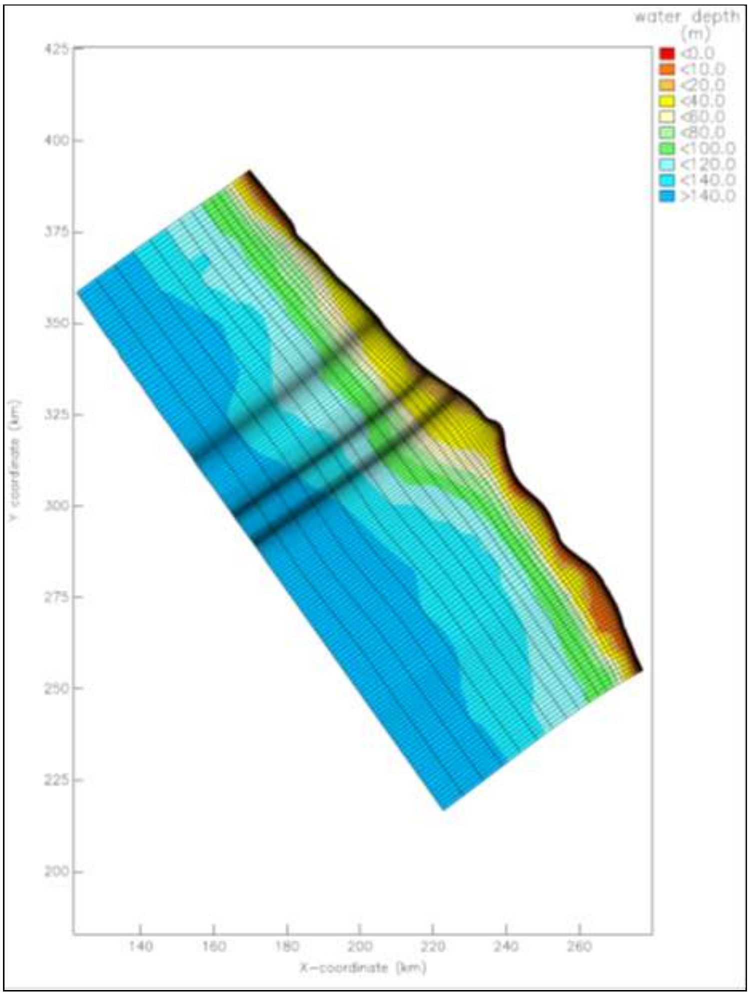

Figure A2. The computational grid and bathymetry used in the flow and sediment dynamics model

The model bathymetry used in this study is the same as that used in the 2006 BCLME study (Smith et al., 2006) that consisted of a combination of:

- South African Navy Hydrographic charts;
  - Chameis Bay to Orange River;
  - Orange River to White Point.
- South African Navy electronic data for the near-shore and offshore area of the Orange River to the Holgat River;
- CSIR surveys of the near-shore (1995), mid-shore (1999) and a combination survey of the offshore and a detailed nearshore survey (2002);
- Namdeb ALS beach survey data (2002)

All data were projected onto a common projection (Clark 1880 spheroid and SALO 17 projection). These data were then converted to model co-ordinates using the following relationships:

$X_{model} = (80\ 000 - Y_{SALo17}) + 200\ 000$

$Y_{model} = (3\ 300\ 000 - X_{SALo17}) + 200\ 000$

It is realised that this is not ideal however these are the co-ordinates used in the original BCLME study (i.e. co-ordinates used by the mining companies in the area).

A.3.2 Delft3D-wave

The wave model used in the Delft3d-wave modelling suite (Deltares, 2011a) is the third generation SWAN model (Simulating Waves Nearshore) that was developed at Delft University of Technology (e.g. Holthuijsen et al., 1993; Booij et al., 1999; Ris et al., 1999). SWAN has been validated and verified in several laboratory and (complex) field cases (e.g. Ris et al.,1999; WL|Delft Hydraulics, 1999, 2000). The SWAN model is based on the discrete spectral action balance equation and is fully spectral (in all directions and frequencies). The latter implies that short-crested random wavefields propagating simultaneously from widely different directions can be accommodated (e.g. a wind sea with super-imposed swell). SWAN computes the evolution of random, short-crested waves in coastal regions with deep, intermediate and shallow water and ambient currents. The SWAN model accounts for (refractive) propagation due to current and depth and represents the processes of wave generation by wind, dissipation due to whitecapping, bottom friction and depth-induced wave breaking and non-linear wave-wave interactions (both quadruplets and triads) explicitly with state-of-the-art formulations. Wave blocking by currents is also explicitly represented in the model.

The wave model was run in an 'off-line' mode. This means that the representative set of wave conditions for the model simulations was simulated in a stand-alone mode and only later coupled into the modelled runs. This was done for the purposes of computational efficiency, i.e. the wave conditions do not have to be simulated anew for each model simulation of flow ranges under consideration. The reader is referred to Smith et al. (2006) for greater detail on the offshore environmental conditions used in the model simulations. The model parameters used in the wave modelling are given in Table A1 below.

Table A1. Model parameterisations used in Delft3D-Wave

Wave model parameters and settings	
Computational Grid filename	wavgrd.grd
Model bathymetry filename	wave01c.dep
Wave conditions filename	md-vwac.001
Wave direction resolution	Circle with 72 directions resolved
Wave frequency resolution	0.05 to 1 Hz with 36 frequency bins
Model duration	27 Feb 1988 to 1 May 1988
Wave spectrum parameterisation	Jonswap (peak enhancement factor 3.3)
Wave forces model	Wave energy dissipation rate
Generation mode for physics	3rd generation
Depth-induced breaking	B&J model (α=1; γ=0.73)

Wave model parameters and settings	
Bottom friction	Madsen et al (friction co-efficient = 0.05)
Wave set-up	None
Whitecapping	Komen et al.
Wave propagation in spectral space	Includes refraction and frequency shifts
Minimum depth in model	0,05 m
Accuracy criteria	
Allowable change	Relative change (Hs-Tm01) = 0.02 Relative change: mean value Hs = 0.02 Tm01 = 0.02
Percentage of grid cells where condition is to be met to terminate iterative computations	99%
Maximum number of iterations	10

The wave conditions applied at the open boundaries of the large wave grid are those used by Smith et al. (2006) in their study of the 1988 Orange River floods (Figure A3).

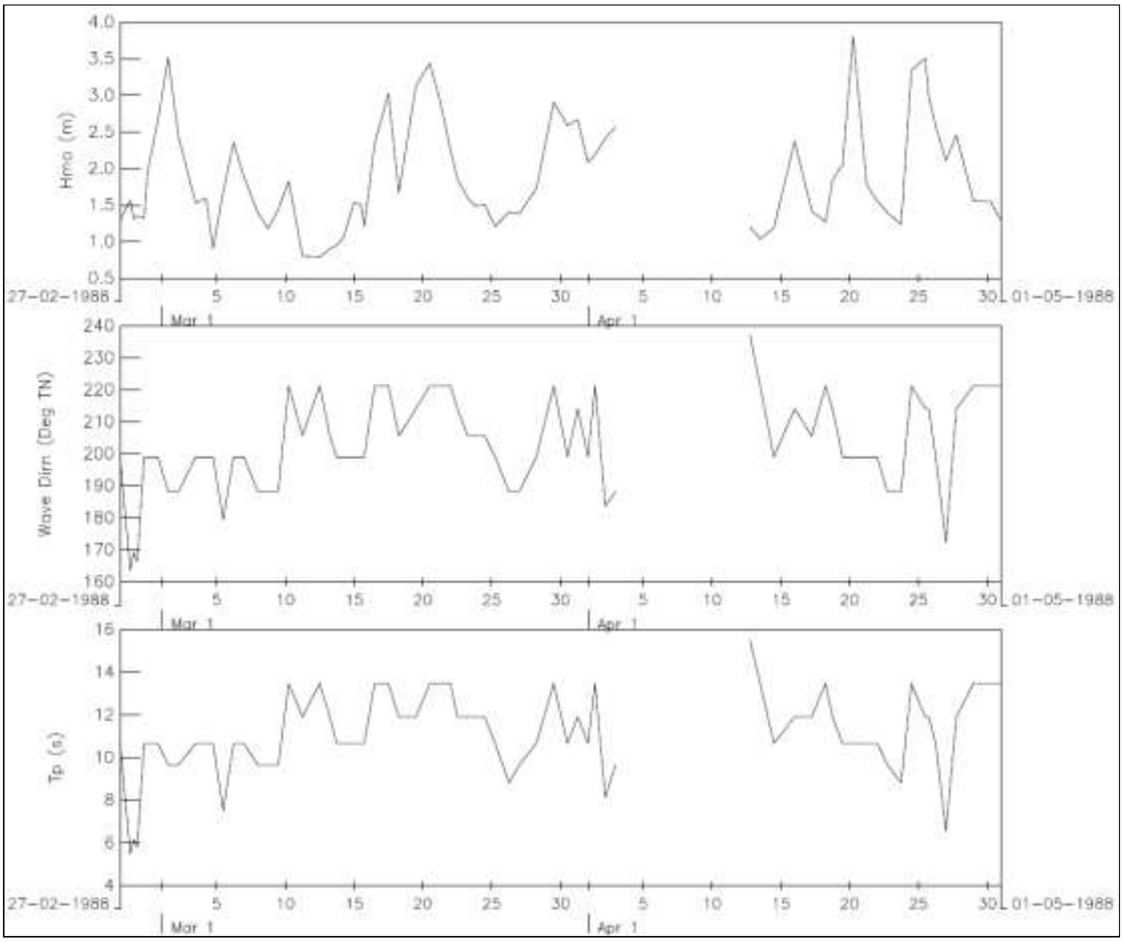

Figure A3. The offshore wave conditions used in the model simulations

These offshore data were obtained as follows. Offshore wave data (non-directional) were extracted from measurements conducted in 100 m water depth offshore from Port Nolloth (29.29°S and 16.81°E). The coverage period spanned the period 01/01/1988 to 30/06/1988. However, these data excluded wave direction measurements. In order to add directional information, a relationship between wave height and direction was established for available directional measurements in the region and applied to the 1988 omnidirectional data set. The directional measurements used for this analysis were conducted by Shell International Exploration and Production at the Kudu Gas Field site. These directional measurements were conducted between 08/03/1998 and 13/04/1999 at a location approximately 180 km west of the Orange River mouth (at 28°37'36" S and 14°34'59"). It should be noted that the relationship between wave direction and period used to estimate the wave directions is relatively poor, however Smith et al. (2006) demonstrated that their model results were not overly sensitive to these potential uncertainties.

