BCLME

Benguela Current Large Marine Ecosystem Programme

FINAL REPORT

Project: BEHP/CEA/03/04:

Assessment of Cumulative Impacts of Scouring of Sub-Tidal Areas and Kelp Cutting by Diamond Divers in Near-Shore Areas of the BCLME Region

Prepared for

Benguela Current Large Marine Ecosystem Programme

February 2007

By

Pisces Environmental Services (Pty) Ltd

Contact Details:
Dr A. Pulfrich
P O Box 31228, Tokai 7966
South Africa
Tel: +27 21-7154238
Fax: +27 21-7122578
Email: apulfrich@pisces.co.za

This report was compiled by
Andrea Pulfrich

This report should be cited as:

PISCES ENVIRONMENTAL SERVICES, 2007. *Project BEHP/CEA/03/04: Assessment of cumulative impacts of scouring of sub-tidal areas and kelp cutting by diamond divers in near-shore areas of the BCLME region.* Final Report to the BCLME mining and petroleum activities task group.

EXECUTIVE SUMMARY

Project BEHP/CEA/03/04 consists of two major components; the gathering of quantitative data, and a comprehensive review of existing information, on the effects of kelp cutting by diamond diver operations and the discharge of sediments by shore-based processing plants on the near-shore environment. During the initial field survey conducted in April 2005, a manipulated kelp cutting experiment was set up in three demonstration areas near Lüderitz, namely Atlas Bay, Wolf Bay and South Jetty. The latter site is impacted by fines discharged into Elizabeth Bay from the nearby mine. The Atlas Bay and Wolf Bay sites served as reference sites. Data collected included rock lobster abundances, kelp abundance and biomass and cover of benthic organisms. In addition, puerulus collectors were installed to monitor rock lobster puerulus settlement. At each site all the kelp was subsequently cut in two experimental lanes. During follow-up surveys conducted in September 2005, April and May 2006 and October 2006, rock lobster abundances, seabed community cover and kelp re-growth were assessed in both cut lanes and adjacent uncut control areas, to determine if the removal of kelp affected these biological parameters. Puerulus settlement was monitored during numerous interim surveys, when collectors were cleaned and serviced.

Results show that:

- The proportion of unconsolidated sediments on the seabed increased at all three sites during the course of the project.
- At the start of the project, sheltered sites had a higher proportion of the kelp *Ecklonia maxima*, whereas exposed sites are dominated by *Laminaria pallida*.
- Benthic communities at the start of the experiment differed between the three study sites, with those at the fines-impacted South Jetty site being characterised by green foliose algae and high cover of filter-feeders.
- The removal of kelp resulted in rapid bleaching of encrusting coralline algae and a proliferation of foliose red and green algae.
- Kelp density increased dramatically after cutting at all but the fines-impacted South Jetty site, with the recovering kelp community being dominated by *Laminaria*, even in sheltered localities.
- Recovery of benthic communities appeared to be rapid, although differences in recovery rate between sites were observed. Recovery to functional similarity occurred within 8-12 months.

- At the start of the experiment, rock lobster abundances were significantly higher at the fines-impacted site. Removal of kelp, however, resulted in a steady decline in lobster densities at this site.
- At the other two sites, lobster abundances were significantly higher in areas where kelp had been removed.
- At all localities there was a slight skewing of sex ratios towards males, and over 90% of the lobsters were below the minimum legal size limit of 65 mm carapace length.
- Puerulus collectors constructed of galvanised steel corroded badly over short periods in the water and were all replaced with stainless steel collectors during the course of the project.
- Collectors at the fines-impacted South Jetty site caught significantly more pueruli than the other sites implying that the presence of mining-related sediments does not negatively influence puerulus settlement.
- Settlement tended to be higher on collectors in areas where kelp had not been removed.

The results of the experiment are subsequently discussed in the light of information from the literature on kelp removal, concomitant community responses, the effects of seabed topography and sedimentation on the distribution of kelp, rock lobsters and benthic communities, and the influence of large-scale natural events. It was concluded that, when seen in context relative to the wide distribution range of the two kelp species in question and the effects of large-scale natural disturbances characteristic of the Benguela ecosystem, the cutting of kelp and discharge of sediments by diamond divers is insignificant, even on a cumulative scale. Of far greater concern is the continual sediment inputs through seawall erosion from coastal mining operations and increasing discharges of sediments from land-based processing plants in southern Namibia.

Management recommendations covering kelp-cutting, seawall construction and mining discharges are provided and marine monitoring programmes, particularly large-scale and long-term monitoring of sediment deposition in the coastal environment, are proposed.

Information on capacity building and training, and ministry and industry co-operation for this project, are provided in appendices, as is a financial report, giving a detailed breakdown of costs incurred to date.

CONTENTS

List of Acronyms, Symbols and Abbreviations

ALS	Airborne Lazer Survey
ANOVA	Analysis of Variance
ANOSIM	Analysis of Similarity
BCLME	Benguela Current Large Marine Ecosystem
BEHP/CEA	Biodiversity, Ecosystem Health and Pollution/Cumulative Effects Assessment
CSIR	Council for Scientific & Industrial Research
CL	Carapace Length
cm	centimetres
df	degrees of freedom
g	grams
ha	hectare
HABs	Harmful Algal Blooms
hr	hour
kg	kilograms
km	kilometres
m	metres
m^2	square metres
m^3	cubic metres
mg/ℓ	milligrams per litre
mm	millimetres
MDS	Multi-dimensional Scaling
MFMR	Ministry of Fisheries and Marine Resources
SCUBA	Self-Contained Underwater Breathing Apparatus
SW	South West
%	percentage
‰	parts per thousand
µm	microns
<	less than
>	greater than
UCT	University of Cape Town
UNam	University of Namibia
UWC	University of the Western Cape

List of Figures

List of Tables

List of Plates

1. INTRODUCTION

1.1 Background

Marine diamonds have been extracted from coastal deposits within the BCLME region since early in the 1960s. In nearshore waters, diamonds are distributed in gullies and potholes, having accumulated in gravel beds near the bedrock through wave sorting. Currently, no marine diamond-mining activities take place off the coast of Angola, but alluvial diamonds have been recovered from Angolan river systems, and interest has been expressed in the possibility of marine mining of alluvial diamond deposits along the Angolan coast.

In Namibia and South Africa, shore concession areas under the jurisdiction of the large diamond companies are leased to smaller contractors who mine the ore bodies by diver-operated suction pumps, operating directly from the shore down to depths of 10 m in small bays. A shore-based operation typically consists of 2-3 divers, their land-based assistants, and a tractor modified to drive a rotary classifier and centripetal pump to which a 20-cm diameter suction hose is attached. As the contractors locate this heavy equipment as close to the sea as possible, access to the water is sometimes achieved by blasting or mechanical damage in the supratidal and intertidal regions. The divers, operating on surface-supplied diving equipment, guide the terminal end of the hose into the gravel deposits, which are sucked up and delivered to the classifier on the shore. To reach the deeper deposits, boulders of all sizes usually need to be broken up and overturned, and these may be relocated by divers into rock-piles, or dragged from gullies at low tide by tractor and chains, and deposited at higher tidal levels. Rocks are also uncovered through removal of surrounding gravel. Relocation and removal of rocks during mining operations has a destructive impact on the associated biota and alters the physical characteristics of the gullies. To reach the diamondiferous deposits in the surf zone and beyond, kelp is cut to allow easy movement of the suction hoses and airlines.

The diamond concentrate is separated from the gravel by the classifier, bagged and transported to central sorting houses. The wastes that remain comprise both oversize gravel and pebbles (“tailings”), which are allowed to accumulate around the screening units, and smaller particles (“fines”), which are returned to the sea as a slurry across the intertidal zone. This waste material has smothering and scouring effects. In addition, mechanical disturbance is caused by trampling and abrasion due to the dragging of suction hoses and the movement of machinery.

Insufficient knowledge exists with respect the cumulative impacts of these nearshore diamond mining activities on the BCLME, but it is speculated that coastal and shallow-water diamond mining operations may affect near-shore ecosystems and fisheries resources in a number of ways, including :

- Scouring and/or smothering of near-shore reef habitats as a consequence of sediment input and/or mobilization resulting in alteration (or loss) of the associated benthic communities.
- Alteration of intertidal and subtidal habitats as a result of disturbance by the diver-operated mining tool and other activities associated with the mining operation.
- Changes in kelp bed community structure as a result of repetitive cutting of kelp to access mining target sites.

Although allegations about the effects of diamond mining on marine habitats remain unsubstantiated, concerns about the potential cumulative impact of habitat disturbance and benthic community changes, which could directly or indirectly affect the rock-lobster resource, have resulted in substantial and ongoing conflicts between the rock-lobster industry, fisheries management authorities, and mining sectors.

More specifically, several potential negative effects on rock lobsters resulting from mining activities have been identified in the past:

- Direct mortality of lobsters through capture by the suction hose or crushing by de-stabilised boulder piles.
- Reduction of food availability through direct removal, scouring or smothering of reef communities or depletion of oxygen.
- Degradation of rock lobster habitat through smothering of reefs.
- Reduction in recruitment success through reduction in kelp cover.

Numerous past impact assessment studies of shore- and vessel-based mining operations along both the Namaqualand and Namibian coast, have found most of these impacts to be localised and short-term, with natural recovery to the pre-mining state occurring within three years. What past studies have not been able to determine is what the cumulative effects might be of a range of mining activities, or of repetitive impacts. One of the issues of concern relates to the impact of sediment deposition and remobilisation from continual or expanding mining operations on near-shore reefs, any resultant degradation in kelp cover, and the effect of repeated cutting of kelp by divers in shore-based operations.

Whilst the biota inhabiting the BCLME region is relatively robust and well adapted to the substantial natural variability in the system, any impact which substantially reduces kelp cover over the long-term would be expected to result in substantial changes in benthic community structure, and potentially reduce options for settlement of rock lobster recruits. This may have important implications for both the abundance and distribution of adult and juvenile rock lobsters, as well as the recruitment success of the stocks. Some understanding is thus needed of the relative impacts and contributions that coastal and shallow water mining has on kelp distribution.

1.2 Objectives and Key Questions

The objectives of the Project BEHP/CEA/03/04 : *Assessment of cumulative impacts of scouring of sub-tidal areas and kelp cutting by diamond divers in near-shore areas of the BCLME region* are:

- ***to review existing information on the cumulative effects of scouring of benthic communities by sediment discharges from diamond mining operations, and gather quantitative data on the effects of kelp cutting by shore-based diver-assisted operations on the near-shore environment, in order to effectively minimise and manage these impacts on marine living resources and the Benguela Ecosystem as a whole.***

The study therefore essentially consists of two major components. The first of these is a comprehensive literature search and review. The second, and more substantial component, involved the gathering of quantitative data on the effects of kelp cutting by diamond diver operations on the near-shore environment. This involved a scientifically robust manipulative kelp-cutting experiment, within demonstration areas near Lüderitz, to obtain a baseline and subsequent time series that examines the effects of kelp cutting.

The monitoring programme addresses the following key questions:

- What are the physical characteristics (topographical features, distribution of seabed types) at selected sites in the study area ?
- What is the abundance and distribution of kelp at these sites ?
- What is the abundance and size distribution of rock lobsters at these selected sites, and how are they affected by increase sedimentation and/or decreasing kelp abundance ?
- What is the composition of the benthic and epi-faunal community assemblages, and how are these assemblages affected by increase sedimentation and/or decreasing kelp abundance ?
- What is the rock lobster recruitment rate at the selected sites, and how is this affected by increase sedimentation and/or decreasing kelp abundance ?

The following key questions from BEHP/CEA/03/03: *Assessment of the Cumulative Effects of Sediment Discharges from On-shore and Near-shore Diamond Mining Activities* have also been included :

- How does the potential loss of kelp bed habitat [by long-term deposition of sediments] affect the abundance and distribution of rock lobsters, and the recruitment success of the stocks ?
- How does the periodic and/or long-term deposition of sediment on reefs affect the density and extent of kelp beds, and how does this compare with the loss of kelp bed habitat as a result of kelp cutting by shore-based divers ?

This report presents the results of the kelp cutting experiment and puerulus collectors, with information gathered during the literature review being incorporated in the final discussion.

2. APPROACH AND METHODOLOGY

2.1 Study Areas

The study focuses on a number of small, protected bays located in the vicinity of Elizabeth Bay near Lüderitz, in Namibia. The selected study sites are Elizabeth Bay, Atlas Bay and Wolf Bay (Figure 1, and Plates 1-3), all of which are characterised by shallow kelp beds, with populations of West Coast rock lobster *Jasus lalandii* forming a major part of the subtidal reef community.

The major difference between the sites, is that the South Jetty site, located within Elizabeth Bay, has been impacted by the long-term accumulation of fine sediments discharged into the bay from the Elizabeth Bay Mine (Pulfrich *et al.* 2003b; Pulfrich 2006). The Atlas Bay and Wolf Bay sites are comparatively undisturbed by mining-related sediments, although they have been impacted on a local scale by shore-based diving contractors at various times in the past (Pulfrich *et al.* 2003a). For the purposes of this study, these locations have the following characteristics in common:

- They are relatively sheltered from the predominant SW swells.
- The kelp beds were extensive enough to allow for the clearing of two 5 m-wide experimental lanes, separated by an at least 10 m-wide buffer zone.
- Vehicular access to within a few metres of the upper intertidal zone can be easily gained on existing tracks. This was particularly important during the setting-up survey as machinery and equipment had to be located as close to the shore as possible.

The mine itself is situated ~1 km inland in the western corner of Elizabeth Bay, and has been in production since 1991. The fines (<1.4 mm) from the processing operations have traditionally been deposited as a sediment-slurry through a pipeline directly onto the Elizabeth Bay beach at an average rate of ~1.3 million tons/year. In 2005 the mining plant was upgraded, and as a result the amount of tailings discharged into the bay has increased to some 2.9 million tons/year. The discharged slurry carries a variable load of suspended sediment with median grain sizes ranging from 150-900 μm, of which between 0.16-5.7% has a grain size finer than 63 μm. The finer components are dispersed offshore in turbid water plumes, whilst the coarser fractions tend to remain on the beach, resulting in rapid accretion at the discharge point. Although the suspended sediment plume remains largely confined to the bay, under certain weather and sea conditions it can move westwards around Elizabeth Point (CSIR 2001a, 2002).

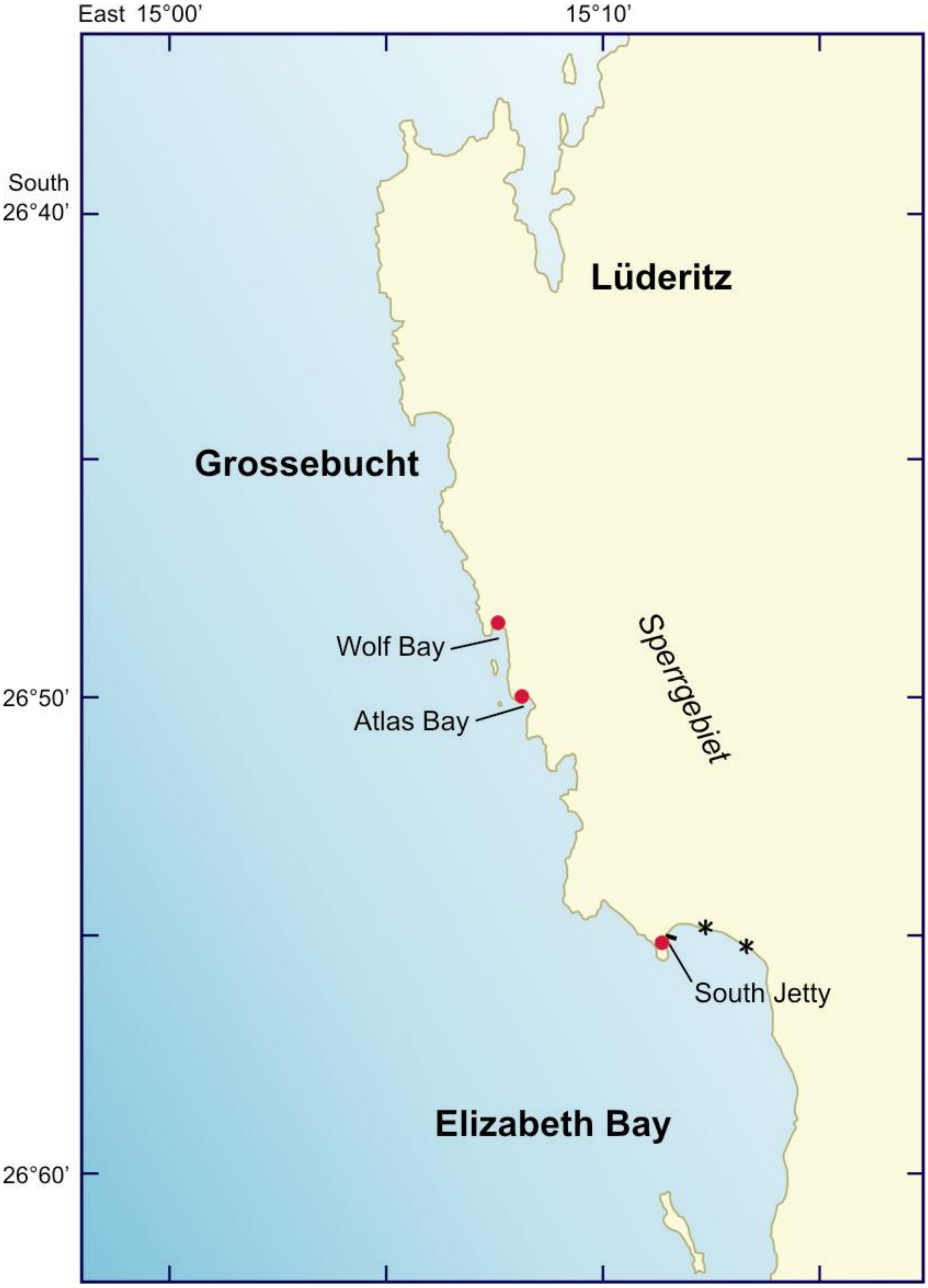


Figure 1. Map of the Elizabeth Bay study area showing the positions of the survey sites. The approximate positions of the Elizabeth Bay Mine fines deposits outlets are indicated by asterisks.

Plate 1. View of the South Jetty study site with the sea-water intake jetty and Elizabeth Bay Mine in the background.

Plate 2. View of Atlas Bay showing the extent of the kelp bed. The access track to the study site ends in the bottom left hand corner of the photograph.

Plate 3. View of Wolf Bay. The access track to the study site ends at the bottom right of the photograph. North Long Island (which supports seals) can be seen in the right of the picture.

2.2 Survey Approach

At each of the study sites, two 5-m wide, approximately 60-m long lanes were cleared, perpendicular to the shore, through the kelp bed (Figure 2). The lanes were cleared by cutting the stipes just above the holdfast. Holdfasts were purposefully not removed. The cleared lanes were separated by a 10-m "buffer zone" which remained uncut and served as the control. In each of the cut lanes, two Booth crevice puerulus collectors (see Plate 7, page 18) were secured to the reef, and a further two were installed in the uncut control lane. The South Jetty site served as the "Sediment Impact" treatment, while Atlas Bay and Wolf Bay served as "Reference" sites, not influenced by discharge of fine sediments.

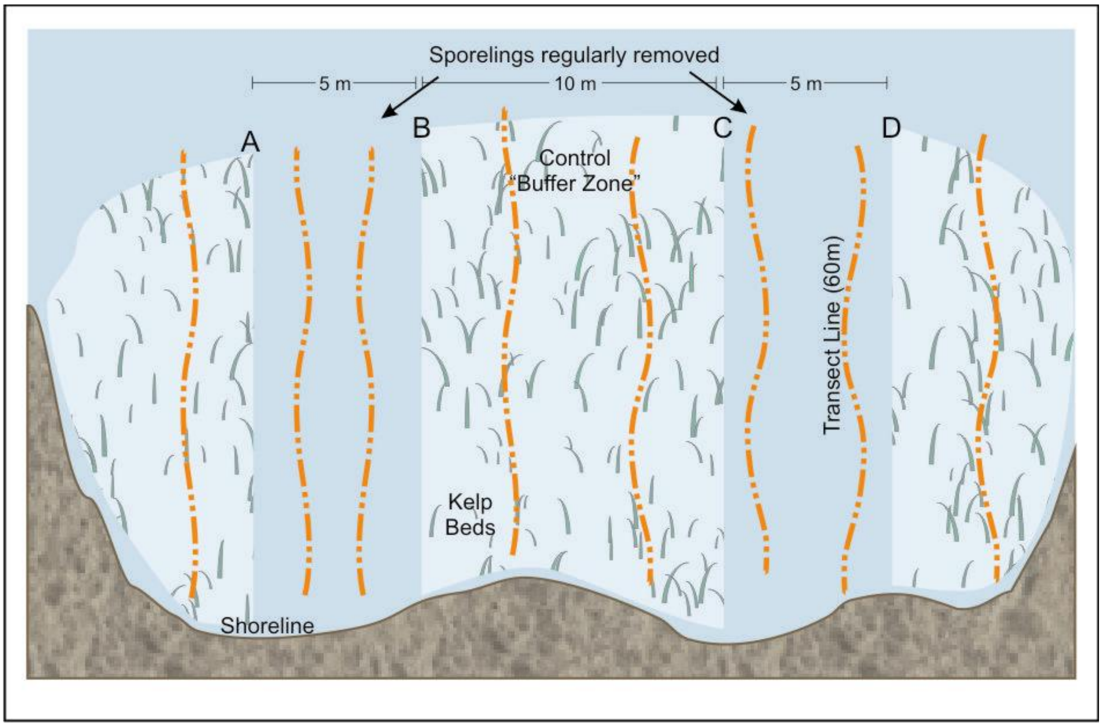


Figure 2. Diagrammatic illustration of the kelp cutting experimental design. The orange dotted-dashed lines represent the diver-transects.

The experiment was set up at each of the study sites during the initial survey between 18-30 April 2005. During this survey the necessary pre-experimental baseline data were also collected. The experiment was set up as follows:

i) The widths of the experimental lanes and buffer zone were measured out at low tide, and the on-shore positions marked with stainless steel eye-bolts (A_1, B_1, C_1 and D_1 in Figures 3-5). The bolts were positioned just above the low spring tide level and buoyed.

ii) A bearing was taken from the high-shore directly into the oncoming swell, and the outer boundaries of the site were similarly marked with bolts (A_0 and D_0 in Figures 3-5).

iii) From the on-shore eyebolt D_1, a diver swam out a 60 m line along the determined bearing. Once at the end of the line, he was guided onto the exact position by a team member who lined up the onshore markers. The off-shore position was buoyed and the widths of the experimental lanes measured out with an underwater reel. These positions were subsequently marked with eyebolts and buoyed (A_2, B_2, C_2 and D_2 in Figures 3-5).

iv) A line was fastened between each of the on-shore and off-shore eyebolts (A_1-A_2, B_1-B_2 etc) to mark the boundaries of the lanes.

v) Within each lane, a diver swam a 1-m wide underwater transect recording the number of lobsters per 10-m section of transect.

vi) A further transect was swum along a line tensioned diagonally across each lane. At 10 m intervals along the diagonal transect, the percentage cover of benthic organisms within a 1-m^2 quadrat was recorded. [Note: due to the frequent poor underwater visibility at most study sites, the 2-m diameter point counts as initially proposed had to be substituted by the 1-m^2 quadrats.]

vii) At 10-m intervals along lines A_1-A_2, B_1-B_2 and D_1-D_2 all the kelp plants within a 1-m^2 quadrat were cut and swum back to shore. For each plant, the species (*Ecklonia maxima* or *Laminaria pallida*), stipe length and plant wet mass (foliose epiphytes removed) were recorded.

viii) During low tide, and/or using snorkel or SCUBA, the kelp was cleared from each of the 5-m wide experimental lanes by cutting the stipes just above the holdfast (Plate 4A). No holdfasts were removed. Although every attempt was made to remove all of the larger kelp plants, juvenile density in the area is extremely high, and it was thus not feasible to clear the lanes completely of kelp.

ix) Once wave action had cleared the lanes of cut kelp (Plates 4B and 4C), the remaining juvenile density was recorded by counting the numbers of plants within a 1-m^2 quadrat placed at 10 m intervals along a transect line.