A gap exists in the wave data time series for the period 4 April to 12 April 1988. In the model study a simple linear interpolation of data were used to 'plug' this data gap.

A.3.3 Delft3D-Flow/Sed

The Deltares three dimensional hydrodynamic modelling suite (Delft3D-flow) has been used for the study (Deltares, 2011b). The model incorporates the following processes:

- wind, wave and tidally driven flows;
- rotational effects (Coriolis) effects);
- vertical mixing based on sophisticated turbulence closure models;
- air-sea fluxes including evaporation.

A curvilinear computational grid (Figure A2) is used in the model. A total of eight layers are used in the model that uses a sigma coordinate in the vertical (i.e. a co-ordinate where the vertical layers thicknesses are normalised to a local water depth (Deltares, 2011b)).

The hydrodynamic model was set-up to simulate salinity but not water temperature. In addition a total of four sediment fractions were included in the model. These comprised three cohesive sediment fractions (silt, coarse clay and find clays) and a non-cohesive fraction comprising sand. The model included the effects of sediment on fluid density. The parameters used in the model are listed in Table A2 below.

Table A2. Model parameterisations used in Delft3D-Flow and Delft3D-Sed

Wave model parameters and settings	
Computational Grid filename	001.grd
Model bathymetry filename	01c.dep
Model duration	27 Feb 1988 to 1 May 1988
8 vertical layers (% of water column)	5 (surface), 8, 10, 15, 20, 20, 12 and 10
Time step	Circle with 72 directions resolved

Wave model parameters and settings				
Open boundary parameters	Water level time series calculated from TILT (simplified dynamics boundary condition) A = 1000 s^2 Thatcher-Harleman lags= 30 min (surface and bottom)			
Bottom friction	White-Colebrooke = 0.02			
Vertical viscosity (m^2/s)	0.0005			
Vertical eddy diffusivity (m^2/s)	0.000 001			
Threshold depth for wetting/drying (m)	0.1			
Threshold depth for wetting/drying (m)	0.05			
Start-up smoothing time (min)	60			
Advection scheme for momentum & transport	cyclic			
Turbulence closure model	k-ε model			
Forester filters	Both horizontal and vertical set to off			
Sigma-correction	On			
Wind drag co-efficients	0.0011 (0 m/s) and 0.0065 (100 m/s)			
Sediment parameters	sand	silt	coarse clay	fine clay
Specific density (kg/m^3)	2 650	2 650	2 650	2 650
Dry bed density (kg/m^3)	1 600	265	265	265
Settling velocity (mm/s)	-	1	0.5	0.1
Reference density for hindered settling (kg/m^3)	1 600	1 600	1 600	1 600
Critical bed shear stress for sedimentation (N/m^2)	-	0.3	0.15	0.075
Critical bed shear stress for erosion (N/m^2)	-	0.06	0.06	0.06
Erosion parameter ($kg/m^2/s$)	-	0.0003	0.0003	0.0003
Initial sediment thickness on seabed (m)	0.0	0.001	0.001	0.001
D_{50}	100	-	-	-

The wave conditions were run off-line (i.e. pre-computed) and coupled to the hydrodynamic model at run time. Air-sea fluxes have been excluded from the model used in this study as these are deemed to be second order effects and mostly superfluous for the purposes of the present study.

The model is forced by:

- Wind-driven water levels (tidal effects excluded) were specified at the offshore boundaries (Figure A4). The offshore winds used were those measured on an offshore rig, the Nymphea (29° 53.4'S: 16° 17.4'E), *i.e.* approximately 140 km south of the Orange River Mouth, and 75 km perpendicularly offshore;
- Freshwater and sediment discharges into the marine environment for the Orange River.

The salinity on the open boundaries was specified to be 35 psu while the sediment loads on these boundaries were specified to be zero.

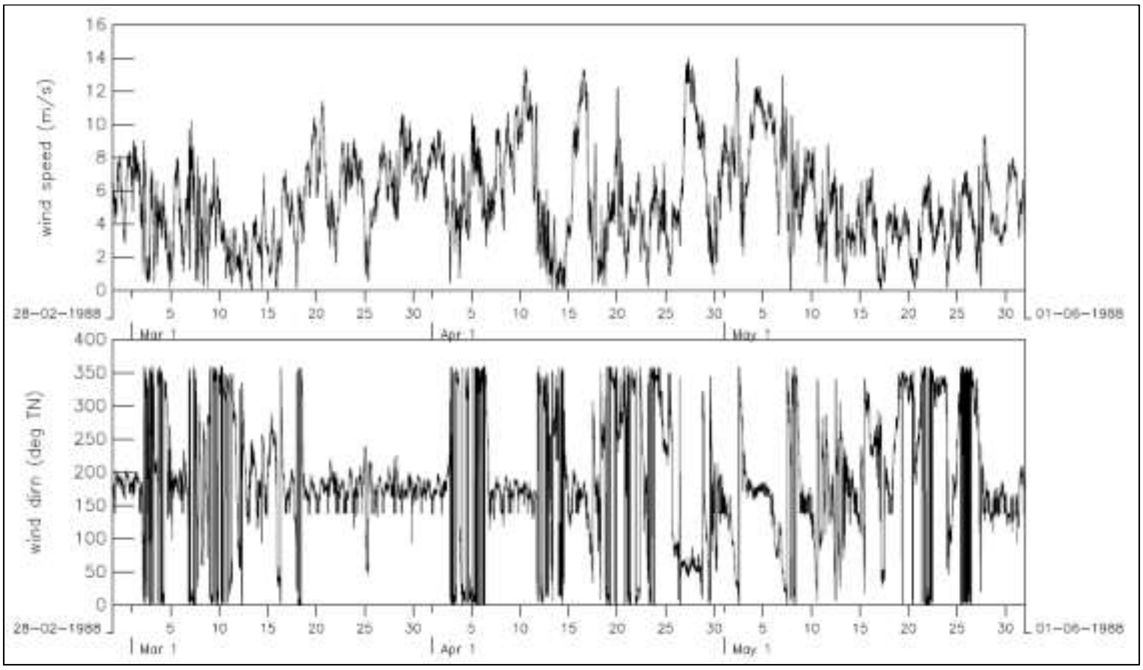

Figure A4. The offshore wind conditions used in the model simulations. These are data measured approximately 84 m above mean sea level on the rig Nymphea (29° 53.4'S; 16° 17.4'E).

Six flood/high freshwater inflow scenarios were simulated in the model. The flows for each of these scenarios are as simulated for the EFR study (ref main report), except for the 1988 flood which is based on flow data reported by Bremner et al., 1990. The sediment loads for each of these flood events have been estimated by regression of the sediment concentration data reported by Bremner et al. (1990) for the 1988 flood. While the grain sizes and concentration of the various fraction varied over the duration of the 1988 floods, in this study we have used all of the data reported by Bremner et al. (1990) to obtain a regression between the total suspended sediment concentration in the flood waters and the flow rates (Figure A5).

The expression for estimating sediment concentration from flow rates is as follows:

$$TSS = (1.97598 \times 10^{-11}) V^3 - (1.25481 \times 10^{-7}) V^2 + (3.6558 \times 10^{-4}) V + 1.05537498$$

where

TSS = total suspended sediment concentration in g/ℓ, and

V = flow rate in m^3/s

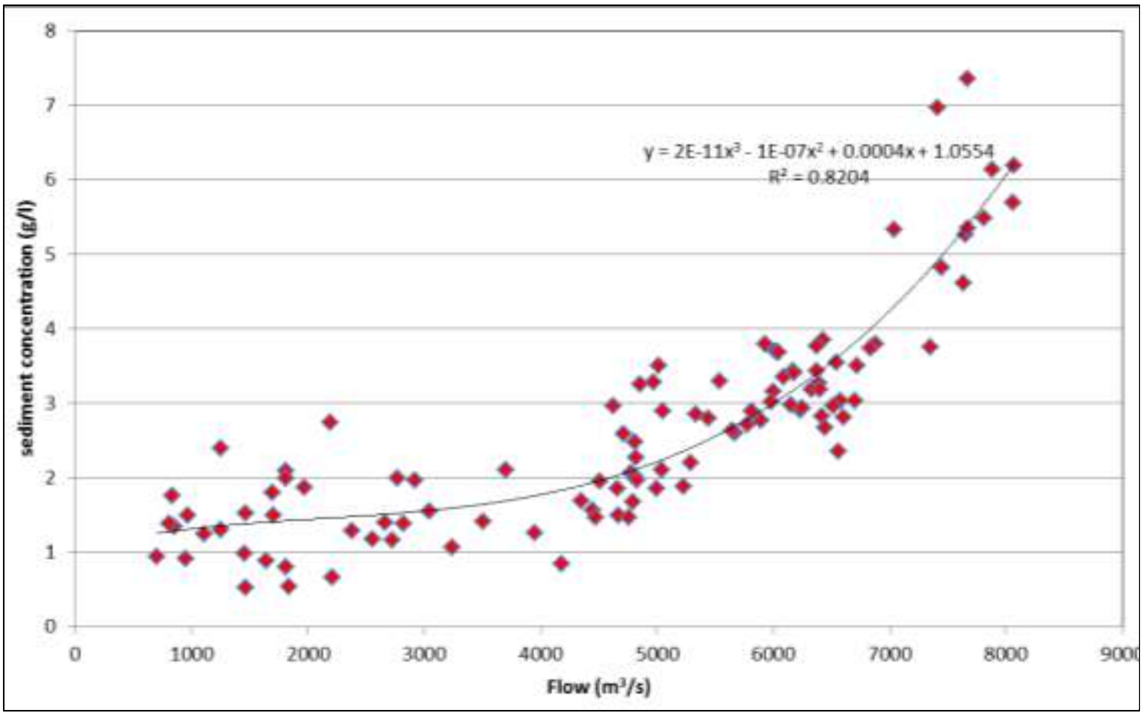

Figure A5. Regression of total suspended sediment concentration versus flow rate for the 1988 Orange River flood

Once the total suspended sediment concentration has been estimated, the fraction of sand, silt and clay were determined based on analyses of Bremner et al. (1990) data by Smith et al. (2006). The percentages used to estimate the rate of the various size fractions of sediments discharged into the marine environment is tabulated below (Table A3). In general the proportion of sand is fairly limited. As no data existed for the split between coarse and fine clays, equal proportions of coarse and fine clays were assumed for all estimated sediment discharge rates.

Table A3. Model parameterisations used in Delft3D-Flow and Delft3D-Sed

Type of sediment	***Size***	***Average percentage of sediment content***	***Minimum percentage recorded***	***Maximum percentage recorded***
Sand	125 to 2000	16	3.4	49.0
Silt	3.9 to 62.5	31	12.8	48.9
Clay	< 3.9	53	24.3	69.9

Analysis of suspended sediment sampled upstream under low flow conditions provides some indication of the detailed grain-size distribution (Table A4). These data confirm existence of a low sand content. A relatively high content of silt was typical of the period prior to major dam construction: historical data show that the silt fraction dominated until the 1970s, after which the clay fraction began to dominate due to the trapping of the coarser fractions by dams (*Bremner et al.*, 1990).

Table A4. Size distribution of Orange River sample at Upington (estimated by Smith et al. (2006) from Rogers (1977)

Percentage of sand is size class type of sediment Size Average percentage of sediment content			***Minimum percentage recorded***	***Maximum percentage recorded***
246 – 420	147 to 246	50* to 147		
Silt	3.9 to 62.5	31	60	22
Clay	<3.9	53	24.3	69.9

* Close to 63 µm which is a typical cut-off between sand and silt.

The estimated flows and sediment discharge rates used in the model simulations are plotted in Figures A6 to A12. Note the scale changes on the vertical axes. This is necessary as the magnitudes of the flows and sediment discharges differ dramatically for the largest (1988 flood) and smallest flows (2009 floods) simulated. As noted above the flows and sediment discharge rates for the 1988 flood are those measured during the flood and reported by Bremner et al. (1990). For all the other flood scenarios simulated the flows used are the simulated flow data produced for the EFR study and the estimated sediment concentrations and associated sediment discharges estimated as described above.

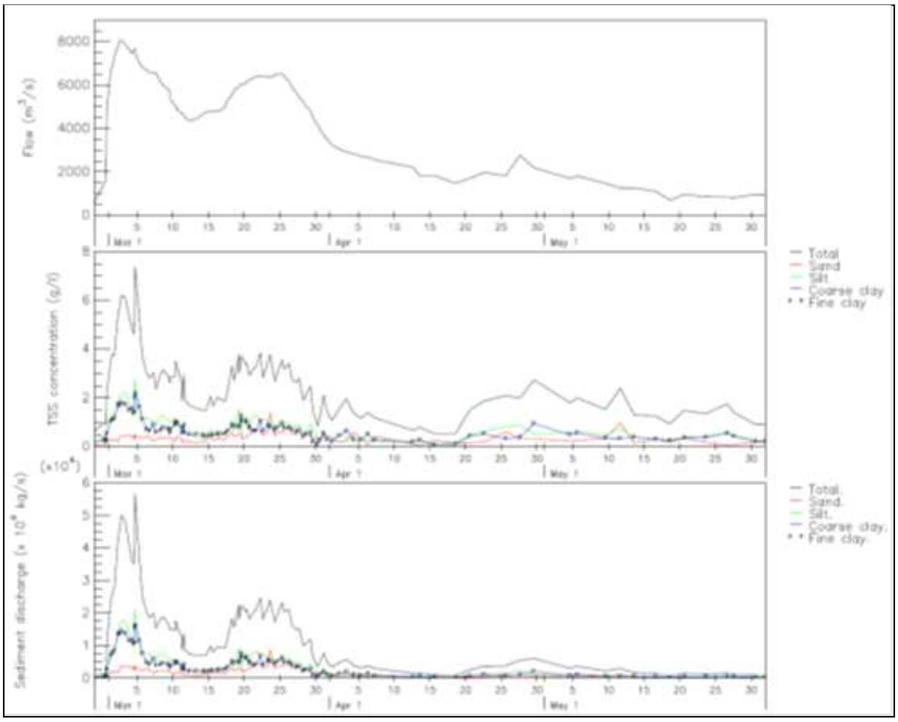

Figure A6. Flow rates, sediment concentrations and sediment discharge rates used to simulate the 1988 flood

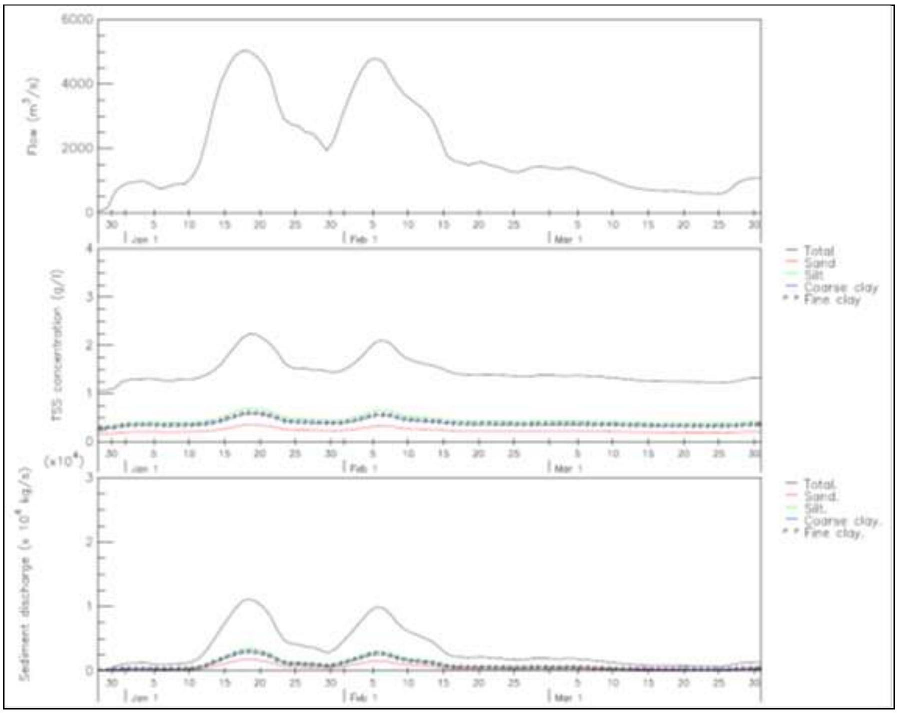

Figure A7.Flow rates, sediment concentrations and sediment discharge rates used to simulate the 2010/2011 flood

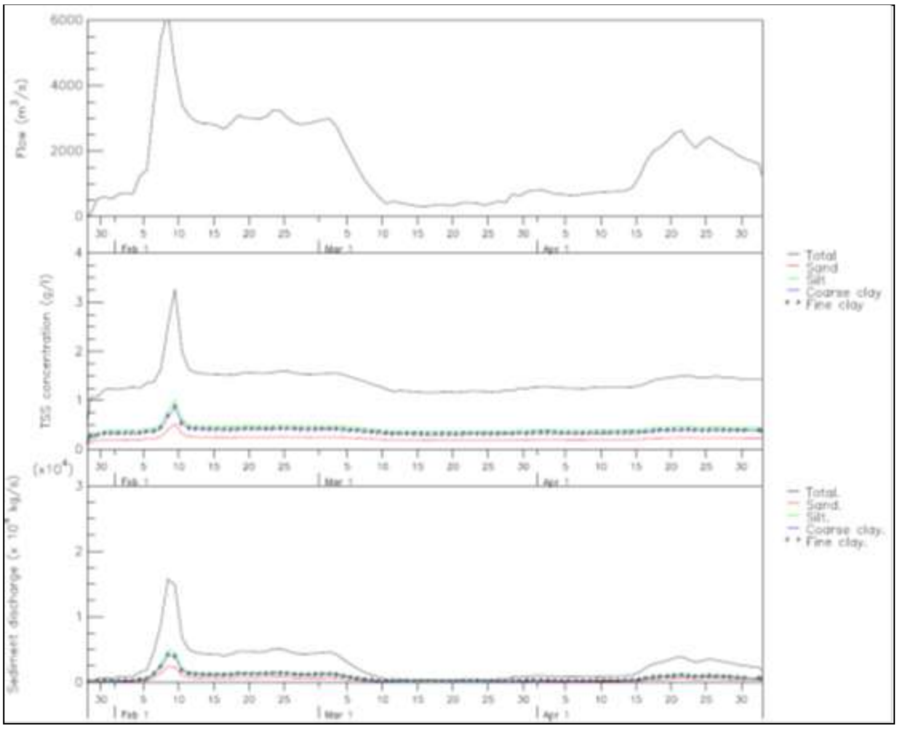

Figure A8. Flow rates, sediment concentrations and sediment discharge rates used to simulate the 1967 flood