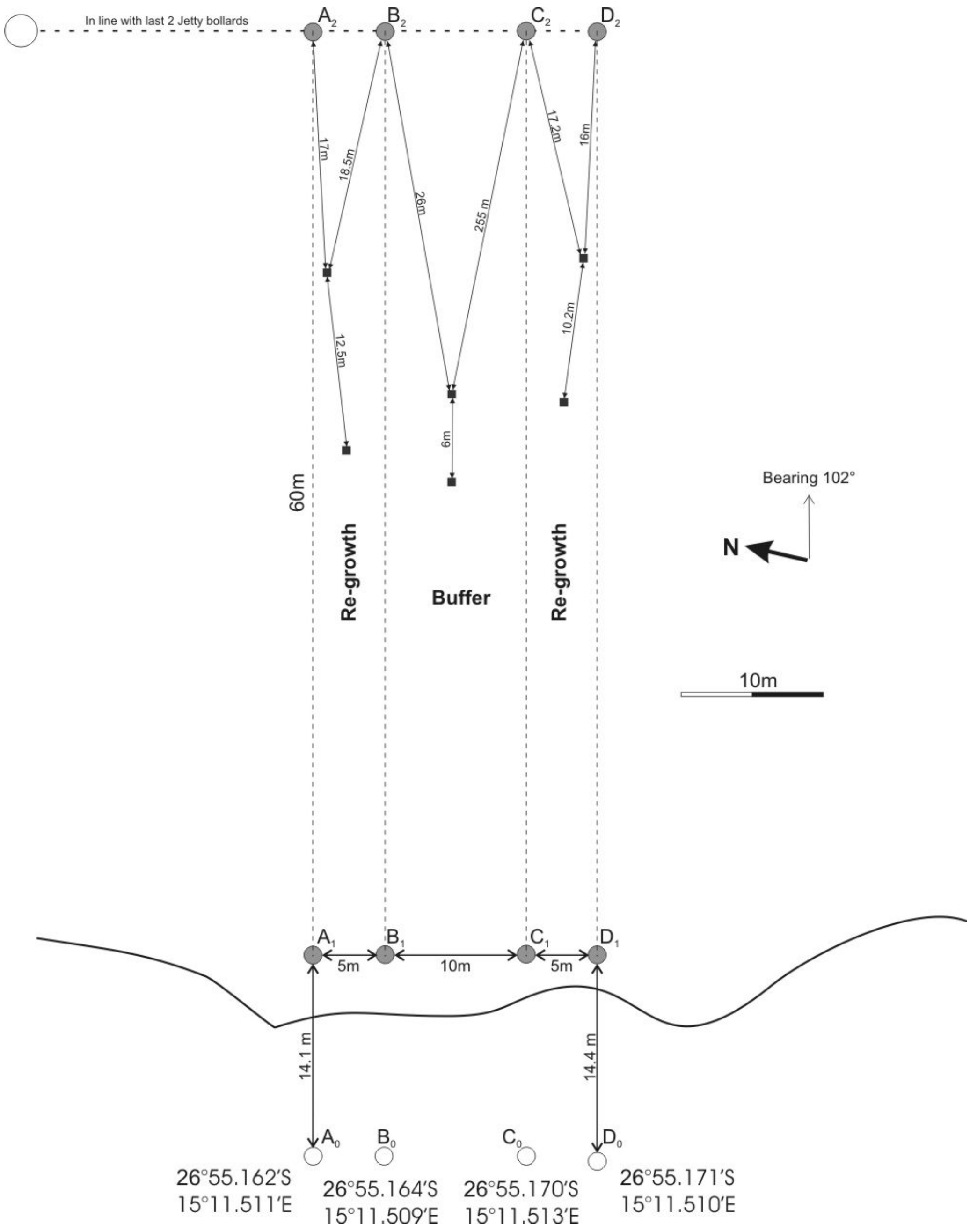


Figure 3. Plan of the experimental layout at the South Jetty site. The approximate positions of the puerulus collectors in each lane are shown as squares. A_0-D_0 = bolts in the mid-shore level; A_1-D_1 = bolts immediately above the low spring tide level; A_2-D_2 = bolts at the seaward ends of the lanes.

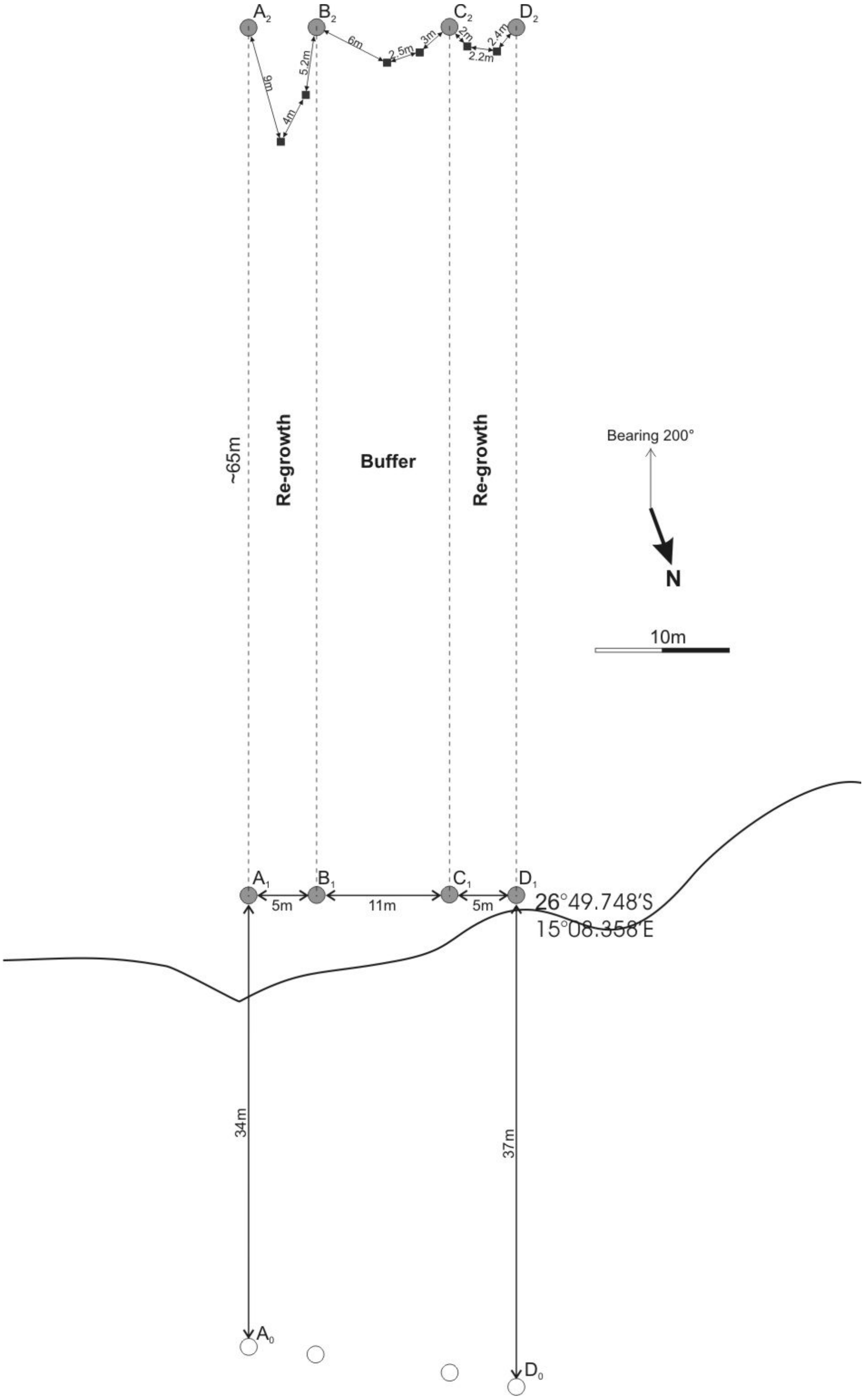


Figure 4. Plan of the experimental layout at the Atlas Bay site. The approximate positions of the puerulus collectors in each lane are shown as squares. A_0-D_0 = bolts in the mid-shore level; A_1-D_1 = bolts immediately above the low spring tide level; A_2-D_2 = bolts at the seaward ends of the lanes.

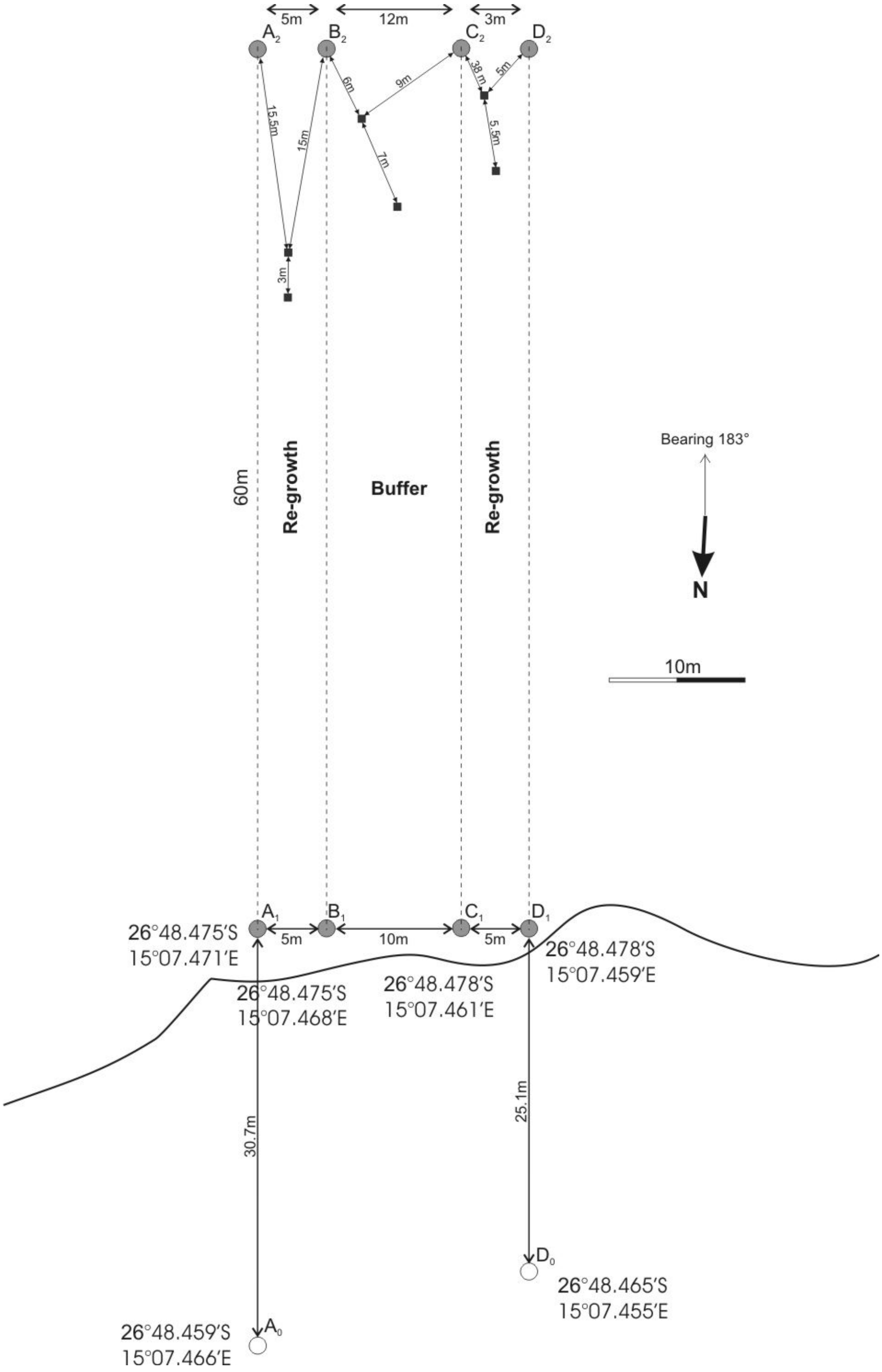


Figure 5. Plan of the experimental layout at the Wolf Bay site. The approximate positions of the puerulus collectors in each lane are shown as squares. A_0-D_0 = bolts in the mid-shore level; A_1-D_1 = bolts immediately above the low spring tide level; A_2-D_2 = bolts at the seaward ends of the lanes.

A.

B.

C.

Plates 4A – 4C. Cutting of kelp in the experimental lanes at Wolf Bay.

x) Having selected and temporarily buoyed suitable locations for the placement of the puerulus collectors, divers secured the collector base-plates to the seabed using stainless steel anchor bolts. The collectors were then swum out andbolted to the bases. Two puerulus collectors were secured in each of the cut lanes, and a further two in the control lane (buffer zone) (Plate 5).

xi) At each site, the off-shore eyebolts and buoys were left in place and the in-shore eyebolts and buoys removed. The inshore bolts were subsequently marked with hazard tape to facilitate relocation on subsequent surveys.

Marking of the lanes with stainless steel bolts, and installation of the base-plates to hold the crevice collectors, was undertaken using a pneumatic drill to drill holes into the reef. This was powered by air fed through an airline from a low pressure compressor located on the shore (Plate 6).

Plate 5. Preparing to install the puerulus collectors at Wolf Bay. The buoys marking the lanes and the cut lanes themselves, are also visible.

Plate 6. Low pressure compressor located on a vehicle on the shore, and airline feeding down to a pneumatic drill used to install collectors and collector base-plates.

2.3 Subtidal Survey Methods

The subtidal benthic community structure at each sampling site was assessed with 1-m^2 quadrats placed every 10 m along a transect line laid diagonally across the experimental lane. For each quadrat, the proportions of rock, gravel and sand were recorded. Dethier *et al*. (1993) have demonstrated that measurement errors using visual estimations are smaller than errors inherent in quantitative (e.g. point - intercept) sampling methods. To maximize the number of quadrats over the relatively large survey areas, quantitative benthic quadrats were therefore not attempted. Instead, the percentage cover of principal benthic community components within each quadrat were estimated and ranked using the Braun-Blanquet scale of coverage categories (Kent & Coker 1992, see Table I). This scale uses smaller categories at lower coverage, ensuring that scarcer species are not outweighed by abundant species in subsequent analyses. Benthic communities are often multi-layered with, for example, a canopy of algae covering mussels and/or coralline algae, or urchins occurring on encrusting coralline algae. Total percentage cover may therefore add up to >100 %.

Table I. Ranking scales used for estimating the percentage cover of benthic organisms.

Benthic Communities Rank	**Braun-Blanquet scale % Coverage**
0	<1%
1	1-5%
2	6-25%
3	26-50%
4	51-75%
5	76-100%

Estimates of rock lobster abundance were obtained by underwater transect counts. During transect counts, the numbers of rock lobsters were counted in a 1-m wide strip along 60-m transect lines laid perpendicular to the shore. The numbers of lobsters in each 5-m section along the transect were recorded separately. Results are expressed as the mean numbers of lobsters per square metre.

At each of the study sites, rock lobsters were caught by a diver during two 15 minute intervals. The rock lobsters were subsequently measured, sexed and returned to the sea.

Once all the biological data had been collected from the experimental lanes, kelp re-growth was cleared from the 5-m wide experimental lanes by cutting the stipes just above the holdfast. This was done using snorkel or SCUBA. During the September 2005 survey, the onset of unfavourable diving conditions precluded the completion of the cutting, and consequently only one lane was re-cut per site. A repeat survey was therefore conducted in January 2006 during which both kelp lanes were recut at all three study sites.

2.4 Puerulus Collectors

The puerulus collectors used for the project, consisted of 12 collectors previously constructed and installed by the Ministry of Fisheries and Marine Resources (MFMR) during a study on rock lobster settlement in Lüderitz Bay, as well as collectors constructed specifically as part of the BCLME project. The MFMR collectors were constructed of galvanised steel, and many of them required substantial maintenance before they could be re-used for this study. These collectors also all require base-plates in order to secure them to the reef. A further 12 new Booth crevice collectors and base-plates were manufactured using only stainless steel components (Plate 7).

After their initial installation, the collectors were left to "cure" over a four-month period. The settlement of pueruli at each of the study sites was subsequently monitored by servicing the collectors at intervals of between six to eight weeks. During this process the puerulus collectors were covered in a custom-made mesh bag to prevent animals from escaping, detached from their bases, and swum back to shore by a diver. The collector was removed from the mesh bag and placed in a large basin. Each collector was photographed to record the degree of algal fouling, and the abundance of various organisms were recorded. The collector was subsequently scraped to remove all macrophytes using the blunt side of a hacksaw blade (Plate 8), and rinsed several times with seawater to retrieve any lobsters and other by-catch species from the crevices of the collector. The collector was then swum back onto site and secured to its base. Missing collectors were replaced where possible, and damaged collectors fully repaired before being returned to the sea.

Plate 7. Booth crevice collector and base-plate made using stainless steel components.

Plate 8. Scraping of Booth crevice collector to remove fouling organisms.

Lobsters were placed in 1-litre sample jar with seawater and transported to the laboratory for morphometric analyses. Pueruli and juvenile lobsters were staged according to the presence of certain morphological features and the degree of pigmentation (Table II and Plate 9). Pueruli are characterised by the presence of feather-like "swimming hairs" on the ventral side of the tail. They have smooth carapaces, are dorso-ventrally flattened, and show different stages of pigmentation depending on their age (Plate 9). In juveniles, the "swimming hairs" are absent, and the laterally flattened, spiny carapace resembles that of an adult lobster. Animals were placed on paper towelling to remove excess water, their carapace lengths measured to the nearest 0.1 mm using vernier callipers, and their wet weight record to the nearest 0.001 g using an electronic balance.

Table II. Staging of rock lobster pueruli and juveniles.

Stage	Morphological characteristics
Pt	Transparent puerulus
Ps	Slightly pigmented puerulus
Pi	Intermediate pigmentation
Pf	Fully pigmented puerulus
J	Juvenile

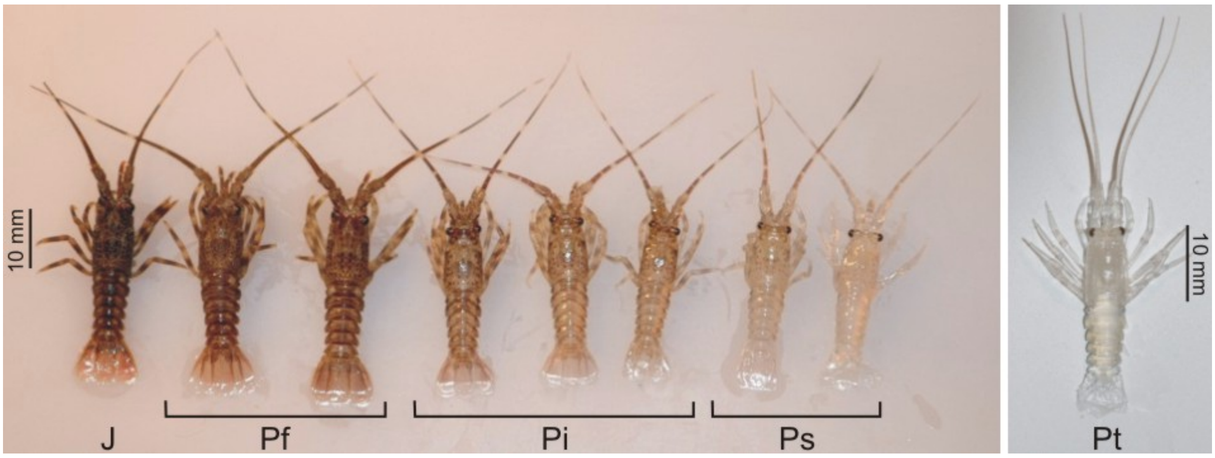


Plate 9. Morphological features and pigmentation stages of *Jasus lalandii* pueruli and juveniles. Pt = transparent puerulus, Ps = slightly pigmented puerulus, Pi = intermediate pigmentation, Pf = fully pigmented puerulus, J = juvenile lobster.

2.5 Data Analysis

Non-parametric multivariate analyses (Clarke 1993) were used to assess patterns of association in the intertidal and subtidal benthic communities, employing the software package PRIMER (Plymouth Routines In Multivariate Ecological Research). These methods are useful for clearly presenting results obtained from the typically large data sets obtained in environmental impact assessments (Clark 1993).

The principle aim of multivariate techniques is to discern the most conspicuous patterns in the community data by objective reduction of the raw data set. Comparisons of samples are based on the extent to which they share particular characteristics (species) at comparable levels of occurrence. The Bray-Curtis similarity coefficients arising from the comparisons are set out in a triangular matrix which forms the basis of subsequent graphical representations of the community patterns (Bray & Curtis 1957; Clarke & Warwick 1994). Patterns in the data are represented graphically through hierarchical clustering and multi-dimensional scaling (MDS) ordination techniques. The former produces a dendrogram in which samples with the greatest similarity are fused into groups, and these successively grouped into clusters as the similarity criteria defining the groups are gradually reduced (Figure 6). MDS techniques compliment hierarchical clustering methods by more accurately 'mapping'

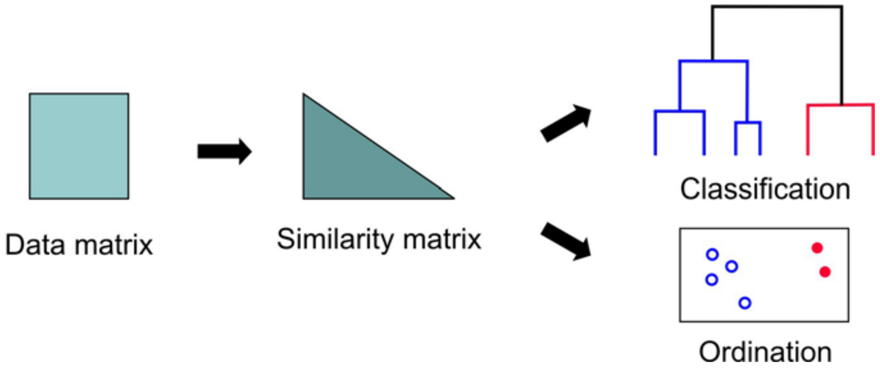


Figure 6. Schematic diagram representing the stages in the multivariate analysis of benthic community data (after Field *et al.* 1982).

the sample groupings two-dimensionally in such a way that the distances between samples represents their relative similarities or dissimilarities.