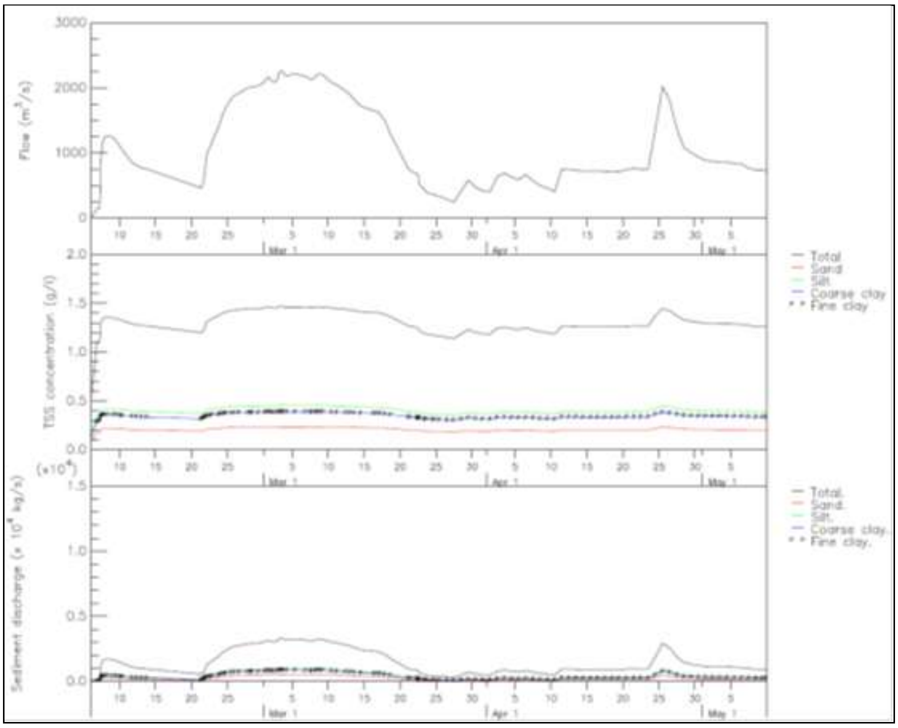

Figure A9. Flow rates, sediment concentrations and sediment discharge rates used to simulate the 1996 flood

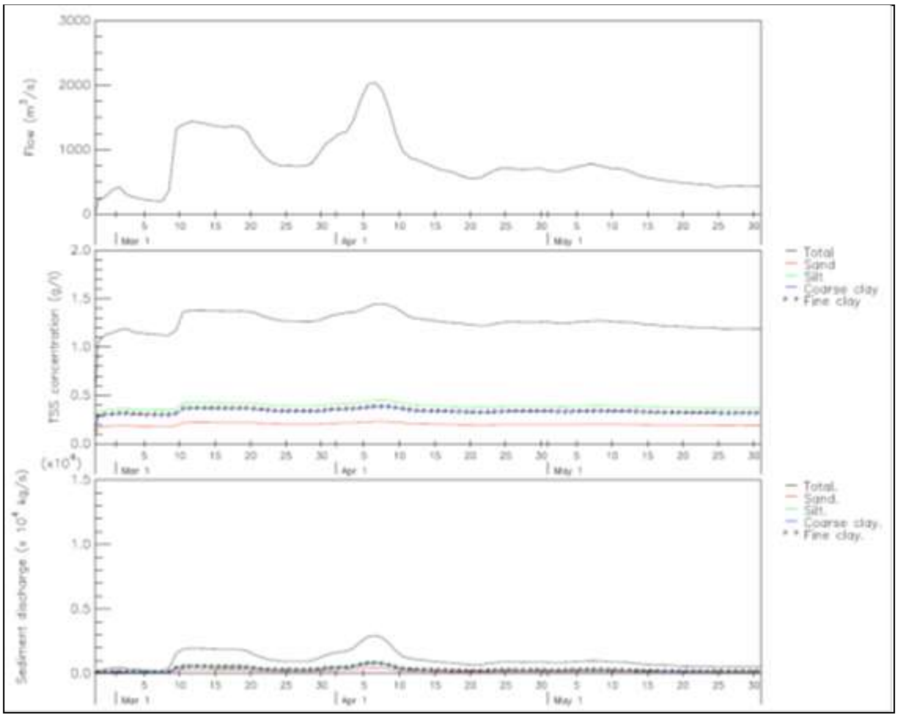

Figure A10. Flow rates, sediment concentrations and sediment discharge rates used to simulate the 2006 flood

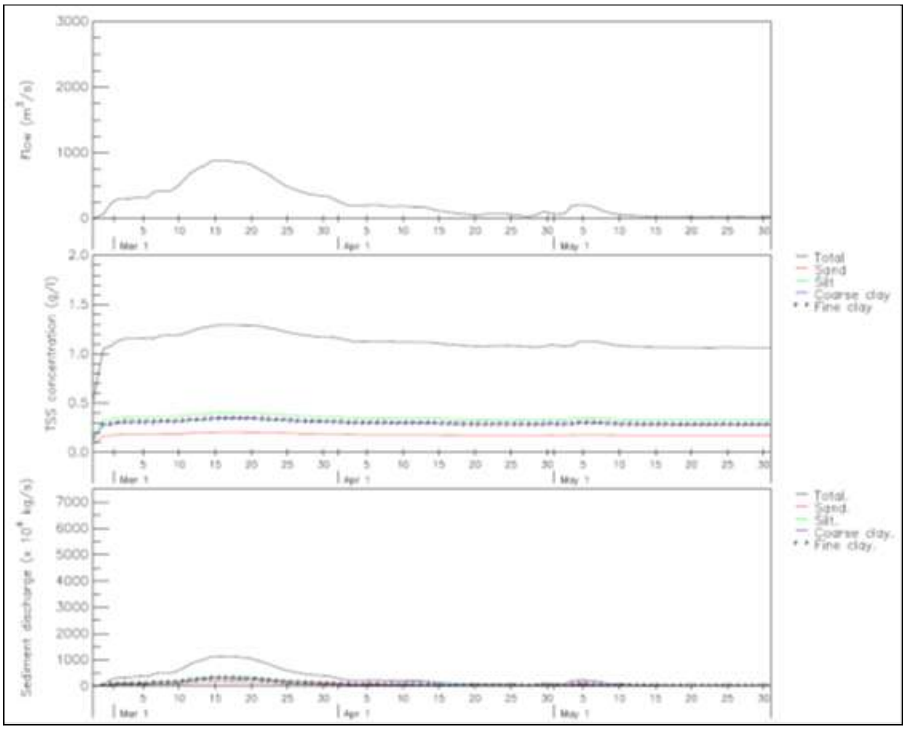

Figure A11. Flow rates, sediment concentrations and sediment discharge rates used to simulate the 2009 flood/'freshette'

A.4 Model scenarios and assessment metrics

In this section of the report a detailed description is provided of the scenarios modelled as well as the metrics used to analyse the model outputs and provide indices of changes for the development being assessed in this study.

A number of flood and high flow periods were selected for simulation in the model, the intention being to characterise the abiotic consequences of such flow events in the offshore marine environment. These characterisations, when used in conjunction with estimated changes in occurrence of these high flow/flood events, can be used to develop indices of change for the various development options under consideration in this study. The flood sizes/flow ranges selected for the model simulations are listed in Table A5 below. Included in the table are approximate flood volumes, sediment tonnages and (very) approximate return periods of the flow ranges simulated.

As noted above, when analysing the results of the simulations, only four flow ranges are used (i.e. as specified in Table A5). This requires that the metrics for two **large floods** and the two **small floods** be averaged to obtain an overall representative metric for each of these two flow ranges.

Table A5. Flow ranges simulated in the modelling study

Flood size	***Estimated return period***	***Year***	***Period***	***Total discharge volume[1] (Mm³)***	***Total discharge of sediments[1] (M tonnes)***
Very large flood	1:100	1988	27/02/1988 – 01/06/1988	24706.99	62.684
Large flood	1:20	2010	27/12/2010 – 01/04/2011	16013.13	26.329
Large flood	1:20	1967	28/01/1967 – 04/05/1967	13764.61	21.570
Small flood	1:2	1996	06/02/1996 – 10/05/1996	10137.15	14.038
Small flood	1:2	2006	27/02/2006 – 02/06/2006	6589.98	8.577
High flow pulse	~annually	2009	27/02/2009 – 02/06/2009	2003.44	2.411

1 Over a three year period.

A.5 Model results

The changes in the water column and on the seabed have been simulated for the six representative flow conditions listed in Table A5. These have been reported in terms of metrics of both pelagic and benthic change for the four flow ranges listed in Table A5.

Plots of surface salinity in the offshore zone are plotted for the approximate peak outflows of the 1988 'Very Large' flood and for the peak out flows of the 1996 'Small' flood. This provides a clear indication of the changes at the sea surface for these two flood sizes. The extent of lower salinity surface waters (<20 psu) is significantly greater (3x) for the very large flood compared to the small flood.

Similarly plots of TSS concentrations in the surface waters and near the seabed (Section 9.5.3) for the same two floods clearly indicate the extent of changes in the surface and near bottom turbidity for a 'Very Large' and a 'Small' flood. The elevation of turbidity in the surface waters is significantly greater for 'Very Large' floods compared to 'Small' floods. The reason for this most probably lies in the changes in surface dynamics resulting from the extensive layer of low salinity (low density) surface waters that spread easily and are strongly influenced by surface winds.

A.6 Changes in freshwater and sediment inputs into the marine environment

The changes in the freshwater, sediment and dissolved reactive silicates inputs into the marine environment have been assessed based on the significance ratings described in Table 9 (section 9.3). Table A6 lists the fluxes into the marine environment over three month period for each of the floods simulated in the model. The fluxes into the marine environment over three month period are listed in Table A7 for the four flood sizes used in the modelling study to characterise floods and significant freshwater pulses into the marine environment from the Orange River. In Table A8 the average annual fluxes (averaged over a 66-year period) are reported for the range of proposed scenarios under consideration in this study. These fluxes are reported as a percentage of those occurring under reference in Tables A9.

In terms of these significance ratings (with respect to inflows to the marine environment) it is not possible to discern between present state and proposed Sc 2 to 5. However Sc 6 and 7, at a rating of -3, compared to the rating of -2 for Sc 2 to 5 (when assessed against reference conditions – see Table A10), are significantly worse than the other proposed scenarios.

The above results (indices of change) characterise only the changes of freshwater and associated fluxes (sediments, nutrients, etc.) into the marine environment. This information can be used to assess potential changes in nearshore and offshore marine environments based on expert opinion and/or model simulated changes in water quality and/or sediment-related marine habitats. The model simulated changes in water quality and sediment-related marine habitats is reported below.

Table A6. Freshwater, sediment and DRS inflows to the marine environment as used in the model simulations

Flood size	***Very large flood***	***Large flood***	***Large flood***	***Small flood***	***Small flood***	***High flow pulse***
Period						
	27/02/1988 – 01/06/1988	27/12/2010 – 01/04/2011	28/01/1967 – 04/05/1967	06/02/1996 – 10/05/1996	27/02/2006 – 02/06/2006	27/02/2009 – 02/06/2009
Total freshwater discharge volume[1] (Mm^3)						
	24706.99	16013.13	13764.61	10137.15	6589.98	2003.44
Total discharge of sediments[1] (M tonnes)						
All sediments	62.684	26.329	13764.61	14.038	8.577	2.411
Sand	9.296	4.213	13764.61	2.246	1.372	0.386
Silt	19.827	8.162	13764.61	4.352	2.659	0.747
Coarse clays	16.781	6.977	13764.61	3.72	2.273	0.639
Fine clays	16.781	6.977	13764.61	3.72	2.273	0.639
Total discharge of DRS[1](M tonnes)						
	0.119	0.077	13764.61	0.049	0.032	0.01

1 Over a three month period

Table A7. Freshwater, sediment and DRS inflows to the marine environment for the characteristic flood sizes used in the assessment of potential future scenarios

Flood size	***Very large flood***	***Large flood***	***Small flood***	***High flow pulse***
Total freshwater discharge volume[1] (Mm³)				
	24706.99	14888.87	8363.57	2003.44
Total discharge of sediments[1] (M tonnes)				
All sediments	62.684	23.949	11.307	2.411
Sand	9.296	3.832	1.809	0.386
Silt	19.827	7.424	3.505	0.747
Coarse clays	16.781	6.347	2.996	0.639
Fine clays	16.781	6.347	2.996	0.639
Total discharge of DRS[1](M tonnes)				
	0.119	0.071	0.04	0.01

1 Over a three month period

Table A8. Freshwater, sediment and DRS inflows to the marine environment under the various proposed future scenarios

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume[1] (Mm³)	39095.77	15550.16	15414.29	15381.13	17429.28	14407.08	8972.78	8972.78
Total discharge of sediments[1] (M tonnes)								
All sediments	53.111	21.246	21.139	21.134	23.281	19.386	11.922	11.922
Sand	8.487	3.388	3.371	3.37	3.714	3.091	1.908	1.908
Silt	16.47	6.592	6.559	6.557	7.223	6.016	3.696	3.696
Coarse clays	14.077	5.633	5.604	5.603	6.172	5.14	3.159	3.159
Fine clays	14.077	5.633	5.604	5.603	6.172	5.14	3.159	3.159
Total discharge of DRS[1] (M tonnes)	0.188	0.075	0.074	0.074	0.084	0.069	0.043	0.043

1 Annual average over a 66-year period.

Table A9. Freshwater, sediment and DRS inflows to the marine environment under the various proposed future scenarios. The inflows for the various scenarios are expressed as percentages relative the inflows occurring under reference conditions.

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume[1] (Mm^3)	100.00%	39.80%	39.40%	39.30%	44.60%	36.90%	23.00%	23.00%
Total discharge of sediments[1] (M tonnes)								
All sediments	100.00%	40.00%	39.80%	39.80%	43.80%	36.50%	22.40%	22.40%
Sand	100.00%	39.90%	39.70%	39.70%	43.80%	36.40%	22.50%	22.50%
Silt	100.00%	40.00%	39.80%	39.80%	43.90%	36.50%	22.40%	22.40%
Coarse clays	100.00%	40.00%	39.80%	39.80%	43.80%	36.50%	22.40%	22.40%
Fine clays	100.00%	40.00%	39.80%	39.80%	43.80%	36.50%	22.40%	22.40%
Total discharge of DRS[1] (M tonnes)	100.00%	39.80%	39.40%	39.30%	44.60%	36.90%	23.00%	23.00%

1 Annual average over a 66-year period.

Table A10. Freshwater, sediment and DRS inflows to the marine environment under the various proposed future scenarios. The inflows for the various scenarios are expressed in terms of the significance ratings specified in Table 9. All ratings are relative to reference conditions.

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume[1]	0	-2	-2	-2	-2	-2	-3	-3
Total discharge of sediments[1]								
All sediments	0	-2	-2	-2	-2	-2	-3	-3
Sand	0	-2	-2	-2	-2	-2	-3	-3
Silt	0	-2	-2	-2	-2	-2	-3	-3
Coarse clays	0	-2	-2	-2	-2	-2	-3	-3
Fine clays	0	-2	-2	-2	-2	-2	-3	-3
Total discharge of DRS[1]	0	-2	-2	-2	-2	-2	-3	-3

1 Annual average over a 66-year period.

A.7 Changes in offshore benthic habitats

Changes in the offshore benthic habitats have been characterised in terms of the thickness of sediments deposited in the marine environment as well as the sediment grain size distributions considered to constitute appropriate benthic habitats for the species utilising these habitats.

The changes in the extent and quality of benthic habitats have been assessed based on the significance ratings described in Table A11. Table A12 lists the extent of deposited sediments and benthic habitats meeting the specified benthic habitat quality criteria from the model outputs for the flood sizes and flow ranges modelled, while Table A13 lists the same but only for the four flow ranges/flood sizes used in this study to characterise floods and significant freshwater pulses into the marine environment from the Orange River. In Table A13 the extent of deposited sediments and the extent of benthic habitats meeting the specified benthic habitat quality criteria are reported for the various scenarios under consideration. These same extents of benthic habitat are reported as a percentage of the extent of habitats expected to occur under reference (Table A14). The significance of the expected changes under the various scenarios have been mapped according to the specifications in Table A15 and are reported as significance rating of between -3 and 3. In terms of these significance ratings (with respect to the extent of benthic habitats) it is not possible to discern between present state and Sc 2 to 5. However Sc 6 and 7, at a rating of -3, compared to the rating of -2 for Sc 2 to 5 (when assessed against reference conditions – see Table A15), are significantly worse than the other proposed scenarios.

Table A11. Surface area of sediment deposition meeting the criteria listed in the table for the various high flow/flood scenarios used in the model simulations

Flood size	***Very large flood***	***Large flood***	***Large flood***	***Small flood***	***Small flood***	***High flow pulse***
Total discharge of sediments[1] (M tonnes)						
	62.684	26.329	21.57	14.038	8.577	2.411
Surface area of sediment deposition > 0.01 m						
Median	1046.176	159.087	238.256	24.759	8.953	0.014
80%tile	1390.41	440.679	553.324	198.48	66.642	26.158
Max	1544.304	673.059	749.43	341.602	244.587	47.334
Surface area of sediment deposition >0.01 m and <40% clay						
Median* compliance	140.679	31.413	69.682	8.199	0.423	0.12
20%* compliance	370.452	224.614	261.52	122.46	72.471	26.485
Minimum* compliance	931.358	603.09	650.242	356.856	281.552	78,990
Surface area of sediment deposition >0.01 m and <40% clay and >20% sand						
Median compliance	3.173	1.581	1.314	0.879	0.422	0.12
20%* compliance	4.484	3.096	2.538	2.072	1.487	0.51
Minimum* compliance	8.154	7.135	7.452	5.596	5.246	3.168

1 Over a 3 month period
* Here median, 20% and minimum compliance refers to the condition being met for 50% of the time, 20% of the time and at least once during the simulation, respectively.