Data collected using a visual ranking system are essentially already in a ranked form similar to the six or seven point scale resulting from 4th-root transformation. Consequently, the data were not transformed. The non-parametric analysis of similarity (ANOSIM) test was used to test for differences between sites or treatments. The test statistic (R) was used to compute the observed differences between samples from the three sites (or cut and uncut treatments) and compare these with differences among replicate samples within each site/treatment, with significant differences among groups being set at $p<0.05$.

The PRIMER programme SIMPER was used to re-examine the data set in the light of the multivariate results, to determine which taxa were responsible for the patterns observed in the community structure, both within sites, as well as between sites.

Parametric and non-parametric statistical analyses were conducted using the software programme STATISTICA (Version 7, 2006 edition).

3. RESULTS

3.1 Seabed Topography at the Study Sites

The proportion of sand, gravel, and rock recorded in subtidal quadrats in both experimentally cut and uncut areas at the three study sites between April 2005 and October 2006 is illustrated in Figure 7. Figure 8 shows the observed changes in the distribution of sand with depth. From this it is evident that the proportion of sand-dominated seabed within the experimental areas was significantly higher at the South Jetty site than at Atlas Bay and Wolf Bay (Kruskal-Wallis ANOVA; $H_{2,\ 64} = 21.135$, $p<<0.001$). At South Jetty, patches of sand were relatively common between the highly structured and profiled reef from just beyond 1 m depth, and the reef itself was often covered by a layer of silt. In contrast, at Atlas Bay, sand became more prominent beyond 2 m depth, and the relatively flat and featureless reef was seldom covered by silt. At Wolf Bay, sand was not observed in the experimental area, but was known to occur towards the centre of the bay, at depths beyond 4 m. Here too the rocky seabed had little profile and structure, thereby offering few crevices and caves as shelter for mobile invertebrates such as rock lobsters.

Of interest is that the proportion of unconsolidated sediments (i.e. sand and gravel) increased at all three study sites since the start of the study in April 2005, although in no case has this increase been significant (Student's t-tests; South Jetty: t-value=-0.970, p=0.337; Atlas Bay: t-value=-1.860, p=0.069; Wolf Bay: t-value=-1.966, p=0.056). From Figure 8 it can be seen that the increase in the occurrence of sand across the study area varied between the sites. At South Jetty and Wolf Bay, sand appeared to have become more common in the deeper areas, whereas at Atlas Bay, the shallow areas (<1.5 m depth), which were previously dominated by rocky substrate, became increasingly covered by sand.

This can be explained by the heavy rains that fell in the area in May 2006, when 158 mm fell over a period of five days (J. Alexander, Namdeb, pers. comm.). This had a substantial impact on the area resulting in the formation of large pans inland of the coast, some of which persisted until January 2007. As the bedrock along this piece of coastline is close to the surface, the terrestrial run-off during and after the rains was substantial. At Atlas Bay, in particular, this resulted in large volumes of sand being

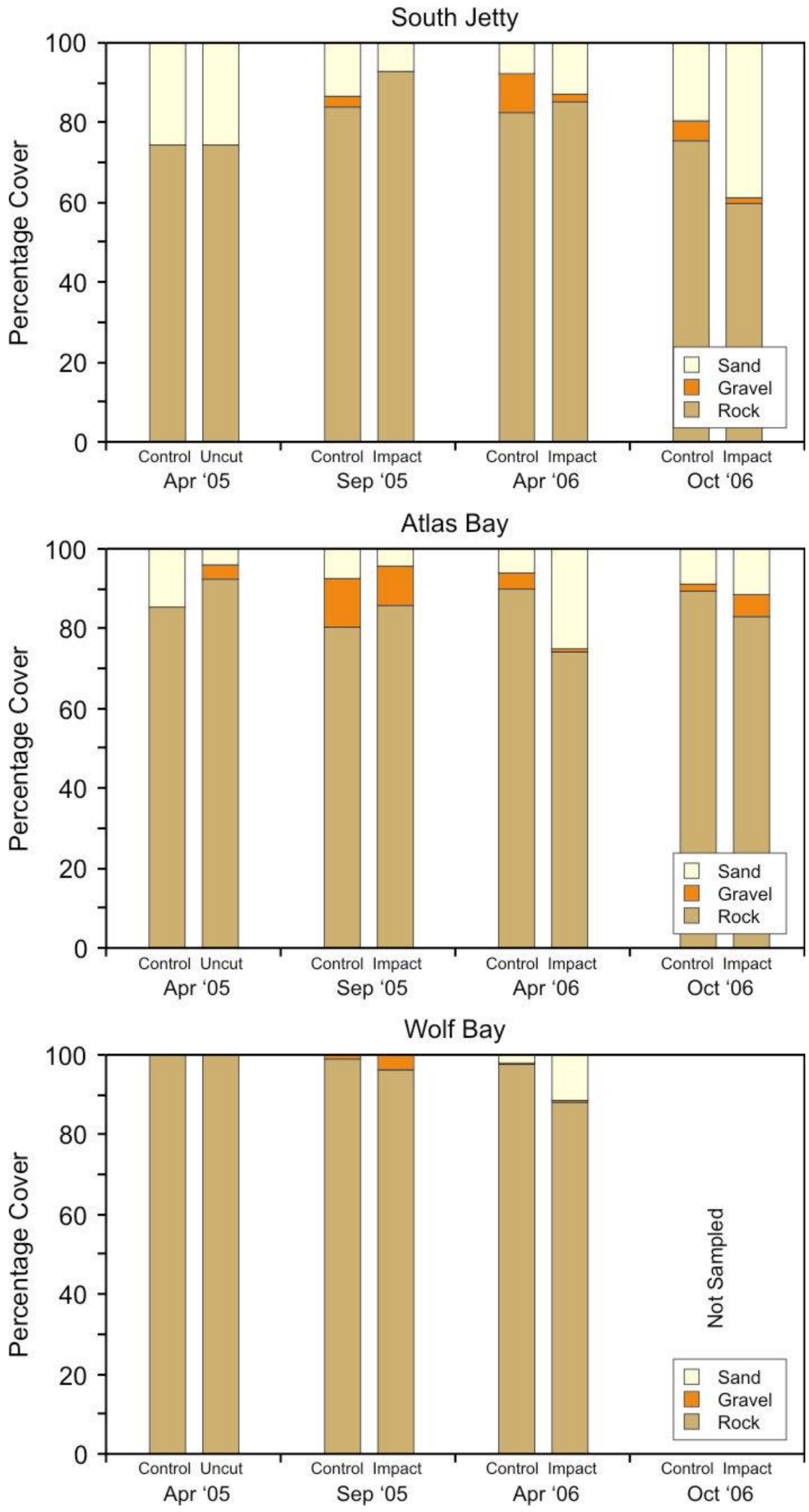


Figure 7. The proportion of sand, gravel, and rock recorded in subtidal quadrats at the three study sites in April (pre-experimental) and September 2005 and April and October 2006 (post-experimental).

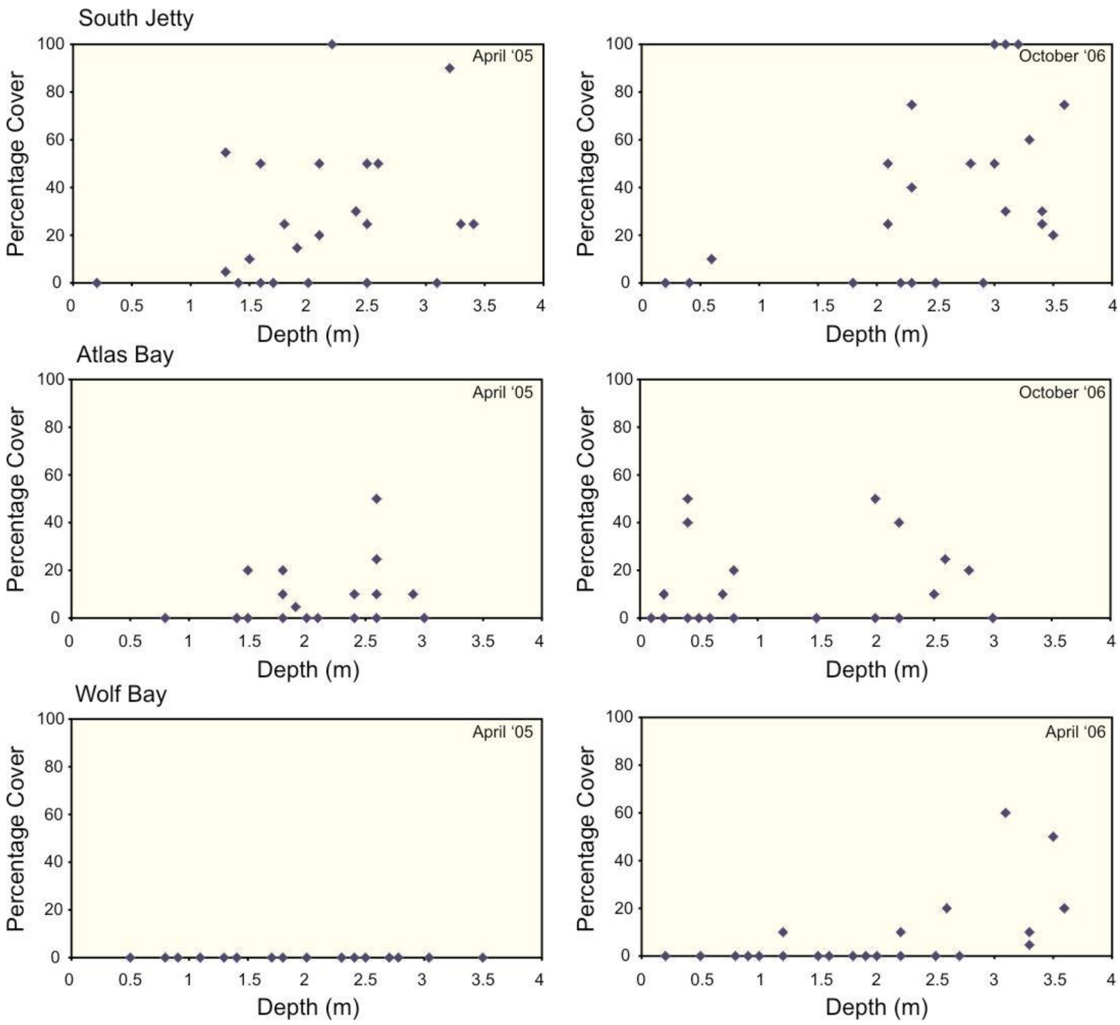


Figure 8. Distribution of sand with depth at the three study sites in April 2005, before kelp was cut, and April/October 2006.

transported into the intertidal and shallow subtidal habitats (Plate 10). In the intertidal, this sand has persisted, smothering considerable portions of the rocky shore. Gradual redistribution of the sand became noticeable during the October 2006 survey, when a proliferation of green foliose algae (*Ulva* spp. and *Cladophora* spp.) was evident in areas that had recently become exposed. These species are all fast-growing, ephemeral opportunists typical of a stressed, grazer-depleted environment.

Plate 10. Substantial terrestrial runoff and inputs of sand into the intertidal and shallow subtidal areas at Atlas Bay following the floods in mid-May 2006.

3.2 Kelp Distribution, Biomass and Abundance

3.2.1 Baseline Conditions

In southern Namibia, the kelps *Laminaria pallida* and *Ecklonia maxima* dominate the algal flora in the shallow subtidal zone. *Ecklonia* is a canopy-forming kelp which in clear water can extend seawards to a depth of about 10 m. The smaller *Laminaria* typically forms a sub-canopy to a height of about 5 m underneath *Ecklonia*, but can continue its seaward extent to about 20 m depth (Velimirov *et al*. 1977; Jarman & Carter 1981).

The mean density of *Laminaria* and *Ecklonia* plants per square metre at the three study sites in April 2005, before cutting of the experimental lanes commenced, is illustrated in Figure 9. ANOVAs indicated that at Wolf Bay the density of *Laminaria* was significantly higher (F=6.217, p=0.003), and the density of *Ecklonia* significantly lower (F=13.544, p=0.0000) than at the South Jetty and Atlas Bay sites. A comparison of total kelp density (i.e. *Laminaria* and *Ecklonia*) could, however, not detect a significant difference between the study sites (F=1.038, p=0.360).

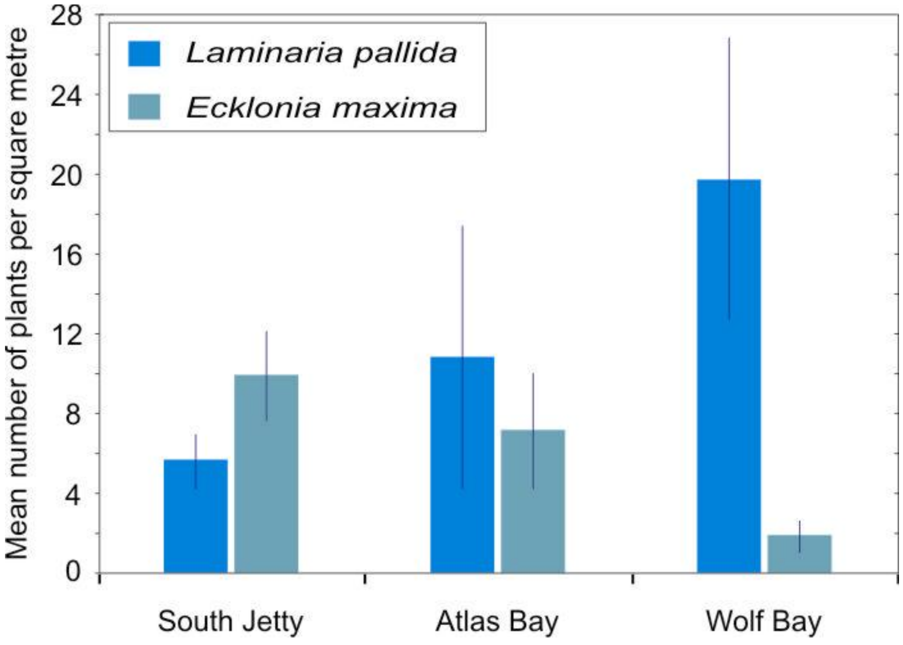


Figure 9. Mean density of the two kelp species *Laminaria pallida* and *Ecklonia maxima* at the three study sites in April 2005 before kelp-cutting began. The standard error of the means are also shown. n = 21 quadrats at each of the three sites.

The percentage contribution of the two kelp species to the overall kelp density is provided in Table III.

Table III. Percentage contribution of the two species of kelp (*Laminaria pallida* and *Ecklonia maxima*) to the overall kelp density and biomass at the three experimental sites at the start of the study in April 2005.

SITE	**Density**		**Biomass**	
	% Laminaria	***% Ecklonia***	***% Laminaria***	***% Ecklonia***
South Jetty	60.4	39.6	18.6	81.4
Atlas Bay	36.1	63.9	21.7	78.3
Wolf Bay	91.6	8.4	82.2	17.8

The mean total kelp biomass per square metre at the three study sites, and the contribution to this by the two kelp species, is illustrated in Figure 10 and presented in Table III. An ANOVA could not detect a significant difference in total biomass between the sites (F=0.336, p=0.716). The total biomass of kelp ranged from a minimum of 1 kg.m^{-2} to a maximum of 27 kg.m^{-2}.

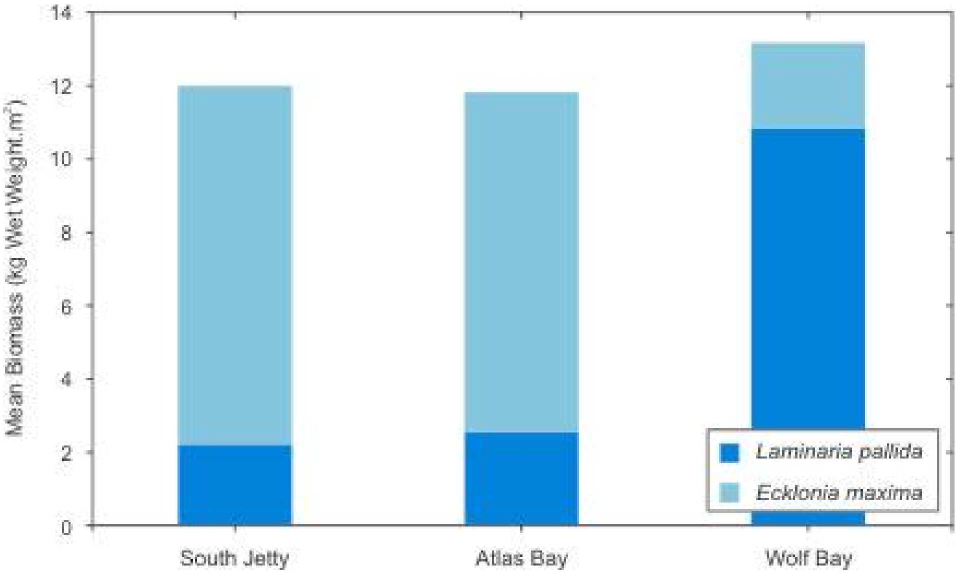


Figure 10. Mean biomass per square metre of the two kelp species *Laminaria pallida* and *Ecklonia maxima* at the three study sites in April 2005 before kelp-cutting began. n = 21 quadrats at each of the three sites.

Figure 11 illustrates the distribution of the two kelp species with depth recorded during the four surveys between April 2005 and October 2006, at the three study sites. Unfortunately, due to the onset of unfavourable diving conditions, the Wolf Bay site could not be sampled during the October 2006 survey. The number of plants per square metre declines with depth for both *Laminaria* and *Ecklonia*. At South Jetty, the offshore extent of the kelp is limited by seabed type, with the cover of sands and silts increasing beyond ~40 m from the low water mark. At Atlas Bay and Wolf Bay, where the seabed is dominated by flat reef, the kelp bed extends a further 10-20 m beyond the end of the experimental lane[1].

Measurements taken of stipe length and wet weight of individual plants (Figure 12) showed that *Ecklonia* displayed a fairly clear stipe-length / wet-weight relationship. This relationship was, however, not so clear for *Laminaria*, as plants with large fronds (and therefore high wet weights) are often characterised by short stipes, particularly in shallower water. The exceptionally high density of kelp (up to 86 plants per square metre) and large fronds of *Laminaria* at the study sites, make underwater work especially difficult, as the kelp forms a virtually impenetrable canopy. Due to the short stipe lengths, it is not possible to work underneath this canopy. It is therefore understandable why shore-based diamond diving operations find it necessary to cut lanes in the kelp to facilitate the movement of suction hoses and airlines and ensure diver safety.

3.2.2 Removal of Kelp

The initial effects of the removal of kelp from the experimental lanes on the benthic communities became noticeable within 2-3 days after the kelp had been cut at the start of the experiment in April 2005. The most obvious effect was the rapid bleaching of crustose coralline algae in the shallow subtidal areas (Plate 10).

The onset of unfavourable diving conditions, and time constraints before expiry of the work visas, precluded the collection of data on kelp density and biomass in the experimental lanes during the September 2005 survey. One lane was however re-cut at each site. The study sites were subsequently revisited in January and February

---

[1] The length of the lanes was limited to 60 m due to the maximum length of airline hose available.

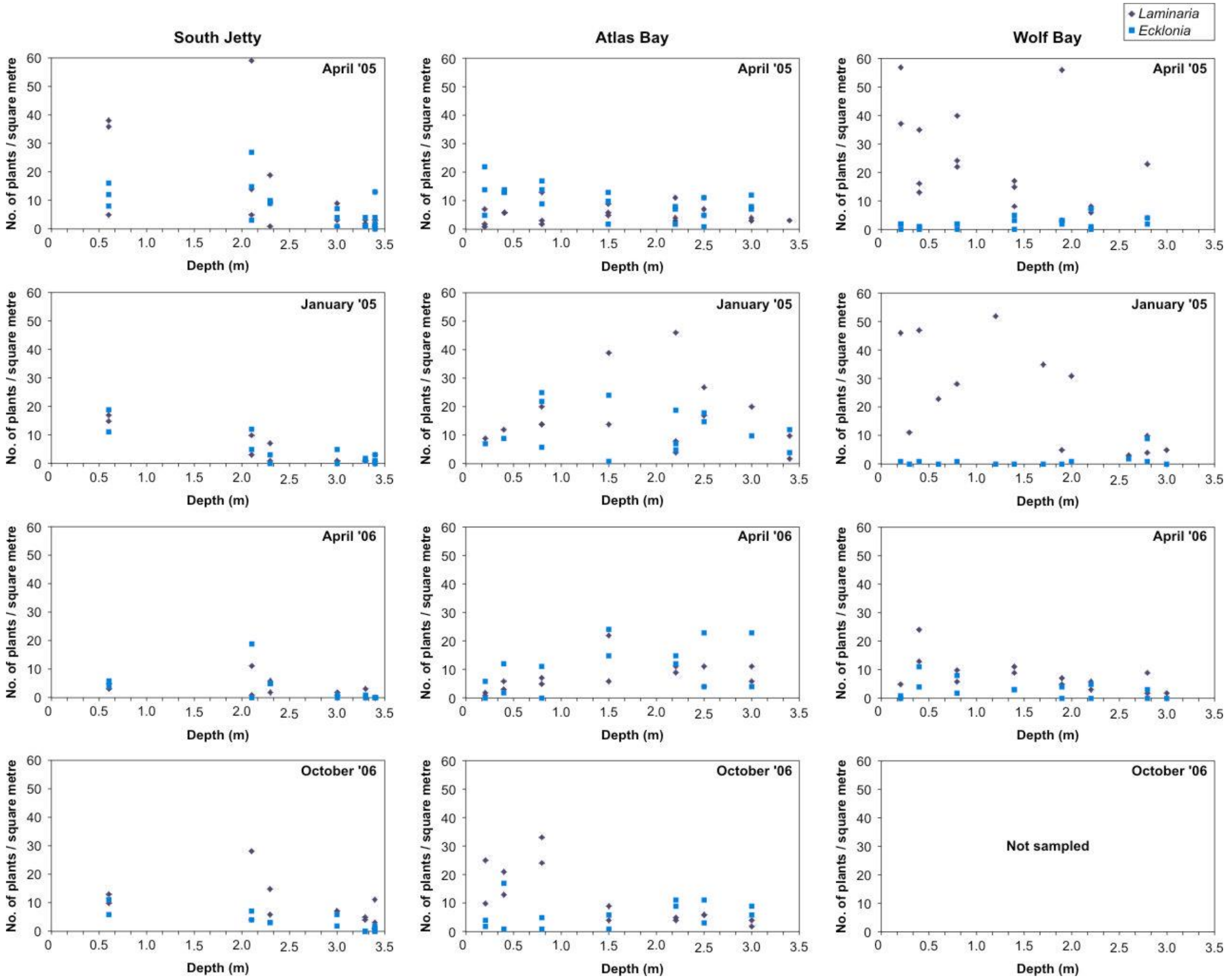


Figure 11. The distribution of the two kelp species *Ecklonia maxima* and *Laminaria pallida* with depth in the cut lanes, during the four surveys between April 2005 and October 2006, at the three study sites.