Table A12. Surface area of sediment deposition meeting the criteria listed in the table for the characteristic flood sizes used in the assessment of proposed future scenarios

Flood size	***Very large flood***	***Large flood***	***Large flood***	***Small flood***	***Small flood***	***High flow pulse***
Total discharge of sediments[1] (M tonnes)						
	62.68	23.95	11.31	2.41	62.68	23.95
Surface area of sediment deposition > 0.01 m						
Median	1046.18	198.67	16.86	8.95	1046.18	198.67
80%tile	1390.41	497	132.56	26.16	1390.41	497
Max	1544.3	711.24	293.09	47.33	1544.3	711.24
Surface area of sediment deposition >0.01 m and <40% clay						
Median* compliance	140.68	50.55	4.31	0.42	140.68	50.55
20%* compliance	370.45	243.07	97.47	26.49	370.45	243.07
Minimum* compliance	931.36	626.67	319.2	78.99	931.36	626.67
Surface area of sediment deposition >0.01 m and <40% clay and >20% sand						
Median compliance	3.17	1.45	0.65	0.12	3.17	1.45
20%* compliance	4.48	2.82	1.78	0.51	4.48	2.82
Minimum* compliance	8.15	7.29	5.42	3.17	8.15	7.29

1 Over a 3 month period
* Here median, 20% and minimum compliance refers to the condition being met for 50% of the time, 20% of the time and at least once during the simulation, respectively.

Table A13. Surface area of sediment deposition meeting the criteria listed in the table for the various high flow/flood scenarios under the various proposed future scenarios

Flood size	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total discharge of sediments[1] (M tonnes)								
	53.111	21.246	21.139	21.134	23.281	19.386	11.922	11.922
Surface area of sediment deposition > 0.01 m								
Median	181.289	74.677	73.137	75.924	86.805	70.539	28.557	28.557
80%tile	700.033	277.498	276.786	279.876	297.786	247.74	144.022	144.022
Max	1327.954	527.143	528.273	528.596	554.177	466.093	295.231	295.231
Surface area of sediment deposition >0.01 m and <40% clay								
Median* compliance	34.448	13.217	13.304	13.9	13.151	10.922	5.259	5.259
20%* compliance	498.79	196.451	194.06	194.917	223.675	183.114	110.096	110.096
Minimum* compliance	1526.596	604.829	598.993	598.765	680.836	561.117	346.982	346.982
Surface area of sediment deposition >0.01 m and <40% clay and >20% sand								
Median compliance	2.975	1.184	1.182	1.182	1.269	1.062	0.668	0.668

Flood size	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
20%* compliance	8.623	3.436	3.386	3.378	3.978	3.259	1.99	1.99
Minimum* compliance	34.278	13.647	13.078	13.134	18.038	14.224	7.784	7.784

1 Annual average over a 66-year period.
* Here median, 20% and minimum compliance refers to the condition being met for 50% of the time, 20% of the time and at least once during the simulation, respectively.

Table A14. Surface area of sediment deposition meeting the criteria listed in the table for the various high flow/flood scenarios under the various proposed future scenarios reported as a percentage of the surface areas occurring under reference conditions

Flood size	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total discharge of sediments[1] (M tonnes)								
	53.111	21.246	21.139	21.134	23.281	19.386	11.922	11.922
Surface area of sediment deposition > 0.01 m								
Median	100.00%	41.20%	40.30%	41.90%	47.90%	38.90%	15.80%	15.80%
80%tile	100.00%	39.60%	39.50%	40.00%	42.50%	35.40%	20.60%	20.60%
Max	100.00%	39.70%	39.80%	39.80%	41.70%	35.10%	22.20%	22.20%
Surface area of sediment deposition >0.01 m and <40% clay								
Median* compliance	100.00%	38.40%	38.60%	40.30%	38.20%	31.70%	15.30%	15.30%
20%* compliance	100.00%	39.40%	38.90%	39.10%	44.80%	36.70%	22.10%	22.10%
Minimum* compliance	100.00%	39.60%	39.20%	39.20%	44.60%	36.80%	22.70%	22.70%
Surface area of sediment deposition >0.01 m and <40% clay and >20% sand								
Median compliance	100.00%	39.80%	39.70%	39.70%	42.70%	35.70%	22.40%	22.40%
20%* compliance	100.00%	39.80%	39.30%	39.20%	46.10%	37.80%	23.10%	23.10%
Minimum* compliance	100.00%	39.80%	38.20%	38.30%	52.60%	41.50%	22.70%	22.70%

1 Annual average over a 66-year period.
* Here median, 20% and minimum compliance refers to the condition being met for 50% of the time, 20% of the time and at least once during the simulation, respectively.

Table A15. Surface area of sediment deposition meeting the criteria listed in the table for the various high flow/flood scenarios under the various proposed future scenarios reported as a significance rating relative to reference conditions

Flood size	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total discharge of sediments[1] (M tonnes)								
	53.111	21.246	21.139	21.134	23.281	19.386	11.922	53.111
Surface area of sediment deposition > 0.01 m								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-2	-2	-3	-3
Surface area of sediment deposition >0.01 m and <40% clay								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-2	-2	-3	-3
Surface area of sediment deposition >0.01 m and <40% clay and >20% sand								
Median	0	-2	-2	-2	-2	-2	-3	0
80%tile	0	-2	-2	-2	-2	-2	-3	0
Max	0	-2	-2	-2	-1	-2	-3	0

1 Annual average over a 66-year period.

A.8 Changes in pelagic habitats in the offshore marine environment

The changes in offshore pelagic habitats have been characterised using the distribution of water quality parameters such as salinity, TSS concentrations/turbidity and dissolved nutrients, particularly DRS.

A.8.1 Salinity

Two measures of low salinity habitat have been used to characterise the effect of freshwater inflows on the salinity of marine pelagic environments. The first is the **spatial** extent of surface waters where the salinity remains below specified threshold(s), the second being the **volume** of marine waters where the salinity remains below the specified threshold(s). Given the nature of the plume dynamics, i.e. that the low salinity waters in the flood water plumes in the marine environment are confined primarily to the surface waters, it is felt that the measure based on the volume of water not exceeding the specified thresholds would be a better measure of change in the extent of pelagic habitats associated with the various flood conditions. Two thresholds were considered, namely a large change (acute) in salinity represented by waters with a salinity <28 psu and a more subtle change (chronic) in salinity represented by waters with a salinity <33 psu.

The extent of changes in pelagic habitats (surface areas and volumes of low salinity waters) is reported in Table A16 (for the events simulated in the model simulations) and Table A17 (for the flood sizes used in the assessment). The annual average extent of these low salinity conditions for the various scenarios are reported in Table A18, while these extents expressed as a percentage of those expected under reference. The corresponding significance ratings of these changes are reported relative to reference and provided in Table A19. In terms of these significance ratings (with respect to the extent of changes in pelagic habitats) it is not possible to discern between present state and Sc 2 to 5. However Sc 6 and 7, at a rating of -3, compared to the rating of -2 for Sc 2 to 5 (when assessed against reference conditions – see Table A20), are significantly worse than the other proposed scenarios.

Table A16. Extent of low salinity waters (<28 psu and <33 psu) expressed as both surfaces areas of low salinity surface waters and volumes of lows salinity waters (i.e. includes a depth component) for the various high flow/flood scenarios used in the model simulations

Flood size	***Very large flood***	***Large flood***	***Large flood***	***Small flood***	***Small flood***	***High flow pulse***
Total freshwater discharge volume[1] (Mm^3)						
	24706.99	16013.13	13764.61	10137.15	6589.98	2003.44
Area of sea surface where salinity < 33 psu[1] (km^2)						
Median	898.34	451.735	211.038	180.091	97.525	5.901
80%tile	2980.952	1203.231	957.483	521.253	319.07	89.672
Max	6852.077	4470.679	5009.462	1987.991	1761.222	732.854
Volume of water where salinity < 33 psu[1] (km^2)						
Median	4.941	1.894	0.874	0.698	0.315	0.018
80%tile	26.214	7.62	5.423	2.328	1.293	0.273
Max	93.029	53.519	58.044	17.053	14.004	4.484
Area of sea surface where salinity <28 psu[1] (km^2)						
Median	117.178	53.824	27.181	19.277	12.562	0.532
80%tile	412.269	140.141	118.721	49.252	32.229	8.46
Max	1084.704	605.617	586.938	165.278	150.875	54.94
Volume of water where salinity <28 psu[1] (km^2)						
Median	0.42	0.175	0.096	0.053	0.034	0.003
80%tile	1.664	0.442	0.388	0.144	0.094	0.02
Max	5.439	2.833	2.612	0.641	0.539	0.179

1 Annual average over a 66-year period.

Table A17. Extent of low salinity waters (<28 psu and <33 psu) expressed as both surfaces areas of low salinity surface waters and volumes of lows salinity waters (i.e. includes a depth component) for the characteristic flood sizes used in the assessment of proposed future scenarios