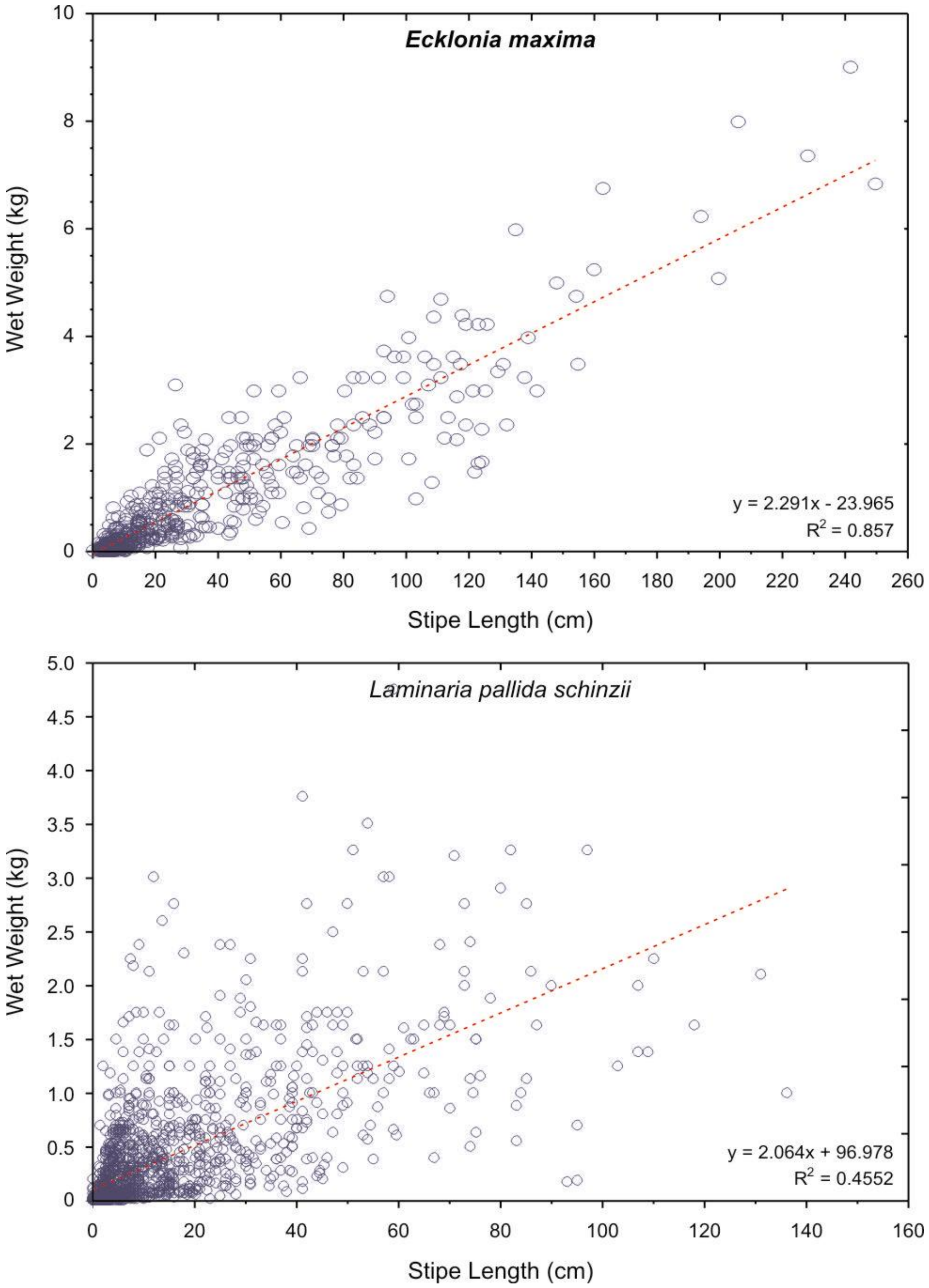


Figure 12. The relationship between stipe length and wet weight for the two kelp species *Ecklonia maxima* and *Laminaria pallida* at the three study sites.

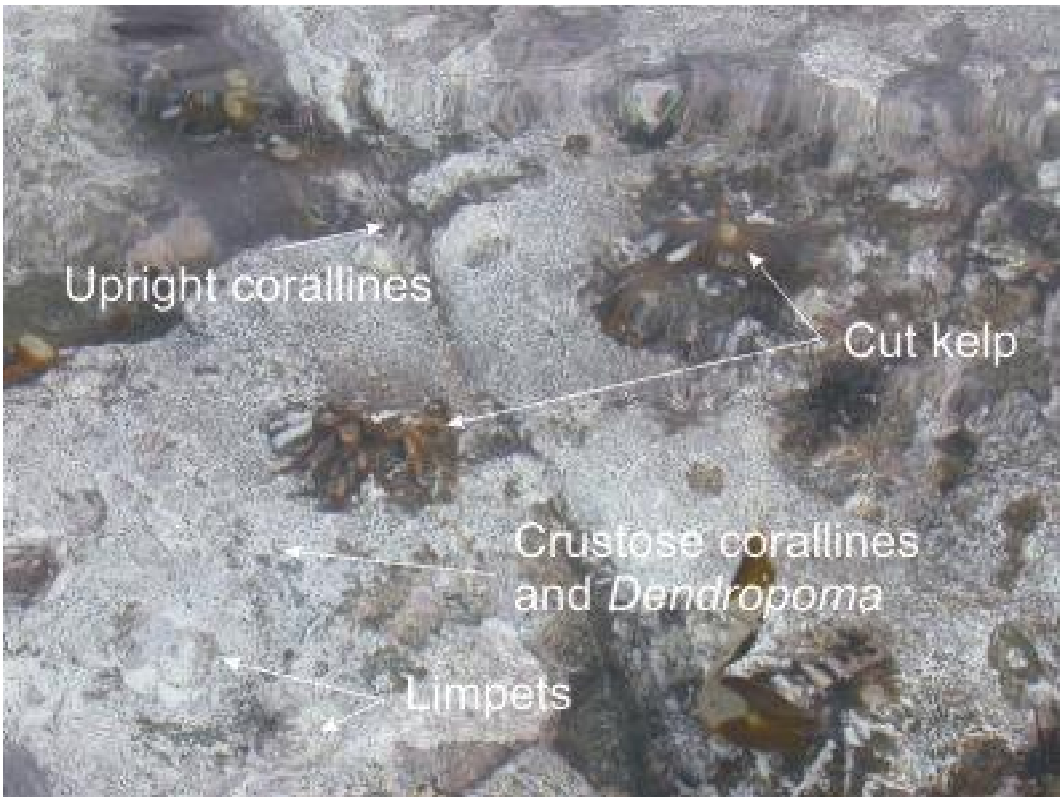


Plate 11. Photograph showing the rapid bleaching of crustose coralline algae a few days after kelp has been cut and removed.

2006 to collect data on density and biomass of kelp re-growth in the experimental lanes, and both experimental lanes were then again clear-cut at each site.

Figures 13 and 14 illustrate the changes in density and biomass of *Laminaria pallida* and *Ecklonia maxima* in the two experimentally cut lanes at the three study sites. Unfortunately, due to the onset of unfavourable diving conditions, the Wolf Bay site could not be sampled during the October 2006 survey. As quadrat samples of kelp in the control areas could not be taken during the course of the survey, without affecting the kelp density in the controls, similar density and biomass data were not collected from uncut areas. Figure 15, however, provides the percentage cover data of kelp at the three study sites over the same period. This provides some indication of the changes in percentage cover in the uncut control areas compared to the cover in the experimental lanes. At Atlas Bay and Wolf Bay, percentage cover of kelp remained relatively stable throughout the study, whereas at South Jetty cover declined in late 2005, recovering again the following year.

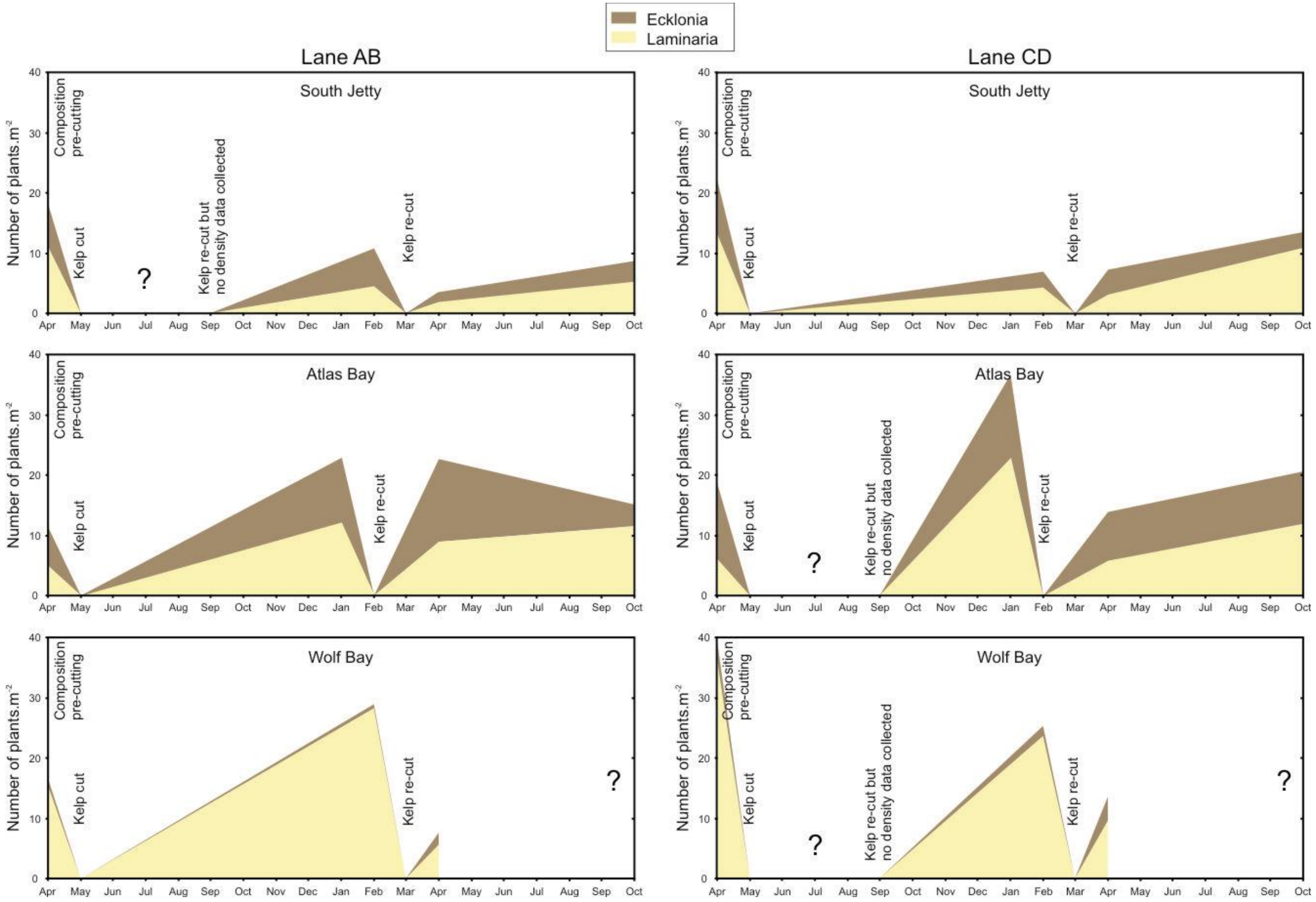


Figure 13. Recovery of kelp abundance (number of plants per square meter) in the two experimentally cut lanes at the three study sites.

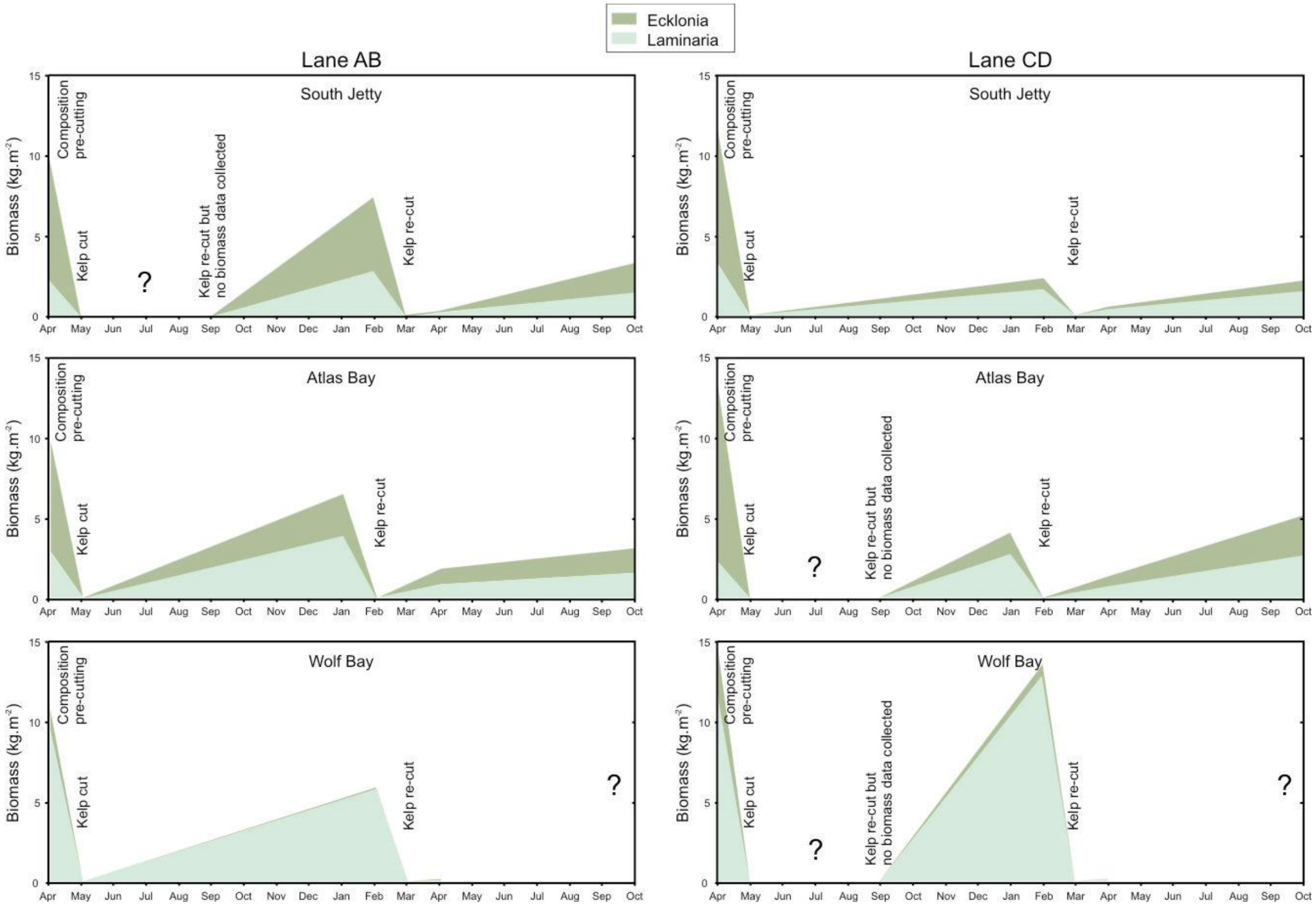


Figure 14. Recovery in the biomass (wet weight in kgs per square meter) of kelp in the two experimentally cut lanes at the three study sites.

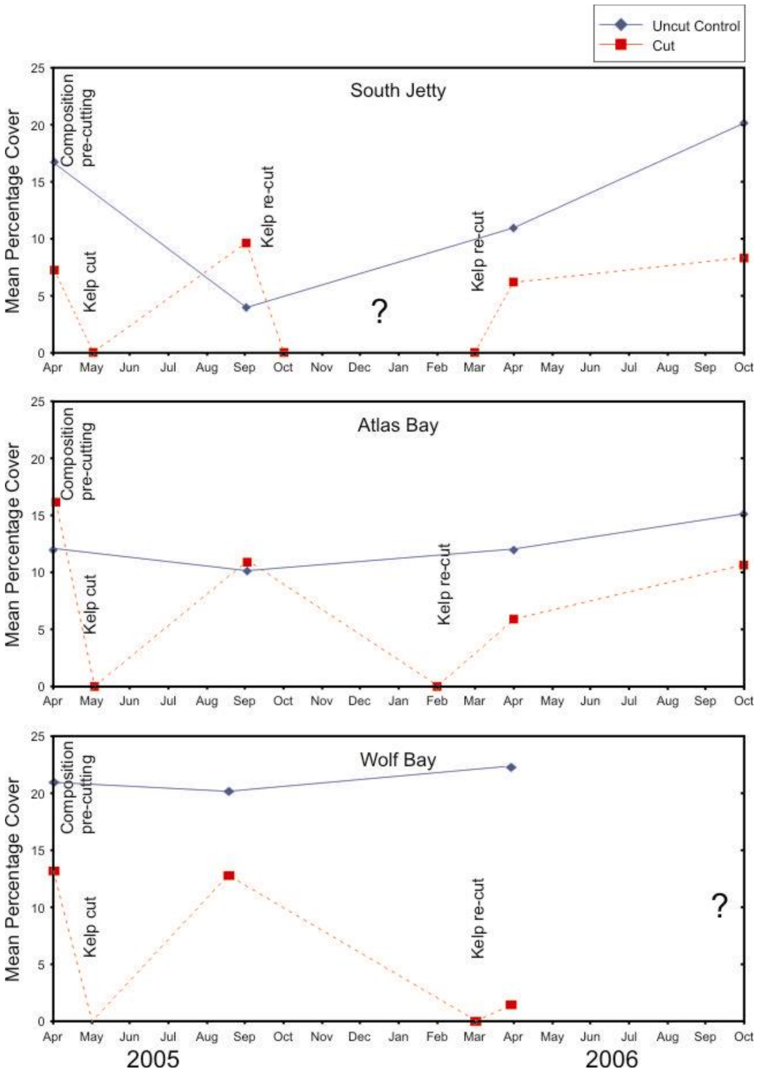


Figure 15. Changes in the percentage cover of kelp in uncut control areas and in experimentally cut lanes at the three study sites.

From Figure 13 it can be seen that at the control sites at Atlas and Wolf Bays kelp density (number of plants per square meter) increased dramatically after kelp clearing as juveniles developed rapidly in the space made available. In most cases, densities in fact exceeded those of the pre-cut situation. Only at the site impacted by fines discharged from the Elizabeth Bay mine, i.e. the South jetty site, did kelp abundance in the cut lanes not reach or exceed previous levels. *Ecklonia* has persisted at the South Jetty and Atlas Bay sites throughout the study, even after cutting of the lanes. Of interest, however, is the change in proportion of the two kelp species over time; at South Jetty where *Laminaria* contributed on average 60% to the kelp density at the start of the study, this had increased to 71% in October 2006. At Atlas Bay *Laminaria* contributed only 36% to the overall kelp density in April 2005, increasing dramatically to 65% eighteen months later.

Biomass (particularly of *Laminaria*) generally appeared to recover to previous levels within four to eight months (Figure 14). With *Ecklonia*, this is expected to take longer as the plants originally cut were large adults with substantial bladders and fronds.

3.3 Benthic Communities

3.3.1 Baseline Conditions

The results of the multivariate analysis performed on the benthic quadrat data collected at the three sampling sites in April 2005, before any kelp was cut in the experimental lanes, is shown in Figure 16. ANOSIM testing showed that the benthic community at the South Jetty site, which is impacted by mining-related sediments, was significantly different from those at Atlas Bay and Wolf Bay (ANOSIM, R=0.344, p=0.001). In contrast, the community structure at Atlas Bay and Wolf Bay did not differ from one another. An overlay of the proportion of sandy seabed in the quadrats on the MDS plot helps explain the observed differences in the community structure at the Jetty site.

The ubiquitous kelp (*Laminaria pallida* and *Ecklonia maxima*), red algae (*Epymenia obtusa* and *Hymenema venosa*) and encrusting corallines provided the majority of the subtidal algal cover at all sites. The taxonomic groups identified by the SIMPER analysis as being consistently abundant throughout the sites were red algae and kelp. Of the taxonomic groups that played a part in determining the dissimilarity between the sites, encrusting coralline algae, upright coralline algae and the reef-building polychaete *Gunnarea capensis* were consistently good discriminating taxa. The coralline algae were characteristic of the Atlas Bay and Wolf Bay sites, whereas the reef worm was recorded only at the Jetty site.

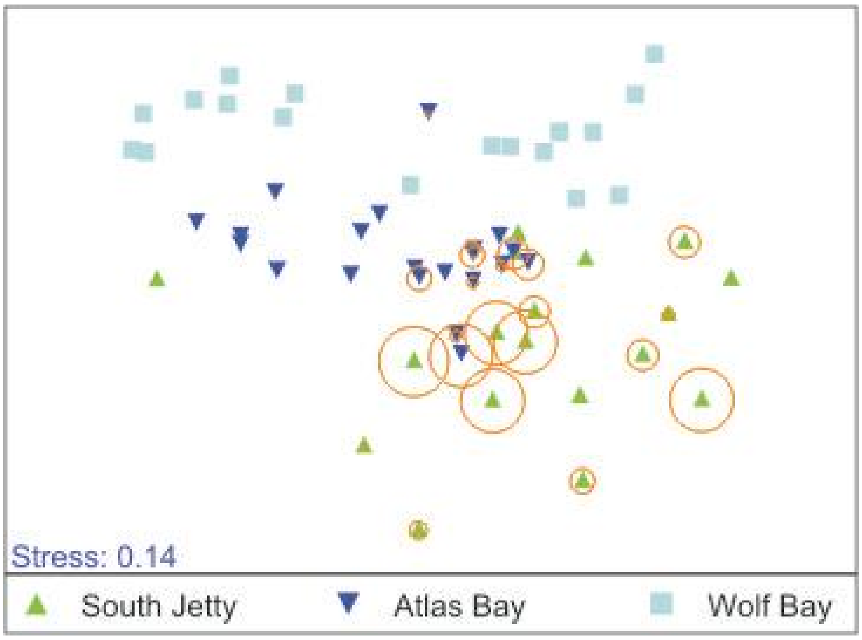


Figure 16. MDS ordination plot illustrating the differences in benthic community structure at the three study sites at the start of the project in April 2005. An overlay of the proportion of sandy seabed (orange circles) recorded in the quadrats is provided (Data range from 0-55% sand).