Flood size	***Very large flood***	***Large flood***	***Small flood***	***High flow pulse***
Total freshwater discharge volume[1] (Mm³)				
	24706.99	14888.87	8363.57	2003.44
Area of sea surface where salinity < 33 psu[1] (km²)				
Median	898.34	331.39	138.81	5.9
80%tile	2980.95	1080.36	420.16	89.67
Max	6852.08	4740.07	1874.61	732.85
Volume of water where salinity < 33 psu[1] (km²)				
Median	4.941	1.384	0.507	0.018
80%tile	26.214	6.522	1.811	0.273
Max	93.029	55.782	15.529	4.484
Area of sea surface where salinity <28 psu[1] (km²)				
Median	117.18	40.5	15.92	0.53
80%tile	412.27	362.17	40.74	8.46
Max	1084.7	596.28	158.08	54.94
Volume of water where salinity <28 psu[1] (km²)				
Median	0.42	0.136	0.044	0.003
80%tile	1.664	0.415	0.119	0.02
Max	5.439	2.723	0.59	0.179

1 Annual average over a 66-year period.

Table A18. Extent of low salinity waters (<28 psu and <33 psu) expressed as both surfaces areas of low salinity surface waters and volumes of lows salinity waters (i.e. includes a depth component) for the various high flow/flood scenarios under the proposed future scenarios

Flood size	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume[1] (Mm³)								
	39095.77	15550.16	15414.29	15381.13	17429.28	14407.08	8972.78	8972.78
Area of sea surface where salinity < 33 psu[1] (km²)								
Median	541.873	216.819	221.609	220.678	201.595	177.061	121.637	121.637
80%tile	2053.085	819.694	815.695	818.4	895.472	744.88	448.866	448.866
Max	10797.988	4251.821	4148.265	4179.29	5154.614	4142.301	2363.272	2363.272
Volume of water where salinity < 33 psu[1] (km²)								
Median	2.047	0.827	0.846	0.845	0.766	0.675	0.445	0.445
80%tile	9.127	3.694	3.706	3.739	3.828	3.239	1.867	1.867
Max	87.588	34.18	33.731	34.142	38.966	31.715	18.346	18.346

Flood size	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Area of sea surface where salinity <28 psu[1] (km^2)								
Median	62.5	25.041	25.627	25.55	23.049	20.309	13.868	13.868
80%tile	293.301	108.154	107.827	111.975	115.103	92.943	51.068	51.068
Max	953.334	372.621	365.655	370.834	437.111	352.857	197.758	197.758
Volume of water where salinity <28 psu[1] (km^2)								
Median	0.19	0.076	0.077	0.077	0.073	0.063	0.04	0.04
80%tile	0.605	0.245	0.245	0.247	0.257	0.217	0.125	0.125
Max	3.622	1.41	1.39	1.416	1.606	1.303	0.725	0.725

1 Annual average over a 66-year period.

Table A19. Extent of low salinity waters (<28 psu and <33 psu) expressed as both surfaces areas of low salinity surface waters and volumes of lows salinity waters (i.e. includes a depth component) for the various high flow/flood scenarios under the various proposed future scenarios , reported as a percentage of the surface areas observed under reference conditions

Flood size	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume[1] (Mm^3)								
	100.00%	39.80%	39.40%	39.30%	44.60%	36.90%	23.00%	23.00%
Area of sea surface where salinity < 33 psu[1]								
Median	100.00%	40.00%	40.90%	40.70%	37.20%	32.70%	22.40%	22.40%
80%tile	100.00%	39.90%	39.70%	39.90%	43.60%	36.30%	21.90%	21.90%
Max	100.00%	39.40%	38.40%	38.70%	47.70%	38.40%	21.90%	21.90%
Volume of water where salinity < 33 psu[1]								
Median	100.00%	40.40%	41.30%	41.30%	37.40%	33.00%	21.80%	21.80%
80%tile	100.00%	40.50%	40.60%	41.00%	41.90%	35.50%	20.50%	20.50%
Max	100.00%	39.00%	38.50%	39.00%	44.50%	36.20%	20.90%	20.90%
Area of sea surface where salinity <28 psu[1]								
Median	100.00%	40.10%	41.00%	40.90%	36.90%	32.50%	22.20%	22.20%
80%tile	100.00%	36.90%	36.80%	38.20%	39.20%	31.70%	17.40%	17.40%
Max	100.00%	39.10%	38.40%	38.90%	45.90%	37.00%	20.70%	20.70%
Volume of water where salinity <28 psu[1]								
Median	100.00%	40.00%	40.60%	40.80%	38.50%	33.30%	21.30%	21.30%
80%tile	100.00%	40.50%	40.50%	40.90%	42.50%	35.80%	20.60%	20.60%
Max	100.00%	38.90%	38.40%	39.10%	44.30%	36.00%	20.00%	20.00%

1 Annual average over a 66-year period.

Table A20. Changes in the extent of low salinity waters (<28 psu and <33 psu) expressed as both surfaces areas of low salinity surface waters and volumes of lows salinity waters (i.e. includes a depth component) for the various high flow/flood scenarios under the various proposed future scenarios, reported as a significance rating relative to reference conditions

Flood size	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume[1] (Mm^3)								
	0	-2	-2	-2	-2	-2	-3	-3
Area of sea surface where salinity < 33 psu[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-2	-2	-3	-3
Volume of water where salinity < 33 psu[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-2	-2	-3	-3
Area of sea surface where salinity <28 psu[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-2	-2	-3	-3
Volume of water where salinity <28 psu[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-2	-2	-3	-3

1 Annual average over a 66-year period.

A.8.2 Nutrients

Compared with concentrations typically measured in seawater in the area (<1,000 µg/ℓ), river inflow can be a significant source of DRS (average 4,800 µg/ℓ) to the adjacent marine environment. Consequently the spatial extent of surface waters as well as the volume of marine waters exceeding the typical ambient concentrations of DRS in the marine environment by 20% and 75% have been calculated (i.e. where the resulting DRS concentrations are 1,2 and 1,75 times greater than the ambient DRS concentrations in the ocean). These areas and volumes are considered to represent the extent to which river inflows create and enhanced nutrient environment with respect to DRS in the adjacent marine environment. Similar to the measures of changes in the extent of benthic habitats, these spatial extents and volumes are reported in terms of the median, 80%tile and maximums observed.

Nutrient distributions have not been explicitly simulated in the model. However based on dilution arguments, the contours for a salinity of 28 psu represents DRS concentration contours of 1,75 times ambient DRS concentrations in the marine environment (i.e. a 75% elevation in DRS compared to ambient due to the freshwater inflows from the Orange River). Similarly salinity contours of 33 psu roughly approximate DRS elevations of 25% above ambient. For this reason figures and tables are not provided for DRS concentrations as all of the relevant information can be gleaned from the distributions and spatial extents and volumes of salinities.

As a consequence of the above relationships, the conclusions for elevated DRS in the marine environment are the same as those for reductions in salinity. The corresponding significance ratings of the changes in DRS reported reference conditions (see Table A19 and A20, respectively), indicate that in terms of these significance ratings (with respect to the extent of changes in pelagic habitats) it is not possible to discern between present state and proposed Sc 2 to 5. However Sc 6 and 7, at a rating of -3, compared to the rating of -2 for Sc 2 to 5, are significantly worse than the other proposed scenarios.

A.8.3 Total suspended solids

The extent of waters with high TSS concentrations or highly turbid waters also is considered to be a measure of pelagic habitat quality. The effects of elevated TSS concentrations (and turbidity) may be both positive and negative, however excessively high water column turbidity (e.g. TSS >100 mg/ℓ) is considered to constitute a degraded environment for most species in the region (EMBECON, 2004). More subtle changes are expected at lower TSS thresholds (e.g. 20 mg/ℓ). Consequently the spatial extent of surface waters as well as the volume of marine waters exceeding the TSS concentrations of 100 mg/ℓ and 20 mg/ℓ has been calculated. These areas and volumes are considered to represent the extent to which river inflows have discernible (positive or negative) TSS or turbidity effects in the marine environment. Similar to the measures of changes in the extent of benthic habitats, these spatial extents and volumes are reported in terms of the median, 80%tile and maximum TSS concentrations observed. In the model all TSS concentrations observed are related to the river inflows as the background or ambient TSS concentrations of the marine waters have not been simulated in the model.

Elevated TSS concentrations occur mainly to the north of the Orange River mouth. However there is also elevated TSS concentrations observed to the south of the Orange River mouth that most probably are due to the initial plume dynamics occurring during the peak inflows of freshwater into the marine environment. The theoretical behaviour for a large freshwater plume discharged into a relatively quiescent offshore marine environment is that the plume should move southwards (Shillington et al., 1990). The bias of high TSS concentrations towards the north is associated with the strong south to southeasterly winds that prevailed during the model simulation period. The elevated TSS concentrations extend further offshore in the bottom waters that in the surface waters. This is related to the re-suspension, advection and re-distribution of the bottom sediments (mostly into deeper waters) over time. The elevated TSS concentrations in the surface waters are related primarily to the initial plume dynamics which result in fairly extensive distribution of low salinity but high turbidity surface waters. Small floods and high freshwater inflow pulses result in significantly less extensive elevations in TSS concentrations in the surface waters, however the extent of elevated turbidity in the bottom waters is fairly extensive for all but the smallest flows simulated (i.e. high freshwater inflow pulses) where elevated turbidity is restricted mainly to the near-shore areas.