The data for percentage cover were analysed further at higher taxonomic level by dividing the faunal components into functional groups (Figure 17). At the start of the experiment in April 2005, red foliose algae were the most dominant cover at most sites, commonly forming a dense sub-canopy beneath the kelp. Kelp was likewise common at all the study sites. Green algae were present only at the Jetty site, but provided <1% coverage on average. In contrast, the cover of encrusting algae was significantly lower at South Jetty than at the other two sites (Kruskal-Wallis ANOVA, $H_{2, 64}$=19.413, p=0.0001). Filter-feeders also showed significant differences in percentage cover between sites, with Atlas Bay having significantly lower cover than the Jetty and Wolf Bay (Kruskal-Wallis ANOVA, $H_{2, 64}$=20.333, p<<0.0001). More specifically, however, the percentage coverage of filter-feeders at South Jetty was dominated by the Cape reef worm *Gunnarea capensis*, which was recorded only at this site. A possible explanation for this is the high availability of unconsolidated sediments at South Jetty providing the species with material to build its reefs. In contrast, at Wolf Bay the filter-feeders were dominated by *Dendropoma corallinaceus*, a small colonial species that forms an intertwining mass of worm-like tubes sunken into crustose coralline algae.

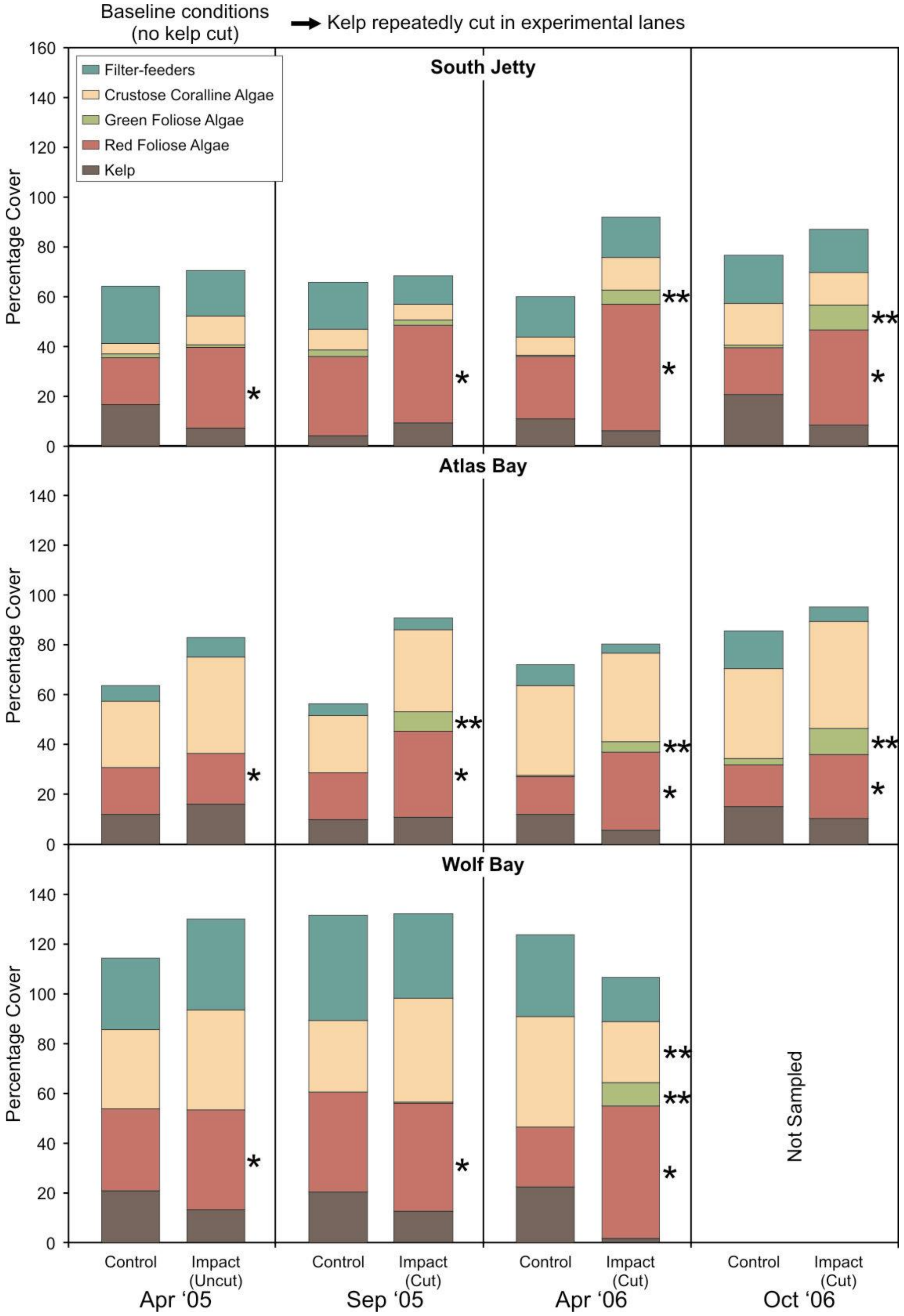


Figure 17. Mean percentage cover of kelp, red foliose algae, green foliose algae, crustose corallines and filter-feeders recorded in cut (impact) and uncut (control) areas at the three study sites during April and September 2005, and April and October 2006. The number of quadrats assessed in each treatment is 14. Asterisks note consistent (*) or significant (**) changes.

The percentage cover of predators and grazers were <1% at all sites and are thus not included in the analysis.

The results of the multivariate analysis performed on the benthic quadrat data collected at the three sampling sites during April 2005 are shown in Figure 18. The three sites were tested separately.

3.3.2. Removal of Kelp

Figure 17 shows that the repetitive removal of kelp in the experimental lanes resulted in an increase in the cover of red and green foliose algae at all three study sites since the start of the experiment. Although the increase in red algae was not significant at any of the sites, the increase in green algae was significant at all three sites. The removal of kelp from the experimental lanes had varying effects between the sites: at South Jetty the percentage cover of kelp after April 2005 has remained relatively consistant, whereas at Atlas Bay and Wolf Bay a decline in the percentage cover of kelp was evident between April 2005 and April 2006, with that at Wolf Bay being significant. With the exception of a significant decrease in the cover of filter-feeders (primarily *Dendropoma*) at Wolf Bay, with a concomitant decrease in crustose coralline cover, none of the differences in percentage cover of the other functional groups between April 2005 and October 2006 were significant. The percentage covers of predators and grazers were <1% at all sites and are thus not included in these analyses. The results of the Students t-tests are provided in Table IV.

Once kelp had been removed from the experimental lanes, multivariate analyses identified that some separation in benthic community structure between impact and control lanes was evident at all three sites (Figure 18), particularly separation of response groups such as green and red foliose algae. Subsequent SIMPER analyses confirmed that the biotic components that distinguished the cut lanes were red and green foliose seaweeds. The foliose algae in particular grew prolifically on removal of the kelp canopy, forming a thick ‘carpet’ covering the seabed. Green algae were virtually absent in uncut treatments but appeared in most cut treatments.

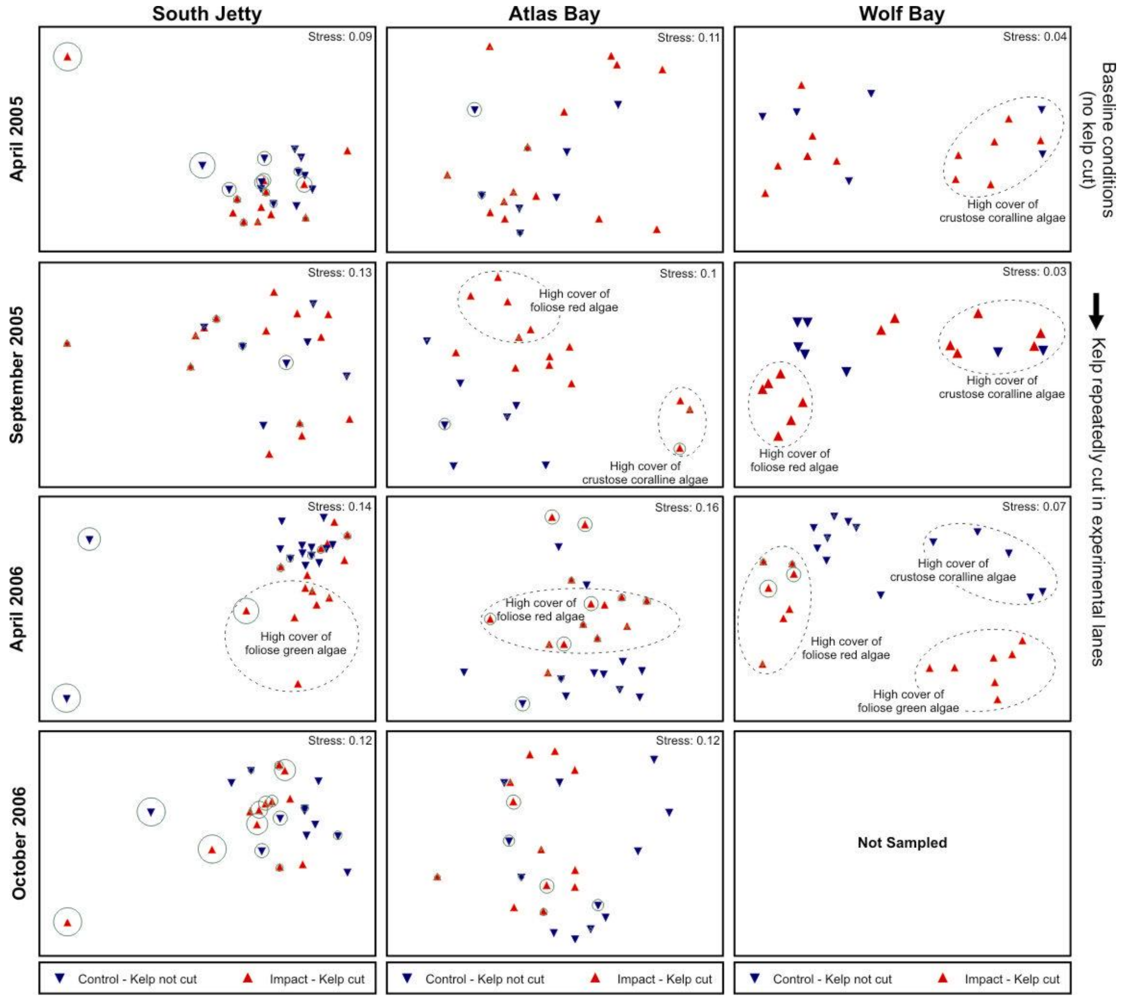


Figure 18. MDS ordination plots illustrating the differences in benthic community structure between control and impact lanes within each of the three study sites, between April 2005 and October 2006. An overlay of the proportion of sandy seabed (green circles) recorded in the quadrats is provided (Data range from 0-100% sand). Rings (dotted black lines) distinguish response groups.

Table IV. Results of Students t-tests comparing the percentage cover of functional groups at the start (i.e. before cutting) and end of the study in cut (impact) and uncut (control) areas at the three experimental sites. Significant values are shown in bold. Negative t-values indicate an increase; positive t-values indicate a decrease.

Functional Group		**South Jetty**		**Atlas Bay**		**Wolf Bay**	
		t-value	**p**	**t-value**	**p**	**t-value**	**p**
Kelp	C	-0.525	0.605	-0.821	0.423	-0.193	0.849
	I	-0.279	0.783	1.548	0.135	4.478	**0.000144**
Red foliose algae	C	0.000	1.000	0.301	0.767	0.778	0.447
	I	-0.571	0.574	-0.897	0.379	-1.016	0.319
Green Foliose algae	C	-0.525	0.605	-1.141	0.270	-0.988	0.337
	I	-2.306	**0.031**	-2.846	**0.00892**	-2.464	**0.0209**
Coralline algae	C	-2.191	**0.0393**	-0.687	0.501	-0.759	0.458
	I	-0.231	0.819	-0.325	0.748	1.130	0.269
Filter-feeders	C	0.512	0.614	-1.594	0.129	-0.427	0.675
	I	0.210	0.836	0.752	0.459	2.837	**0.00890**
df		22		24		25	

The results of ANOSIM tests (Table V) to some extent confirm the patterns illustrated by the MDS ordination plots. However, the observed differences were significant only in some cases. For example, in September 2005, five months after the kelp had first been removed from the experimental lanes, the impacted communities were significantly different at Atlas Bay only, suggesting that recovery of communities may be relatively rapid, or that it takes time for the impact to become evident. Kelp was re-cut at Atlas Bay in February 2006, and at South Jetty and Wolf Bay in March 2006, shortly before the benthic survey in April. At that time, the communities in the cut lanes all showed a significant difference to those in un-cut control areas. By October 2006, 7-8 months after removal of the kelp, communities at both the Jetty site and at Atlas Bay appeared to have recovered to the extent of being indistinguishable from un-cut control areas. Unfortunately, due to the onset of unfavourable diving conditions, the Wolf Bay site could not be sampled during the October 2006 survey.

An overlay of the percentage cover of sand at the three sites during the four sampling periods (Figure 17) suggests that sand tends to become more prevalent in areas where kelp has been removed. Student's T-tests, however, revealed that the cover of sand was significantly higher in the experimentally cut lanes only at Atlas Bay in April 2006 (t-value-2.104, df=24, p=0.046).

Table V. Results of ANOSIMs to test for differences in community structure between experimental lanes and uncut control areas at the three experimental sites during the study. The R-statistic (R) and significance value (p) are given. Values in bold are significant.

Site	April '05		September '05		April '06		October '06	
	R	p	R	p	R	p	R	p
South Jetty	0.096	0.059	-0.168	0.994	0.134	**0.018**	0.034	0.199
Atlas Bay	-0.089	0.826	0.297	**0.016**	0.244	**0.003**	0.096	0.079
Wolf Bay	0.063	0.193	-0.001	0.395	0.207	**0.016**	Not sampled	

Further multivariate analyses using the data for all time periods for each study site were conducted to investigate the "recovery rate" of benthic communities in experimentally cut lanes at the three study sites. "Recovery" in this case refers to the progressive change in the benthic community structure in the experimentally cut lanes until its composition is no longer significantly distinguishable from areas where kelp remained uncut. The intra-site variability in community structure was high, most likely as the result of differences in depth and seabed composition between quadrats. Despite this heterogeneity, separation of communities at different stages of recovery after the removal of kelp is evident. Subsequent ANOSIMs identified that recovery rates varies between sites (Table VI), with the sediment impacted South Jetty site appearing to recover within five months, whereas at Wolf Bay and Atlas Bay differences were still significant five and eight months, respectively, after kelp had been removed.

Table VI. Results of ANOSIMs to test for temporal changes in benthic community structure at various time periods (0, 1, 5 & 8 months) after the removal of kelp, at the three experimental sites. The R-statistic (R) and significance value (p) are given. Significant values are shown in bold.

	South Jetty		Atlas Bay		Wolf Bay	
Groups	R	p	R	p	R	p
0, 1	0.169	**0.028**	0.389	**0.001**	0.333	**0.002**
0, 5	0.126	0.072	0.279	**0.003**	0.140	**0.012**
0, 8	-0.038	0.615	0.270	**0.004**	--	--
Global R	0.106		0.301		0.235	
p	**0.033**		**0.001**		**0.001**	

Investigating the differences between cut and control lanes further, Student's t-tests identified that during the last survey in October 2006, the percentage cover of red and green foliose algae were higher in the cut lanes compared to the uncut control areas, although this difference was only significant at South Jetty and Wolf Bay (Table VII). Conversely, as would be expected, the cover of kelp was lower in the cut lanes than in the uncut lanes, although only at Wolf Bay was this difference significant. At both Atlas Bay and Wolf Bay the cover of filter-feeders (primarily *Dendropoma*) was also significantly lower in the cut lanes, probably as a consequence of a decrease in the cover of crustose coralline algae. These algae provide the dominant understory cover in the flat, shallow reef areas, and are the first to respond to the cutting of kelp by showing bleaching within a day or two after removal of the canopy.

This confirms the results of the multivariate statistical analyses, which suggested that benthic communities within the cut lanes are significantly different from uncut control areas.

Table VII. Results of Student's t-tests comparing differences in the percentage cover of functional groups in cut (impact) and uncut (control) areas at the three experimental sites during October 2006. Significant values are shown in bold. Negative t-values indicate higher cover in cut (impact) lanes; positive t-values indicate higher cover in uncut (control) lanes.

Functional Group	South Jetty		Atlas Bay		Wolf Bay	
	t-value	p	t-value	p	t-value	p
Kelp	1.795	0.086	1.074	0.294	5.646	**0.000007**
Red foliose algae	-2.250	**0.035**	-1.349	0.191	-2.504	**0.019**
Green Foliose algae	-2.306	**0.031**	-1.830	0.081	-2.444	**0.022**
Coralline algae	0.561	0.580	-0.612	0.547	1.667	0.108
Filter-feeders	0.523	0.605	2.139	**0.044**	2.773	**0.010**
df	22		22		25	

3.4 Rock Lobster Abundance and Distribution

Figure 19 illustrates the mean abundance of rock lobsters recorded in the cut experimental lanes (AB and CD) and in the uncut control area (BC) at the three study sites. Data from both the baseline survey in April 2005, and the three follow-up surveys in September 2005, April and October 2006 are presented. The overall

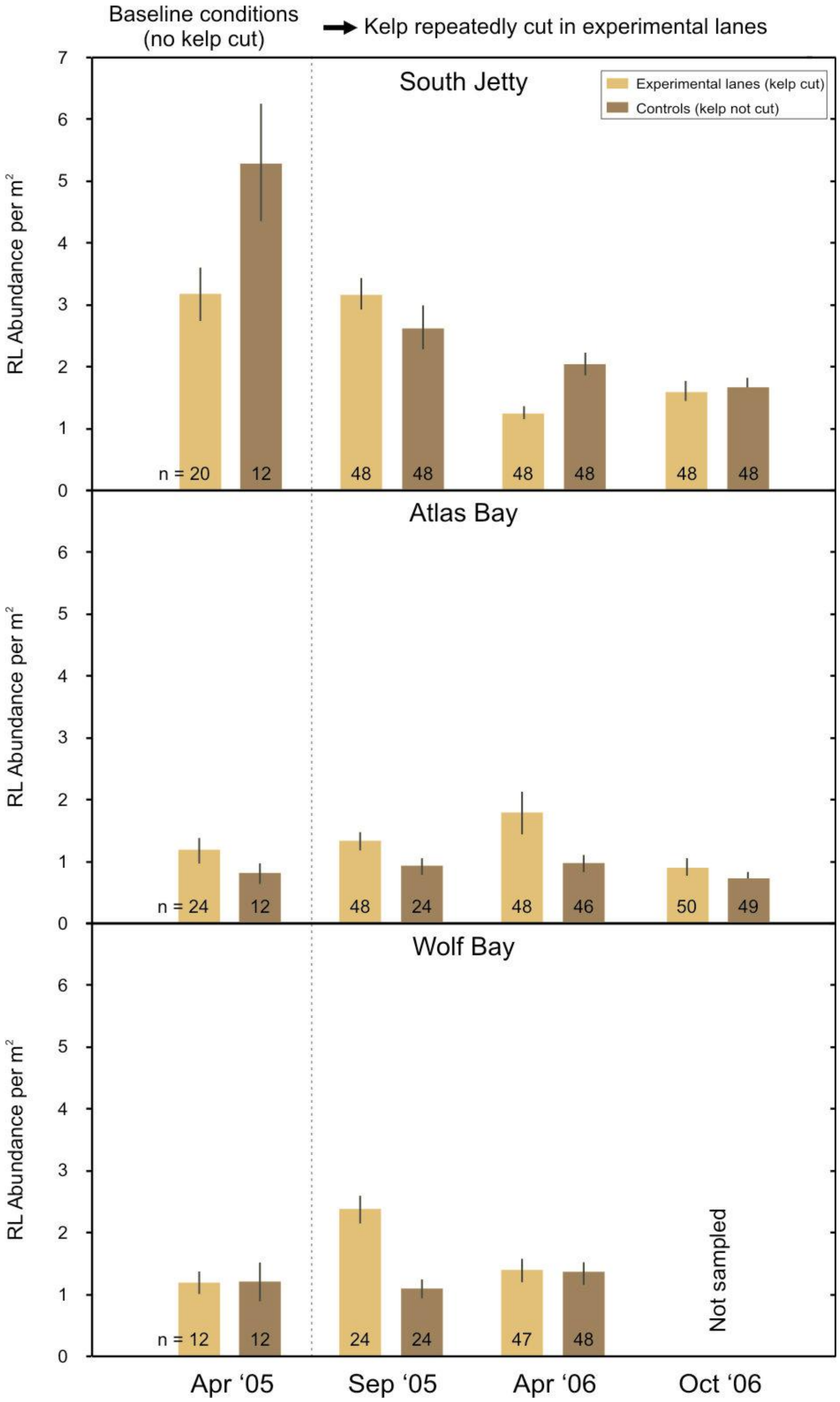


Figure 19. Rock lobster (RL) abundance (numbers per square meter) in cut and uncut lanes at the three study sites during April and September 2005, and April /May and October 2006. The standard error of the mean, and the number of counts for each lane are also shown.

abundance of lobsters was significantly different between the three sites (Kruskal-Wallis ANOVA: $H_{2,\,788} = 100.650$, $p<<0.001$), being higher at the South Jetty site than at the Atlas Bay or Wolf Bay sites.

Superimposed on the inter-site variability in rock lobster abundance, there was also considerable intra-site, temporal variability in abundance. When comparing the lobster densities before and after removal of the kelp in the experimental lanes, Kruskal-Wallis ANOVAs identified that there was a steady decrease in lobster abundance in both the experimentally cut as well as uncut lanes at the South Jetty site, particularly between the 2005 and 2006 surveys. At Atlas Bay and Wolf Bay, in contrast, this pattern was not evident, and temporal differences in both cut and uncut lanes were in most cases insignificant. Overall, however, temporal variability in abundance of rock lobsters at Atlas and Wolf Bays was significant. Unfortunately, due to the onset of unfavourable diving conditions, the Wolf Bay site could not be sampled during the October 2006 survey. The results of the ANOVAs are provided in Table VIII.

Table VIII. Results of Kruskal-Wallis ANOVAs testing for temporal changes in rock lobster abundance in the cut and uncut lanes, and overall at the three study sites. Significant differences are indicated in bold.

Site	Cut Lanes		Uncut Lanes		Overall	
	H	p	H	p	H	p
South Jetty	39.956	**<<0.001**	26.649	**<<0.001**	57.467	**<<0.001**
Atlas Bay	5.640	0.131	2.802	0.423	9.500	**0.0233**
Wolf Bay	15.526	**0.0004**	0.291	0.864	10.358	**0.0056**

After cutting, there were no significant differences in rock lobster abundance between cut and uncut lanes at South Jetty (Student's T-test, t-value=-0.014, df=262, p=0.989). In contrast, densities were significantly higher in the cut lanes at both Atlas Bay (Student's T-test, t-value=3.016, df=263, p=0.00282) and Wolf Bay (Student's t-test, t-value=3.040, df=165, p=0.00275). Because of this variability, changes in rock lobster abundance in response to the removal of kelp should be interpreted with caution. However, it is clear that there was no evidence that cutting kelp had a negative effect on rock lobster numbers at any of the sites.

The distribution of rock lobsters with depth recorded at the three study sites during the four surveys is illustrated in Figure 20. At Atlas Bay and Wolf Bay, no clear trend in rock lobster density with depth was evident, however, at the South Jetty site, abundance appeared to decline with depth. This is likely due to the increasing cover of sands and silts on the seabed beyond ~40 m from the low water mark (or a depth of 3.5 m).

3.5 Rock Lobster Population Size Structure and Sex Ratio

The length frequency distributions and mean carapace lengths (CLs), and sex ratios of rock lobsters captured at the three study sites during the three surveys are shown in Figure 21 and Table IX, respectively.

During all four surveys, the sex ratio of lobsters at the three sites was slightly, but insignificantly, skewed towards males. At all sites, over 90% of the rock lobsters in the diver's catch were below the minimum legal size limit of 65 mm CL, confirming that the small bays along this section of the coastline serve as recruitment and/or nursery areas for *Jasus lalandii*.

3.6 Puerulus Collectors

Puerulus collectors obtained from MFMR, which were constructed out of galvanised steel, rather than stainless steel, required constant maintenance and repair throughout the course of the study. The collectors suffered from severe corrosion, even over short periods (Plates 12 & 13). In most cases, the bolts holding the collector plates together, or the dorsal cross-pieces corroded through resulting in subsequent de-lamination of the collector plates.

Subsequent to submission of Progress Report III, approval was given to extend the field aspects of the project until September 2006 in order to obtain a more sound database. Following this approval, and interest shown by MFMR in continuing with the collection of data from the puerulus collectors, stainless steel components were purchased and manufactured. During the October survey, all galvanised components on existing collectors were replaced with stainless steel. Collectors previously removed were repaired with stainless components and re-installed. All study sites therefore now have stainless collectors.

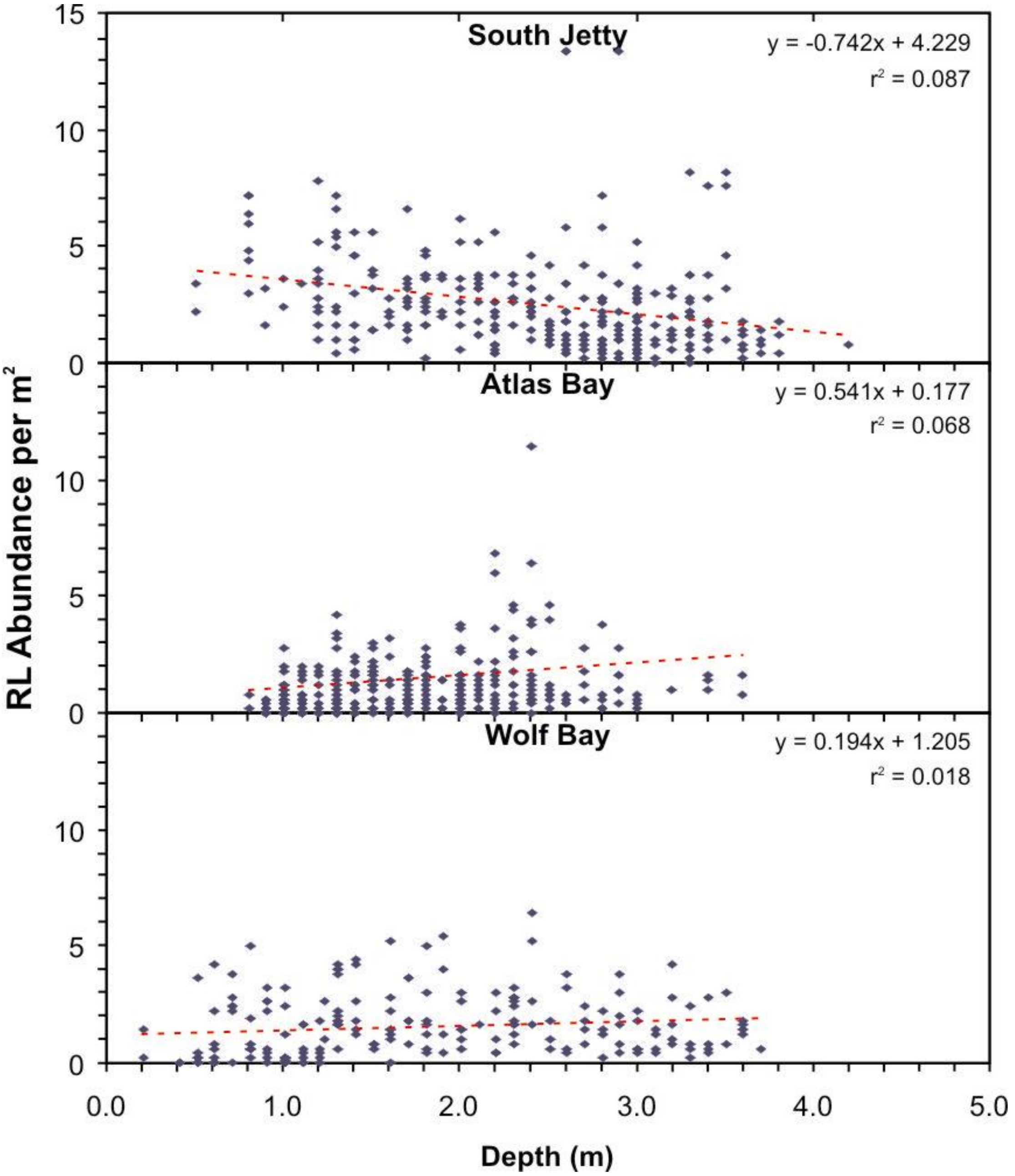


Figure 20. Changes in rock lobster abundance (numbers per square meter) with depth at the three study sites.

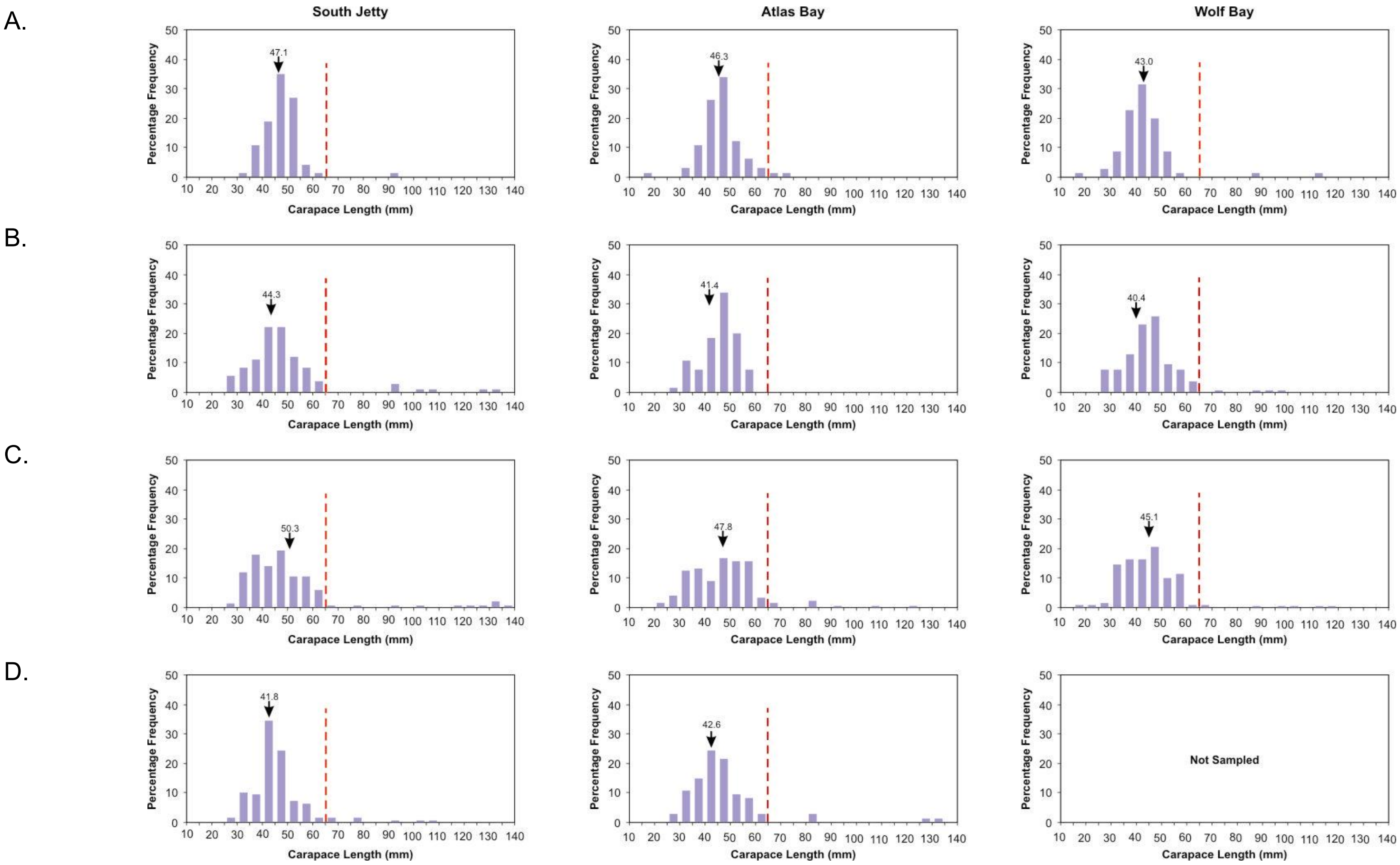


Figure 21. Length frequency distributions of rock lobsters caught at the three study sites in A) April 2005, B) September 2005, C) April 2006, and D. October 2006. The mean carapace length of lobsters at each site is also shown, as is the minimum legal size of 65 mm CL (dashed lines).

Table IX. Mean carapace length (CL) of rock lobsters measured at the three study sites, in April and September 2005, and April and October 2006. Maximum and minimum lengths, numbers measured and the percentage of rock lobsters under the minimum legal size of 65 mm, and the sex ratio are also given.

SITE	Year	Mean CL	Min CL	Max CL	n	% Undersize	Sex Ratio %M : %F
South Jetty	Apr '05	47.1	34.0	92.0	74	98.65	53 : 47
	Sep '05	44.3	22.0	126.0	108	93.52	55 : 45
	Apr '06	50.3	29.0	143.0	150	92.62	59 : 41
	Oct '06	41.8	22.0	105.0	139	96.4	62 : 38
Atlas Bay	Apr '05	46.3	17.0	72.0	65	96.92	51 : 49
	Sep '05	41.4	24.0	55.0	65	100.00	57 : 43
	Apr '06	47.8	21.0	123.0	119	93.28	56 : 44
	Oct '06	42.6	22.0	126.0	74	94.6	53 : 47
Wolf Bay	Apr '05	43.0	15.0	110.0	70	97.14	51 : 49
	Sep '05	40.4	21.0	91.0	170	97.65	59 : 41
	Apr '06	45.1	16.0	119.0	164	95.73	56 : 44
	Oct '06	- Not Sampled -					

Plate 12. Corrosion of galvanised steel collector bolts and cross-pieces.

Plate 13: Corrosion of galvanised steel collector after a six-month period in the sea.

Plate 14. Fouling of puerulus collector by kelp juveniles, and red and green foliose algae.

Fouling of the collectors by kelp juveniles, various filamentous (*Pterosiphonia cloiophylla*) and foliose red algae (*Hymenema venosa* and *Botryoglossum platycarpa*), and green algae (*Ulva* sp. and *Cladophora flagelliformis*) was recorded to varying degrees at all of the study sites (Plate 14).

The mean number of pueruli and juveniles of *Jasus lalandii* caught in the collectors in the experimental lanes and uncut control zone at the three study sites since the start of the study are illustrated in Figure 22. On average, more pueruli were recorded in collectors from areas were kelp was not removed, although Student's t-tests indicated that differences were only significant at the South Jetty site (Table X). In contrast, juvenile lobsters were more abundant in collectors in the cut lanes, at all three sites, although nowhere was the difference significant (Table X). Of interest, however, is that collectors at the fines-impacted South Jetty site caught significantly more pueruli than those at the Atlas Bay and Wolf Bay reference sites (ANOVA, F=17.494, p<<0.0001). Catches of juveniles at the South Jetty site was also significantly greater than at the two control sites (ANOVA, F=3.516, p=0.0333). This implies that the presence of mining-related sediments at the South Jetty site does not negatively influence puerulus settlement as previously conjectured, and that other environmental factors need to be taken into account when considering settlement stimuli for rock lobster pueruli.

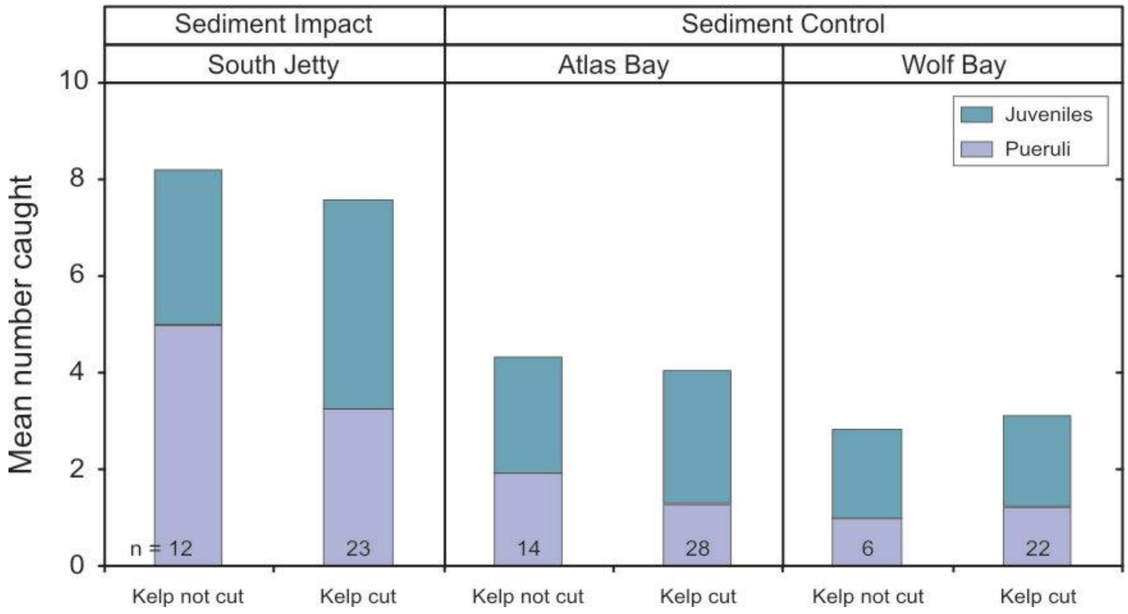


Figure 22. Mean number of *Jasus lalandii* pueruli and juveniles recorded per Booth crevice collector in the experimental and control lanes at the three study sites. The number of collectors sampled (n) is also provided.

Table X. Results of Student's t-tests comparing differences in the numbers of pueruli and juvenile lobsters recorded in collectors from cut and uncut lanes at the three study sites. Significant differences are indicated in bold.

	Pueruli			Juveniles		
Site	**t-value**	**df**	**p**	**t-value**	**df**	**p**
South Jetty	2.084	40	**0.0436**	-0.780	40	0.440
Atlas Bay	1.042	33	0.305	-0.297	33	0.769
Wolf Bay	-0.305	26	0.762	-0.038	26	0.970

The length frequency distribution of the pueruli and juveniles caught in the Booth crevice collectors between September 2005 and October 2006 is illustrated in Figure 23, and the relationship between carapace length and wet weight is shown in Figure 24.

Temporal trends in the catches of pueruli at the three study sites are presented in Figure 25, and emphasise the generally higher values at South Jetty. Unfortunately interim surveys scheduled for the period between April and September 2005 could not be undertaken as planned due to delays in the issuing of mine clearances for the survey team. Likewise, work visas were only re-issued in late December 2005, precluding interim surveys between October 2005 and January 2006.

By-catch species found in the collectors included various clinid species, rocksuckers (*Chorisochismus dentex*), Cape rock crabs (*Plagusia chabrus*), the reticulate kelp louse (*Paridotea reticulata*), sand shrimps (*Palaemon pacificus*), six-gill hagfish (*Eptatretus hexatrema*), and the common octopus (*Octopus vulgaris*). Over longer periods of exposure, colonisation by colonial encrusting ascidians and various polychaete fan-worms also occured.

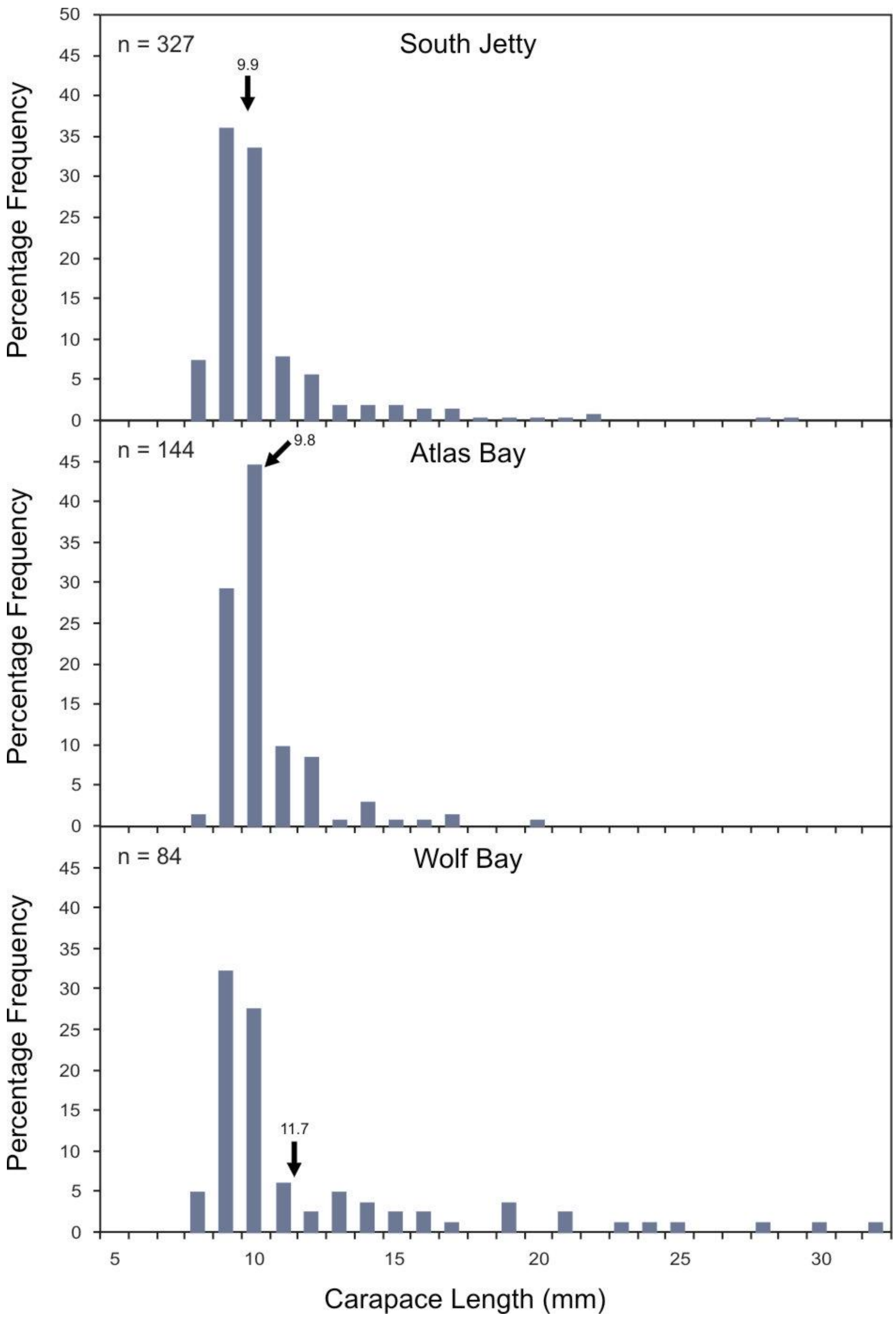


Figure 23. Length frequency distributions of rock lobster pueruli and juveniles caught in Booth crevice collectors at the three study sites between September 2005 and October 2006. The mean carapace length is indicated by an arrow. The number of lobsters measured is also provided.

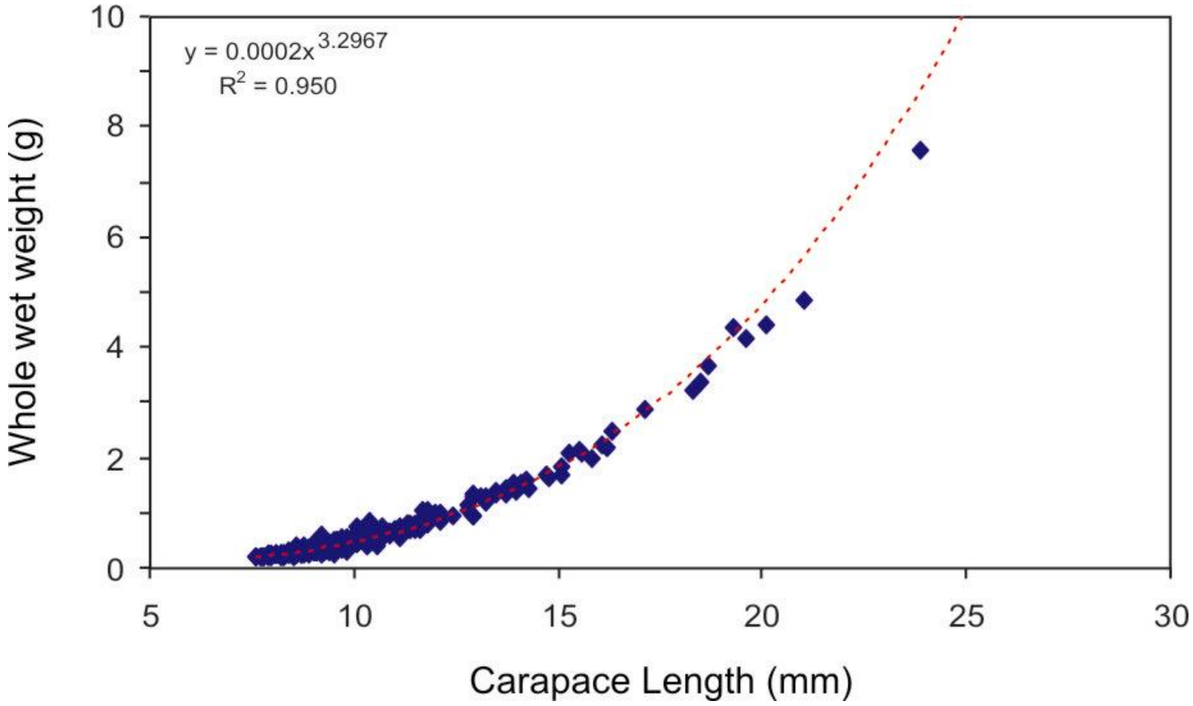


Figure 24. The relationship between carapace length and wet weight for rock lobster pueruli and juveniles at the three study sites.