The model simulations are of limited duration therefore the model outputs do not necessarily fully reflect the persistence of this elevated turbidity. It is expected that elevated turbidity will persist near the seabed over a fairly extensive area for relatively long periods after flood events. Persistent elevation of turbidity in the surface waters is expected only in the near-shore and even then will only persist until the (predominantly fine) sediments have been re-distributed into deeper waters (water depths deeper than -25 to -30 m chart datum) by wind and wave-driven turbulence. The elevated turbidity for small floods and freshwater pulses is indicated by the modelling results to extend only northwards. This is expected as nearshore flows are predominantly northwards, however the model results may be biased in that the offshore conditions simulated in the model are summer conditions when predominantly south to southeasterly flows predominate, i.e. the model does not fully take into account potential south-easterly wind-driven flows driven north-westerly winds associated with larger coastal lows and storms during the winter months.

Based on the findings in EMBECON (2004), the higher the elevation of TSS concentrations (and therefore turbidity) and the more extensive the elevated TSS concentrations, the greater the impacts in the marine environment. However it should be noted that changes in turbidity can be both detrimental and beneficial, depending on the specific ecological effects being considered. In keeping with the principle that changes from the status quo are deemed to be detrimental, the indices used to assess the effects of changes in TSS concentrations on the marine environment can be interpreted as being highly detrimental for highly significant increases or decreases in the indices measuring change in TSS concentrations, moderately detrimental for moderately significant increases or decreases in these indices and of more limited detriment for changes that are considered to be discernable but not large enough to be of moderate to high significance.

As for the other metrics used in this study (e.g. low salinities and higher dissolved reactive silicates), the significance ratings of the changes in TSS concentrations, normalised to reference conditions (see Table A21 to A24), indicate that not possible to discern between present state and proposed Sc 2 to 5 (with respect to the extent of expected changes in pelagic habitats). However Sc 6 and 7, at a rating of -3, compared to the rating of -2 for Sc 2 to 5 (when assessed against reference condition conditions – see Table A25), are significantly worse than the other proposed scenarios.

Table A21. Extent of surface and bottom waters with elevated turbidity (>100 mg/ℓand >20 mg/ℓ) for the various high flow/flood scenarios used in the model simulations

Scenario	***Very large flood***	***Large flood***	***Large flood***	***Small flood***	***Small flood***	***High flow pulse***
Total freshwater discharge volume1 (M m³)						
	24706.99	16013.13	13764.61	10137.15	6589.98	2003.44
Area where TSS >100 mg/ℓ in the surface waters[1]						
Median	126.8	77.56	44.48	41.82	32.86	3
80%tile	282.75	153.45	124.48	101.2	90.04	28.27

Scenario	***Very large flood***	***Large flood***	***Large flood***	***Small flood***	***Small flood***	***High flow pulse***
Max	1537.49	691.34	474.84	257.41	230.19	152.48
Area where TSS >100 mg/ℓ in the near bottom waters[1]						
Median	581.8	254.19	159.06	163.99	118.1	8.92
80%tile	1470.53	836.46	650.38	461.56	351.88	124.4
Max	3844.7	1839.39	2010.89	1202.81	963.41	655.36
Area where TSS >20 mg/ℓ in the surface waters[1]						
Median	462.7	258.19	175.53	143.71	117.7	13.87
80%tile	1325.05	746.4	608.33	485.02	424.35	163.54
Max	3110.5	2320.68	2006.72	1138.1	1141.07	761.17
Area where TSS >20 mg/ℓ in the near bottom waters[1]						
Median	1341.8	592.76	471.63	446.49	335.03	85.58
80%tile	2476.2	1457.29	1282.05	879.14	767.52	379.85
Max	5258.82	2360.18	2796.98	1769.47	1555.71	1008.01

1 Annual average over a 66-year period.

Table A22. Extent of surface and bottom waters with elevated turbidity (>100 mg/ℓand >20 mg/ℓ) for the characteristic flood sizes used in the assessment of proposed future scenarios

Scenario	***Very large flood***	***Large flood***	***Small flood***	***High flow pulse***
Total freshwater discharge volume1 (M m³)				
	24706.99	14888.87	8363.57	2003.44
Area where TSS >100 mg/ℓ in the surface waters[1]				
Median	126.8	61.02	37.34	3
80%tile	282.75	138.96	95.62	28.27
Max	1537.49	583.09	243.8	152.48
Area where TSS >100 mg/ℓ in the near bottom waters[1]				
Median	581.8	206.62	141.05	8.92
80%tile	1470.53	743.42	406.72	124.4
Max	3844.7	1925.14	1083.11	655.36
Area where TSS >20 mg/ℓ in the surface waters[1]				
Median	462.7	216.86	130.71	13.87
80%tile	1325.05	677.36	454.69	163.54
Max	3110.5	2163.7	1139.59	761.17
Area where TSS >20 mg/ℓ in the near bottom waters[1]				
Median	1341.8	532.19	390.76	85.58
80%tile	2476.2	1369.67	823.33	379.85

Max	5258.82	2578.58	1662.59	1008.01

1 Annual average over a 66-year period.

Table A23. Extent of surface and bottom waters with elevated turbidity (>100 mg/ℓand >20 mg/ℓ) for the various high flow/flood scenarios under the proposed future scenarios

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume1 (M m^3)								
	39095.77	15550.16	15414.29	15381.13	17429.28	14407.08	8972.78	8972.78
Area where TSS >100 mg/ℓ in the surface waters[1]								
Median	141.29	56.58	57.5	56.91	54.88	47.68	33.41	33.41
80%tile	464.06	185.77	182.83	182.31	216.35	177.22	107.49	107.49
Max	1705.14	679.62	651.43	658.42	894.04	704.5	368.71	368.71
Area where TSS >100 mg/ℓ in the near bottom waters[1]								
Median	513.77	207.65	211.76	209.02	197.14	173.14	122.84	122.84
80%tile	2055.38	819.14	805.58	805.9	956.29	780.94	466.36	466.36
Max	7171.22	2857.06	2737.48	2757.14	3772.25	2973.05	1593.81	1593.81
Area where TSS >20 mg/ℓ in the surface waters[1]								
Median	513.91	205.77	208.14	206.32	205.54	176.76	120.72	120.72
80%tile	2366.82	946.15	924.69	924.2	1141	924.42	543.39	543.39
Max	7961.13	3157.4	3013.14	3040.26	4241.88	3323.02	1759.26	1759.26
Area where TSS >20 mg/ℓ in the near bottom waters[1]								
Median	1727.45	695.13	690.85	686.33	769.3	641.32	405.51	405.51
80%tile	4782.88	1905.89	1845.48	1851.83	2400.78	1918.02	1079.62	1079.62
Max	10872.01	4340.21	4156.15	4180.74	5748.49	4531.03	2437.33	2437.33

1 Annual average over a 66-year period.

Table A24. Extent of surface and bottom waters with elevated turbidity (>100 mg/ℓ and >20 mg/ℓ) for the various high flow/flood scenarios under the various proposed future scenarios, reported as a percentage of the surface areas observed under reference conditions

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume1 (M m^3)								
	100.00%	39.80%	39.40%	39.30%	44.60%	36.90%	23.00%	23.00%
Area where TSS >100 mg/ℓ in the surface waters[1]								
Median	100.00%	40.00%	40.70%	40.30%	38.80%	33.70%	23.60%	23.60%
80%tile	100.00%	40.00%	39.40%	39.30%	46.60%	38.20%	23.20%	23.20%
Max	100.00%	39.90%	38.20%	38.60%	52.40%	41.30%	21.60%	21.60%
Area where TSS >100 mg/ℓ in the near bottom waters[1]								
Median	100.00%	40.40%	41.20%	40.70%	38.40%	33.70%	23.90%	23.90%

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
80%tile	100.00%	39.90%	39.20%	39.20%	46.50%	38.00%	22.70%	22.70%
Max	100.00%	39.80%	38.20%	38.40%	52.60%	41.50%	22.20%	22.20%
Area where TSS >20 mg/ℓ in the surface waters[1]								
Median	100.00%	40.00%	40.50%	40.10%	40.00%	34.40%	23.50%	23.50%
80%tile	100.00%	40.00%	39.10%	39.00%	48.20%	39.10%	23.00%	23.00%
Max	100.00%	39.70%	37.80%	38.20%	53.30%	41.70%	22.10%	22.10%
Area where TSS >20 mg/ℓ in the near bottom waters[1]								
Median	100.00%	40.20%	40.00%	39.70%	44.50%	37.10%	23.50%	23.50%
80%tile	100.00%	39.80%	38.60%	38.70%	50.20%	40.10%	22.60%	22.60%
Max	100.00%	39.90%	38.20%	38.50%	52.90%	41.70%	22.40%	22.40%

1 Annual average over a 66-year period.

Table A25. Extent of surface and bottom waters with elevated turbidity (>100 mg/ℓand >20 mg/ℓ) for the various high flow/flood scenarios under the various proposed future scenarios, reported as a significance rating relative to reference conditions

Scenario	***Reference***	***Present***	***Sc 2***	***Sc 3***	***Sc 4***	***Sc 5***	***Sc 6***	***Sc 7***
Total freshwater discharge volume1 (M m^3)								
	0	-2	-2	-2	-2	-2	-3	-3
Area where TSS >100 mg/ℓ in the surface waters[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-1	-2	-3	-3
Area where TSS >100 mg/ℓ in the near bottom waters[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-1	-2	-3	-3
Area where TSS >20 mg/ℓ in the surface waters[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-2	-2	-3	-3
Max	0	-2	-2	-2	-1	-2	-3	-3
Area where TSS >20 mg/ℓ in the near bottom waters[1]								
Median	0	-2	-2	-2	-2	-2	-3	-3
80%tile	0	-2	-2	-2	-1	-2	-3	-3
Max	0	-2	-2	-2	-1	-2	-3	-3

1 Annual average over a 66-year period.