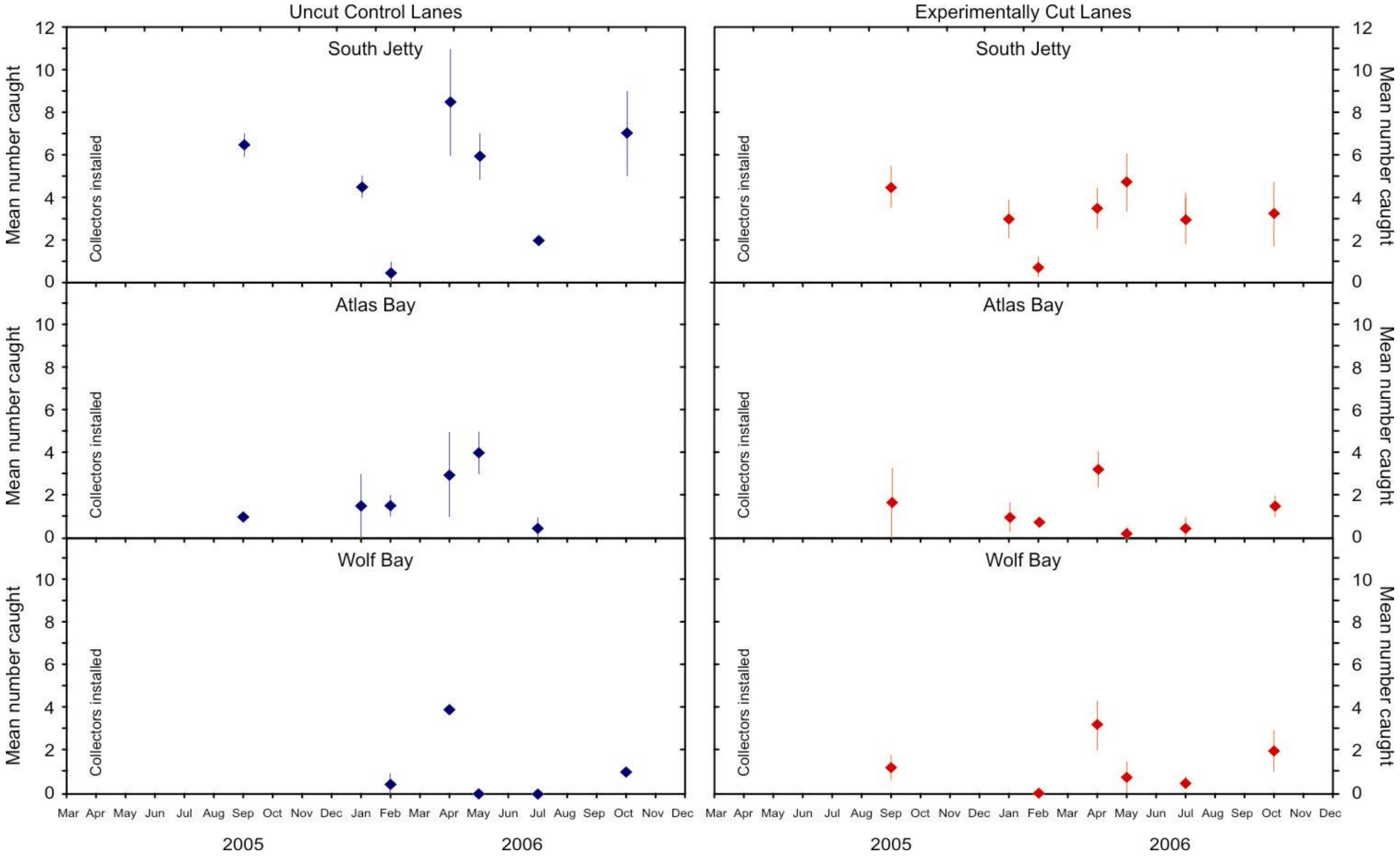


Figure 25. Temporal trends in the settlement of rock lobster pueruli in Booth crevice collectors in uncut control lanes and lanes where kelp had been experimentally removed, at the three study sites.

4. DISCUSSION

4.1 The Effects of Kelp Cutting on Kelp Biomass and Abundance

On the southern African west coast, the kelps *Laminaria pallida* and *Ecklonia maxima* dominate the algal biomass in the shallow subtidal zone. These species are notable for their exposure to large swells, with *E. maxima* in particular, being commonly thought to exert an important breakwater effect for inshore, shallow water communities (Field *et al*. 1977; Velimirov *et al*. 1977). In the clear waters of the southern Benguela, *Ecklonia* can extend seawards to depths of about 10-12 m, forming a dense floating canopy. The smaller *Laminaria*, which typically forms a sub-canopy underneath *Ecklonia*, can continue its seaward extent to the photic limit, having been recorded in up to 30 m depth (Branch *et al*. 1994). In southern Namibia, however, high natural turbidity in nearshore areas affects light penetration thereby limiting primary production and the depth to which the kelps normally grow (Velimirov *et al.* 1977; Field *et al*. 1980; Jarman & Carter 1981). Consequently, the kelps are restricted to shallower waters where substratum is suitable, and *Ecklonia* is replaced by *Laminaria* as the dominant kelp, occurring as a few interspersed plants within the dense *Laminaria* beds (Molloy 1992; Pulfrich & Penney 2006). The ecology of the South African kelp beds is reviewed in Stegenga *et al*. (1997).

Intertidal and subtidal macrophytes, because of their sedentary nature, tend to integrate the effects of long-term exposure to adverse conditions and are thus good indicators of environmental impacts. Consequently, it has been postulated that kelp beds could be used as indicators of climate change in nearshore ecosystems. As the first step towards generating some of the baseline and natural variability data against which the effects of nearshore diamond mining operations might subsequently be assessed, a infra-red aerial survey of the kelp beds between Mittag (~40 kms north of the Orange River mouth) and Diaz Point near Lüderitz, was recently undertaken (Pulfrich & Penney 2006). The principal objective of this study was to calculate the area of kelp beds along the southern Namibian coastline, and so obtain an index of suitable habitat available to rock lobsters. Although the coastline has been influenced by mining activities for up to six decades, it is hoped that this initial survey will facilitate assessments of variations in kelp area in response to natural and mining-induced effects in the long-term. Similar aerial surveys have also been undertaken along the

South African West Coast, with the aim of obtaining an index of suitable habitat available to abalone *Haliotis midae* (Tarr 1993), as well as developing a coastal kelp database as a tool for effectively managing kelp resources (Rand 2006). In his recent study, Rand (2006) compared the use of colour Infra-red photographs (taken in 1996), digital infra-red images (2005 aerial survey), and available Landsat 7ETM and satellite imagery to estimate the area of kelp beds between Cape Point and the Orange River mouth.

Pulfrich & Penney (2006) estimated that the total area of kelp along the southern Namibian coastline amounts to 591.662 ha (Table XIa). Estimates of kelp area between the Orange River mouth and Cape Columbine amounted to 3088.68 ha (Rand 2006) (Table XIb). This is, however, likely to be an underestimate as it is based only on the plants reaching the surface at low spring tide, and excludes sub-surface kelp occurring on subtidal reefs.

In Namibia, kelp area decreases southwards, reflecting the increasing proportion of sandy shoreline and seabed towards the Orange River pro-delta, and emphasising the role that river-related sediments have in structuring the coastal ecology in the region Figure 26). South of the Orange River the pattern is not as consistent, although high kelp area in seaweed concessions 18 and 15 reflect the predominantly rocky shoreline characterising this area (see Section 4.5) (Figure 27).

The high proportion of *Ecklonia* recorded at the South Jetty and Atlas Bay sites during this study was unexpected. Despite this, however, the density of *Laminaria* plants per square metre (6-20) generally tended to be higher than for *Ecklonia* (2-10). This is in agreement with previous reports for the southern Benguela region (Velimirov *et al.* 1977; Velimirov & Griffiths 1979; Simons & Jarman 1981; Rothman 2006). These authors noted that despite lower densities, the bulk of *Ecklonia* resulted in this species nonetheless contributing between 50-70% to the overall kelp biomass in water down to 8 m depth at their study site on the Cape Peninsula. Both the density and biomass of *Ecklonia* was found to decline with increasing depth (Field *et al*. 1977; Rothman 2006), with biomass dropping to as low as 2% beyond 10 m depth, with *Laminaria* becoming increasingly dominant and contributing the remaining 98% to kelp biomass (Field *et al*.

Table XI. a) The estimated total area of kelp beds for each of the Namibian rock lobster zones south of Lüderitz (Pulfrich & Penney 2006), and b) The estimated total area of kelp beds for each of the kelp concessions between the Orange River mouth and Cape Columbine (Rand 2006).

a)

Rock LobsterZone	Kelp bed area (ha)	Length of coastline with potential fringing kelp (km)
10	3.928	0.231
11	3.681	0.417
12	300.053	50.204
13	79.353	60.833
14	65.162	24.946
15	0.929	1.926
16	0.000	0.000
Total	453.105	138.558

b)

Kelp Concession	Kelp bed area (ha)	Length of rocky coastline (km)
19	254.95	48.5
16	206.44	5.0
18	976.0	18.25
15	732.22	104.5
Groen-Spoeg	71.94	~15.0
14	206.64	63.75
13	10.8	4.25
Strandfontein	no data	~15
12	11.74	1.25
11	617.95	28.75
Total	3088.68	289.25

1977). The depth distribution of the two kelp species reported in these studies was, however, not as clear at the sampling sites near Lüderitz.

The biomass of *Ecklonia* varied between 2.3 kg. m^{-2} at Wolf Bay to 9.7 kg. m^{-2} at the South Jetty, thereby falling well within the range of 2.5-14 kg.m^{-2} reported by Levitt *et al.* (2002) for Danger Point in the Southern Benguela Region, but below the 10.5-21.3 kg. m^{-2} recorded by Rothman (2006), or the average 12 kg m^{-2} and 17 kg m^{-2} reported by Mann *et al.* (1979) and Allen & Griffiths (1981), respectively. The high contribution of *Ecklonia* to the overall kelp biomass at South Jetty is as a consequence of this site being characterised by relatively old *Ecklonia* plants, recognisable by the heavy epiphytic growth.

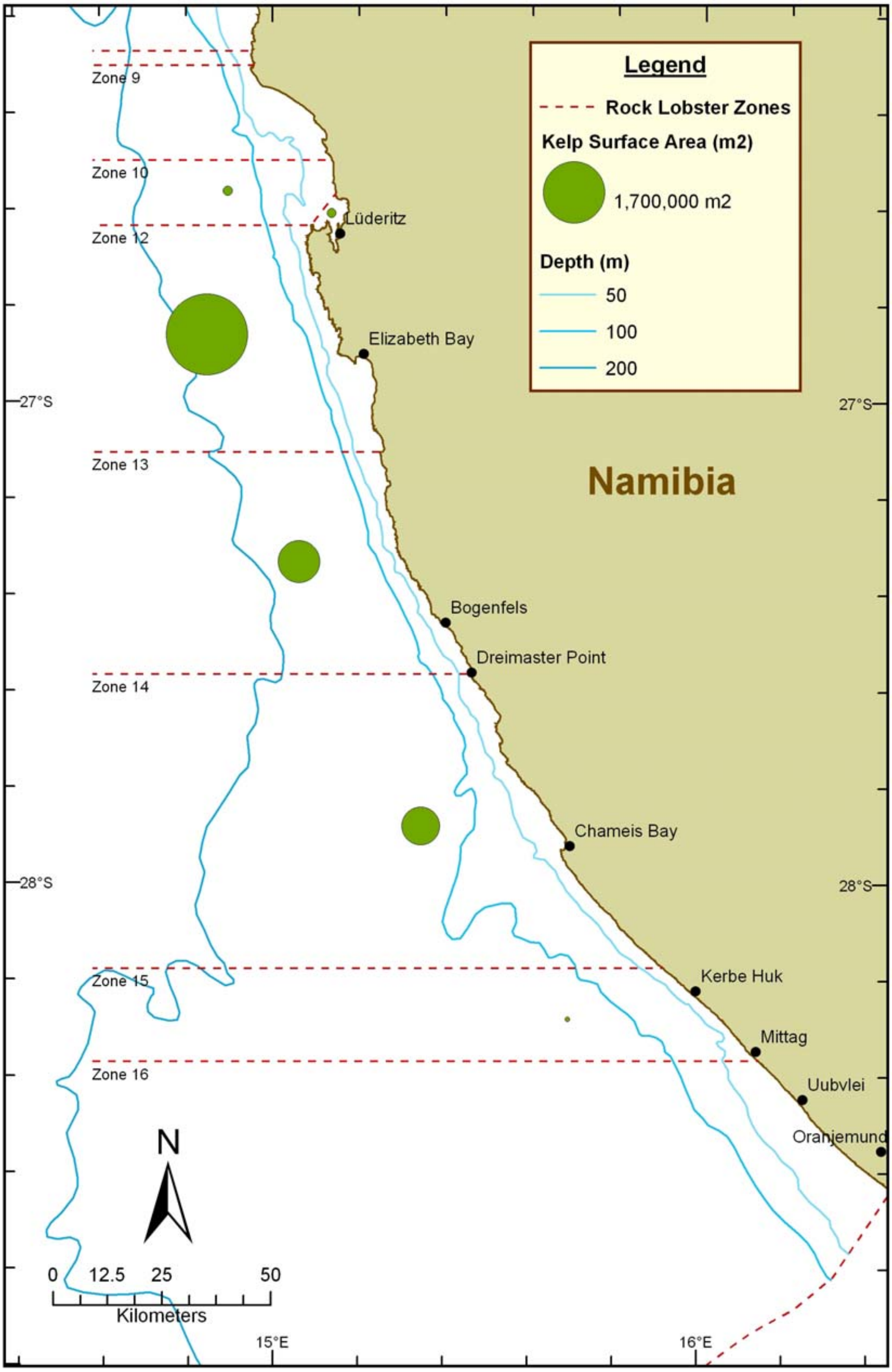


Figure 26. Estimated kelp bed area in the rock lobster fishing zones south of Lüderitz (adapted from Pulfrich & Penney 2006).

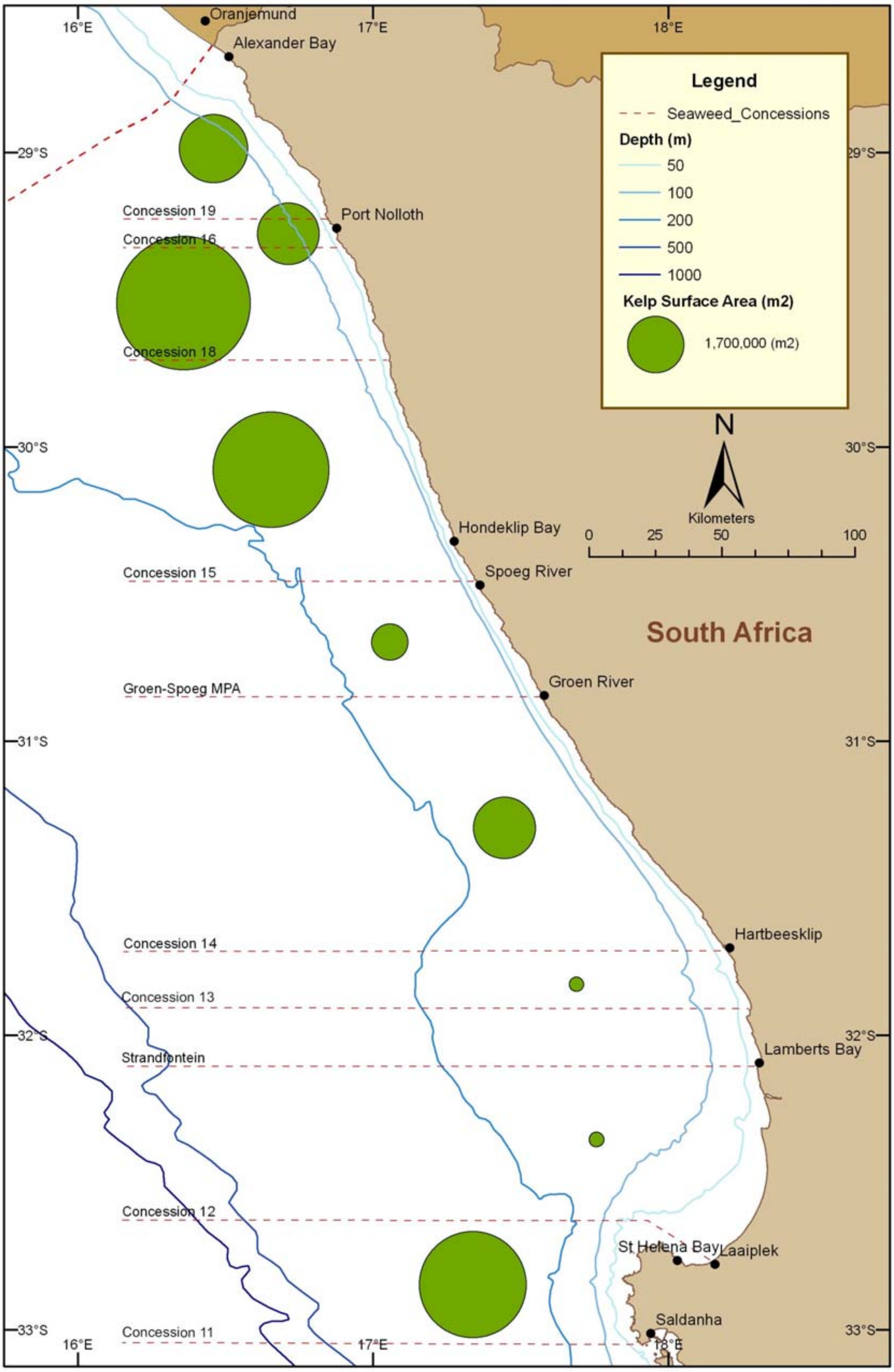


Figure 27. Estimated kelp bed area in the South African kelp concessions between the Orange River mouth and Cape Columbine.

In Namibia, the contribution of *Ecklonia* to the overall kelp density and biomass appears mostly to be related to exposure of the kelp bed to wave action; the higher the exposure, the lower the density of *Ecklonia*. Of the three study sites, the Atlas Bay site is the most sheltered, followed by the South Jetty site in Elizabeth Bay. Although located within Wolf Bay, the experimental area at this site is not protected by an offshore reef, and is directly exposed to the predominant SW swells. This finding is contrary to Field *et al*. (1977) who state that in the southern Benguela region, *Ecklonia* tends to be associated with more turbulent water in shallow inshore areas. Interestingly, Molloy (1992) reports that *Ecklonia* was absent at his sheltered sites near Diaz Point. Dayton (1975) suggests that different canopy guilds develop alternative morphological adaptations in which exploitation of light is enhanced in taller canopies, whereas shorter canopy guilds are better adapted to tolerate stress from wave surge. A large floating canopy species such as *Ecklonia* is thus more likely to be adapted for light exploitation, but be susceptible to severe disturbance from waves. The drag created by the floating canopy in rough seas will be high, often resulting in the holdfast being torn off the seabed. This would also explain why *Ecklonia* is more dominant at the sheltered South Jetty site, which is characterised by highly turbid water as a result of the discharge of fine sediments from the nearby Elizabeth Bay mine. At this site *Laminaria* occurs virtually only in shallow water where light penetration is sufficient for photosynthesis (<3 m).

The biomass of *Laminaria pallida* recorded during this study varied between 2.2 kg. m^{-2} at South Jetty to 10.8 kg. m^{-2} at Wolf Bay. These figures compare well with the 6-16 kg. m^{-2} obtained for *L. hyperborea* beds by Sjøtun *et al.* (1993) in Norway. Kain (1977), in contrast, estimated the biomass for the same species at the Isle of Man at 14 kg. m^{-2}. Reports for other species range from 0.9 kg. m^{-2} for *L. solidungula* in Canada (Chapman & Lindley 1981), through 11.5-16 kg. m^{-2} for *L. longicruris* and 16 kg. m^{-2} for *L. digitata* also in Canada (Mann 1972, 1973), indicating the enormous regional and geographical variability in kelp biomass. This is to be expected, as both depth profile and bottom topography play a major role in the distribution and abundance of kelps.

Spatial variability in kelp forest structure (average age and size of plants) is well documented (Dayton 1975; Molloy 1992; Sjøtun *et al*. 1993, amongst others). Wave action in particular is known to be an important cause of growth adaptations in various macroalgal species, often leading to the development of different community structures in different wave-stress regimes (Gerard & Mann 1979; Cousens 1982; Dayton *et al*. 1984; Cheshire & Hallam 1988; Norton 1994; Molloy & Bolton 1996; Ralph *et al*. 1998; Roberson & Coyer 2004). In exposed habitats in Namibia, *Laminaria pallida* is characterised by long, thick stipes and low frond surface area (fronds have narrow blades, but are comparatively long) (Molloy 1992). These morphological trends are consistent with increased strength and reduced drag, and they correspond to similar responses to exposure observed in other kelps (e.g. Armstrong 1989; Roberson & Coyer 2004; Wernberg & Thomsen 2005). Where *Laminaria* occurs as an understory species in the presence of *E. Maxima*, however, stipes tend to be comparatively shorter, particularly in more exposed locations (Molloy 1992). This is possibly a morphological adaptation to both the high wave exposure and turbid water characteristic of the coastline. Norton *et al.* (1982) found that the surface area/mass ratio of *Laminaria longicruris* growing in exposed areas were a third of that from populations growing in sheltered areas. In must, however, be kept in mind that wave exposure cannot be seen entirely separate from other local processes such as depth, grazing pressure or nutrient levels.

The ecological effects of the natural or anthropogenic removal of the macroalgal canopy in nearshore waters have been discussed in numerous studies. While many of these investigate the effects of large-scale commercial harvesting of kelps (e.g. Simons & Jarman 1981; Kimura & Foster 1984; Barilotti & Zertuche-González 1990; Foster & Barilotti 1990; Sharp & Pringle 1990; Christie *et al*. 1998; Levitt *et al*. 2002; Waage-Nielsen *et al.* 2003, amongst others), others discuss the consequences of smaller-scale commercial or artisanal harvesting (Bustamante & Castilla 1990; Schiel & Nelson 1990; Vasquez & Santelices 1990), and experimental removal (Bodkin 1988; Levitt *et al*. 2002). Still other authors have described the effects of natural disturbances such as severe storms (Dayton *et al*. 1984; Dayton *et al.* 1989), outbreaks of grazers (reviewed by Dayton 1985a, 1985b; Chapman 1986; Harrold & Pearse 1987) or large-scale oceanographic events such as El Niños (Dayton *et al.* 1992; Schroeter *et al*. 1995) (see Section 4.7).

The effects of these disturbances on the kelps themselves are similar, varying only in intensity, extent and frequency. The effects of storm damage for example, are not as extensive as mechanical harvesting, as storms tend to clear smaller areas due to variations in topography and depth, thereby leaving a fragmented landscape, rather than a large cleared area (Rinde *et al.* 1992; Christie *et al.* 1994). Storms also tend to impact patches of the floating adult canopy more severely than the understory canopy (Dayton & Tegner 1984).

The proportion of biomass removed during harvesting depends largely on the harvesting method. Barilotti & Zertuche-González (1990) estimate that 33-50% of total plant biomass is removed during cutting of surface fronds of *Macrocystis pyrifera* in Mexico and California. Dragraking, which removes plants above a certain stipe length, is estimated to reduce stipe density by 40%, whereas spinning entanglement equipment reduced density by up to 68% (Sharp & Pringle 1990). In Norway, kelp trawling removes 6-13% of the kelp population, although on flat, rocky seabeds harvesting efficiency may be considerably higher, and canopy-forming plants may be cleared totally from large areas (Christie *et al*. 1994; Waage-Nielsen *et al.* 2003). Where macroalgae are removed manually by either hand-picking or cutting of the plant just above the holdfast (Bustamante & Castilla 1990; Schiel & Nelson 1990; Levitt *et al*. 2002; this study), harvesting efficiency is likely to be relatively intense, with over 95% of the biomass being removed. The impact is, however, limited to much smaller areas than can be cleared by mechanical means.

Research into the recovery of kelp in disturbed areas has identified that the rate of recovery depends on a number of factors. For example, time of removal has been shown to significantly affect the recovery rate of both *Ecklonia radiata* and *Durvillea antarctica* off New Zealand (Schiel & Nelson 1990), and L*essonia nigrescens* off Chile (Vasquez & Santelices 1990). If these macroalgae are harvested after the reproductive season, natural re-seeding is low, and the available space is invaded by other algal species (usually fucoids) thereby preventing recolonisation by the dominant kelp (see also Reed & Foster 1984). The resulting mixed stands are temporarily dominated by other sub-canopy or perennial understory species. However, the faster growth rates of the canopy-forming kelps ultimately out compete these by over-shading the smaller species (Dayton 1985; Schiel 1988; Vasquez & Santelices 1990).

In contrast, Christie *et al*. (1998) found that for *Laminaria hyperborea* recovery of the kelp bed is independent of the seasonality of harvesting. It appears that *L. hyperborea* juveniles, which can occur at densities of up to 80 individuals. m^{-2}, have the ability to persist in the understory for several years, ready to take over as soon as the adult canopy is removed (Svendsen 1972; Dayton 1985; Røv *et al.* 1990). Sharp & Pringle (1990), for example, reported a pulse of recruits six months after harvesting of *Laminaria longicruris*, with densities exceeding the pre-harvest density by 35%. They found that within three months of removal of the kelp, harvested areas were difficult to distinguish from uncut areas, and within a year, biomass recovered to pre-harvest levels. Episodic recruitment has also been reported for *Macrocystis pyrifera* in California (Dayton 1985), and *Ecklonia maxima* in southern Africa (Levitt *et al*. 2002).

Recruitment in mature *Macrocystis* beds appears to be inhibited due to canopy shading (Reed & Foster 1984; Schroeter *et al*. 1995), occurring naturally only after surface canopies are thinned by storms (Kimura & Foster 1984). Harvesting of the *Macrocystis* canopy therefore results in increased recruitment, even during non-reproductive periods (winter) due to increased light penetration. Hernández-Carmona (1996) established that growth rates in harvested areas in *Macrocystis* forests in Mexico were significantly higher than in uncut forests, with biomass recovery subsequent to harvest being greater than 100% of pre-harvest levels. Re-growth in these species therefore does not depend solely on the recruitment success in the year of harvesting. Due to their faster growth rate than other fucoid species, recruits rapidly form a dominant canopy thereby ensuring the persistence of the kelp forest by the dominant species, and do not provide the opportunity or space for alternative kelp species to become established. In some cases, however, intraspecific competition results in dramatic declines in kelp densities in harvested areas in the first 2-3 years post-cutting as the plants mature (Christie *et al*. 1998). Kimura & Foster (1984) also reported loss of young plants in harvested areas during winter storms following winter recruitment, although Barilotti & Zertuche-González (1990) found winter plant loss after harvesting (46-49% loss) to be insignificant compared to loss in unharvested areas (see also Dayton & Tegner 1984).

In *Ecklonia maxima*, spore release is strongly seasonal, peaking in December (Joska & Bolton 1987). This species appears to be adapted to disturbances as spore output is high (Joska & Bolton 1987) and sporophytes grow rapidly (Bolton & Levitt 1985; Rothman 2006), particularly after removal of the canopy. Recruits are abundant in the sub-canopy and can contribute up to 64% of the plant density (average 9-10 plants.m^{-2}). Recruitment, however, appears to have long-term or perhaps stochastic rather than seasonal patterns, and juvenile densities can drop to as low as 1% in some years (Levitt *et al*. 2002; Rothman 2006). Cyclic reproduction is thus of less importance to the recovery of *Ecklonia* than the remnant stand of young sub-canopy sporophytes. In removal experiments where 99% of the kelp biomass was harvested, biomass recovered to pre-harvest levels within 18 months, although the canopy itself required longer to develop as stipe elongation is relatively slow and plants only reached the same size in 2-3 years (Simons & Jarman 1981; Levitt *et al*. 2002). Christie *et al*. (1998), working on *Laminaria hyperborea*, also found that recovery in plant size varies spatially, but occurred at between 2-6 years. Nonetheless, *E. maxima* can increase its biomass by 1% per day (Mann *et al.* 1979), and while a single plant can produce seven times its own weight per year, it has been estimated that the biomass of an entire kelp bed will turn over about twice a year (Simons & Jarman 1981).

Recovery of kelp forests after harvesting is also dependent on the harvesting frequency and method of removal. It was found that continual canopy harvesting of *Macrocystis pyrifera* over a period of five months or more resulted in decreased production each year, whereas short, pulsed harvesting with a 2-3 month break between harvests greatly improved production (Brandt 1923; Hernández-Carmona 1996). Sharp & Pringle (1990) reported that annual harvesting of *Ascophyllum nodosum* yielded progressively lower annual biomass. Similarly, frequent harvesting of *Durvillaea antarctica* in Chile resulted in higher densities but smaller plants at exploited sites (Bustamante & Castilla 1990). Both these species are harvested intertidally by cutting the stipe at the base and leaving the holdfast in tact. Recovery of *A. nodosum* to pre-harvest standing crop took 2.5-3 years, but under high exploitation levels where up to 95% of the biomass was removed, recovery was reported to take over 10 years (Sharp & Pringle 1990). Removal experiments with *Durvillaea antarctica* conducted in New Zealand, found that if the holdfasts were left intact during harvesting, only 32% of the original biomass could be re-harvest after 15 months. Complete removal of the holdfast, however, cleared larger areas resulting in a re-harvest 1.5 x the original weight removed (Hay & South 1981). In marked contrast, removal of the holdfast of

Lessonia nigrescens resulted in increased inter-plant distances, which favoured the long-term establishment of high grazer densities, thereby inhibiting juvenile kelp recruitment (Vasquez & Santelices 1990). The effects of kelp removal on the kelp itself therefore appears to be an interaction between the severity, extent and frequency of the cutting.

Of interest in the present study is the change in densities of the two kelp species at Atlas Bay, from *Ecklonia*-dominated to *Laminaria*-dominated. Levitt *et al*. (2002) noted that reduction in density of *Ecklonia* recruits in the recovering population was likely controlled by grazers (Fricke 1979) or the availability of substratum (Anderson *et al.* 1997), rather than intraspecific competition (see also Rothman 2006). In Namibia, where grazer abundances are low, it is likely that either the availability of primary space, or limiting light in the turbid waters (especially at South Jetty) may be the most important factors determining recruitment success of *Ecklonia*. The increase in the contribution of *Laminaria* at both the South Jetty site and Atlas Bay suggests that removal of the *Ecklonia* canopy resulted in rapid growth of *Laminaria* recruits, which had most likely been present as a remnant stand in the sub-canopy.

Laminaria pallida sporophytes have a transient distribution being extremely common after the breeding season in autumn and winter (Dieckmann 1978; Velimirov & Griffiths 1979). Growth is highly seasonal starting in July and peaking in October/November, when elongation rates of up to 10 mm per day have been recorded from the Lüderitz area, and 13 mm per day for the Southern Benguela (Dieckmann 1980). Although recruits are often exterminated over much of the area by competition for space or light, or by grazing (in the southern Benguela) (Velimirov & Griffiths 1979), this species has been described as having invasive capabilities (Simons & Jarman 1981). Being at the northern limit of its distribution, and not displaying regular recruitment patterns *Ecklonia maxima* could therefore easily be outcompeted by *Laminaria* when kelp removal coincides with periods of low *Ecklonia* recruitment.

The results of our study, and those those of previous researchers working in the Benguela region (Simons & Jarman 1981; Molloy 1992; Levitt *et al*. 2002; Rothman 2006), have shown that recovery of kelp density and biomass, after experimental and commercial removal, is relatively rapid and attributable primarily to remnant stands of sub-canopy juveniles. Although canopy development may take longer, recovery to pre-harvest biomass levels occurs within two years. Contribution of the two dominant species to the overall recovering kelp density and biomass, however, appears to vary spatially (on scales of meters to kilometers), depending on the availability of primary space, the degree of wave exposure and light limitations in the naturally turbid waters. These physical factors, together with the life history and phenological attributes of the two kelp species appear to play important roles in determining recruitment success, thereby regulating much of the subsequent succession and community structure of the kelp bed.

4.2 The Effects of Kelp Cutting on Benthic Communities

Biogeographically the study area falls into the cool temperate Namaqua Province, which extends from Cape Point up to Lüderitz (Emanuel *et al.* 1992), the intertidal and subtidal ecology of which is largely shaped by the influence of the Benguela upwelling system (Shannon 1985). The intertidal and shallow subtidal benthic communities are in essence ubiquitous throughout the biogeographic province, varying only with wave exposure and substrate availability. In general, sheltered shores are diverse with a relatively low biomass and conversely exposed and semi-exposed shores harbour a high biomass with lower species diversity (Bustamante *et al.* 1997). In the sublittoral habitat, the biological communities can be broadly grouped into an algal-dominated inshore zone (from the supralittoral fringe to a depth of ~10 m), and a faunal-dominated offshore zone (below 10 m depth). The shift between these is not knife-edge, however, representing instead a continuum of species distributions, merely with changing abundances (Velimirov *et al*. 1977).

The community structure of the subtidal benthos in the Elizabeth Bay area is typical of the West Coast kelp bed environment, with a high kelp biomass and an associated understory community (*Field et al*. 1977; Velimirov *et al*. 1977; Velimirov & Griffiths 1979; Barkai & Branch 1988). Possibly the most notable difference to the communities described by these authors for the southern Benguela region, is the low diversity and abundance of subtidal grazers off southern Namibia. 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 and sheltered habitats for the diverse benthic community. Removal of large areas of kelp bed may therefore have important consequences for the structure and functioning of this community.

As both the size and demography of the kelp forest can be expected to be different after a disturbance such as canopy harvesting or removal of the whole plant, the associated community will also be affected. Possibly the most noticeable immediate effect of the removal of the kelp canopy in this study was the bleaching of the often expansive crusts of encrusting coralline algae that occur in the immediate subtidal zone, particularly at Wolf Bay (see Figure 17, pg 37). This has similarly been reported by Irving *et al.* (2004) and Irving *et al.* (2005), who found that canopy loss causes an increase by three orders of magnitude in the intensity of photosynthetically active radiation and ultra-violet radiation reaching the substratum. This results in bleaching of the coralline crusts and a concomitant rapid reduction (up to 50%) in photosynthetic activity. Despite the bleaching, however, the encrusting corallines are able to maintain reduced levels of photosynthetic activity, explaining why under certain conditions bleached corallines can persist in the absence of canopy-forming algae. These authors concluded that the positive association between canopy-forming algae and encrusting coralline algae is maintained because of shade provided by the canopy.

More recently, however, Irving & Connell (2006) demonstrated that abrasion by *Ecklonia radiata* laminae significantly reduces the abundance of articulate corallines occurring under the canopy. The process of laminae 'sweeping' across the substratum has been demonstrated to exclude understory floral and faunal taxa susceptible to such physical disturbance (Velimirov & Griffiths 1979; Kennelly 1989; Jenkins *et al*. 1999; Leonard 1999; River & Edmunds 2001; Connell 2003b; Konar & Estes 2003). However, not all canopies abrade the substratum, as differences in the abrasion intensity depends both on morphological differences among canopy-forming species, as well as differences in the densities and regional variation in the morphology of the key species that exert the abrasive effects. For example, Irving & Connell (2006) found that *E. radiata* in southern Australia were morphologically more suitable for exerting strong abrasion effects as they had shorter stipes and longer laminae relative to the those occurring in eastern Australia, where abrasion was reported to have negligible effects on understory communities (Kennelly 1989). Furthermore, the negative effects of laminar abrasion appears to be greater beneath monospecific canopies than mixed species canopies.

In the present study, the shallow subtidal at Wolf Bay was characterised by a comparatively high cover of crustose corallines, and low cover of foliose understory algae thereby forming a distinct cluster in the community analyses (Figure 18, pg 39). The site is also distinguishable by its significantly higher proportion of *Laminaria* plants (93%) than occurred at the other two sites. In his study of growth morphology of *Laminaria pallida* in different environments near Lüderitz, Molloy (1992) reported that stipe lengths and frond lengths are proportionally shortest in the 0-0.5 m depth range. This is also the depth at which coralline crusts are most abundant (pers. obs.). As removal of the canopy resulted in proliferation of red and green foliose seaweeds in crustose coraline dominated areas, it seems likely that *Laminaria* fronds have a substantial abrasive effect on the benthos (Figure 17, pg 37). Studies conducted in other parts of the world have shown that the effects of abrasion in combination with the shade provided by foliose algal canopies, are responsible for the maintenance of extensive covers of encrusting coralline algae (Kennelly 1989; Figueiredo *et al.* 2000; Connell 2003a). Despite initial bleaching of coralline crusts observed during our study, the overall cover of crustose coralline algae did not change substantially after the removal of kelp. This suggests that bleaching is relatively short-term, and that the coralline crusts recover comparatively quickly, as kelp canopy-shading is replaced by shading by the rapidly proliferating foliose red and green understory algae.

Enhanced irradiance as a consequence of canopy removal benefits many species of opportunistic green and red foliose algae in shallow subtidal areas (Vasquez & Santelices 1990; Clark *et al.* 2004). Where grazer densities are low, as is the case in Namibia, this leads to a proliferation of these opportunists who may subsequently play important roles in the succession of kelp forest restoration (Dayton *et al*. 1984, 1992). In cases where sub-canopy species are also removed, and/or remnant stands of recruits are absent at the time of canopy removal, this can lead to the temporary establishment of a diversity of ephemeral and/or perennial understory species (Dayton 1985; Schiel 1988; Vasquez & Santelices 1990; Clark *et al.* 2004). Temporal variability in competitive interactions between these understory species has been found to further increase the overall variation in kelp forest community composition (Clark *et al.* 2004).

These mixed stands are usually short-lived, however, with significant abundance changes usually occurring within a year after harvesting as the dominant kelps and perennial species re-establish (Sharp & Pringle 1990).

Simons & Jarman (1981) reported that in areas where *Ecklonia maxima* had been completely cleared from the substrate, non-kelp algal species dominated for at least 12 months. Changes in algal diversity following canopy removal were not consistent though, with an increase in diversity following cutting being recorded at one site, while a decline in diversity occurred at another (Levitt *et al*. 2002). This may, however, have been a function of different grazer densities at the two sites, although the authors do not comment on this. Clark *et al*. (2004a) reported that removal of the dominant kelp canopies lead to complex responses in the benthic light regimes and corresponding responses in the benthic algal communities.

Similarly, in California where the *Macrocystis* canopy is cut annually to a depth of 1.2 m, but where the sub-canopy of *Pterygophora californica* is left in tact, the growth of foliose reds (and recruitment of kelps) is greatly reduced even in the absence of the dominant kelp, due to light reduction by the sub-canopy (Kimura & Foster 1984; Clark *et al.* 2004a). Consequently, no differences in plant abundances in the understory vegetation were observed one year after harvesting, compared to adjacent uncut areas (North 1957). Clark *et al.* (2004a) report that significant responses in red algal bottom species were only noticeable two years after the canopies were initially cleared. These authors commented that grouping algae into the commonly used “foliose red” functional group may obscure the various responses of individual red algae to canopy removal. This may explain why significant differences in cover of understory algae following canopy removal are often not detected (North 1957; Reed & Foster 1984; Sharp & Pringle 1990; this study). Repetitive cutting for up to two years, however, resulted in increased mortality of the target kelps (North 1957; Miller & Geibel 1973). Repeated cutting reduces the efficiency of the holdfast due to decreased hapteral growth leading to uprooting during harvesting (McCleneghan & Houk 1985). Mortality of adult plants, or removal of the sub-canopy, in turn resulted in both higher juvenile densities as well as higher cover of red foliose understory algae (Hernández-Carmona 1996; Clark *et al.* 2004a).

As the age structure of the re-growing kelp forest tends to be more homogenous than in natural forests, being dominated by younger plants, this will influence the epiphytic community composition in particular, as this is affected by the age of the host plants (Whittick 1983). Christie *et al.* (1998), established that in harvested Norwegian kelp beds, epiphytes may take more than six years to recover to pre-harvest levels, as their recovery is severely hampered by shading of the developing kelp community. In the case of repeated re-harvesting of an area, recovery of the abundance and diversity of epiphytes may in fact never occur. This may have subtle effects on the understory ecology. Over 50 species have been recorded as growing on the stipes, fronds and holdfasts of *E. maxima* making up between 409% of the biomass of the kelp itself (Allen & Griffiths 1981; Stegenga *et al.* 1997). Of the epiphytic macroalgae, *Carpoblepharis flaccida*, *Polysiphonia virgata* and *Gelidium vittatum* sustain many herbivorous fish, amphipods and isopods (Allen & Griffiths 1981; Field & Griffiths 1991; Anderson *et al.* 1997), and are thus important to the energy flow within the kelp bed habitat.

Diversity of kelp forest fauna, particularly that associated with the holdfasts, is well documented (Field *et al.* 1977; Velimirov *et al*. 1977; Norderhaug *et al.* 2002; Christie *et al.* 1998; Waage-Nielsen *et al.* 2003, amongst others). The holdfasts of *Ecklonia maxima* are also important as a refuge for kelp recruits (Anderson *et al.* 1997). Any harvesting or cutting of canopy-forming macrophytes, which ultimately leads to the loss of the holdfast can therefore be expected to impact the structure of the associated faunal community. An extreme example of this is the harvesting of two *Lessonia* species in central Chile, which are gathered by cutting off the frond above the first dichotomy or just above the holdfast (Vasquez & Santelices 1990). This resulted in 50-60% mortality of the plants six months after cutting, and total loss after 8-10 months. Cutting is therefore equivalent to the delayed removal of the entire plant. Although cutting did not affect the invertebrate benthos inhabiting the holdfast, this community was ultimately lost on mortality of the plant (Vasquez & Santelices 1990). Furthermore, the authors established that plant mortality was primarily due to the development of a heavy epiphyte load on the remaining stipes, and, where the holdfast was lost, predation by grazers due to increased interplant distances. High grazer densities persisted for many years inhibiting juvenile kelp recruitment, and the resulting barren ground was found to last for over a decade if the grazers were not removed.

In their study of recolonisation of artificial holdfasts, Waage-Nielsen *et al*. (2003) established that the recovery rate of benthos associated with kelp bed disturbance depends not only on the size of the cleared area, but also on the presence and abundance of refuges (such as remaining holdfasts/plants), and the presence and abundance of alternative habitats such as boulder fields. Most faunal species associated with the kelp forest were able to disperse rapidly into cleared areas after harvesting, although the rate of colonisation varied between species/taxa, probably being directly related to their dispersal abilities. Polychaetes and gastropods colonised to pre-harvest levels within one year, while others took longer to attain maximum abundance (Christie *et al.* 1998). As would be expected, the diversity and abundance of colonisers increased with length of exposure period (see also Dean & Connell 1987), taking up to six years to attain the same levels as those of a natural forest. Interestingly, crabs failed to recovery completely over the same period, possibly as a result of the smaller holdfasts of younger plants and therefore incomplete recovery of their preferred habitat (Christie *et al*. 1998; Waage-Nielsen *et al.* 2003). What must always be kept in mind, however, is that recovery patterns of communities in kelp forests are likely to vary spatially, especially in canopy-forming species whose distribution covers a wide latitudinal range (Sjøtun *et al*. 1993).

In the case of *Ecklonia maxima*, which is harvested by cutting just above the holdfast, the remaining holdfast also eventually dies (Levitt *et al*. 2002). These authors found that the faunal community structure, however, did not differ between harvested and uncut areas, and the species richness remained relatively constant in these areas. Consequently, they concluded that the fauna communities associated with *E. maxima* were less susceptible to disturbance caused by harvesting than the understory flora, the latter responding to changes in the light regime. The rapid proliferation of red and green understory algae observed during our study after kelp-canopy removal confirm this (Figure 17, pg 37).

The degree of impact of kelp removal on the macrofauna is possibly related to the recovery rate of the kelp canopy. In areas where the canopy, or sub-canopy recover within 1-3 years, the effects on the benthos are probably not significant. However, under high exploitation levels and/or repeated harvesting, where recovery to pre-harvest levels takes longer, significant changes in filter-feeder and grazer densities can be expected (see Sharp & Pringle 1990). This would explain the anecdotal evidence from the Namaqualand coast, where repeated cutting of kelp by diamond divers apparently resulted in the elimination of kelp and the associated understory flora from a small bay, the benthos instead becoming dominated by the reef-building, filter-feeding polychaete *Gunnarea capensis* (G. Koeglenberg, West Coast Lobster Sea Management Association, pers. comm.).

Other community effects mentioned here briefly for the sake of completeness include decreased production potentially available as food to marine organisms, and increased phytoplankton production on removal of the shading effect of the canopy (Borchers & Field 1981). It has been estimated that only about 1% of the kelp biomass is consumed by grazers (e.g. limpets, Turbos, abalone), which feed either directly on the kelp plants or trap the fronds underfoot. Of the kelp production entering the food chain, 85% is lost as mucus or broken off as fragments (Newell *et al*. 1980). Of this between 5-20% is consumed by deposit feeding urchins (Fricke 1979; Branch & Branch 1981; Buxton & Field 1983), or is cast up on the beach where it supports large populations of scavenging sandhoppers and isopods. Both the particulate matter and the mucus form important food sources for bacteria and contribute the major energy source for filter-feeders, many of which in turn are eaten by kelp bed carnivores such as rock lobsters. Large-scale loss of kelp could therefore have cascade effects throughout the inshore ecosystem.

In the Benguela ecosystem, phytoplankton patches formed during upwelling events, are transported shorewards when the prevailing southerly winds relax (Andrew & Hutchings 1980; Field *et al*. 1980). Borchers & Field (1981) found that there was a decline in phytoplankton production shorewards as the shading effect of the kelp bed increased. It was determined that phytoplankton production under the kelp canopy was greatly reduced, being as little as 1-16% of that recorded in open kelp-free areas. The *Ecklonia* canopy was found to reduce phytoplankton production by ~95%, while figures were less under *Laminaria*. Surface canopies of kelps are reported to reduce bottom light by up to <1% of surface intensity (Norton *et al.* 1977; Kimura & Foster 1984; Reed & Foster 1984; Kennelly 1989; Schiel & Nelson 1990; Schroeter *et al*. 1995; Edwards 1998; Clark *et al.* 2004a; Rothman 2006). Large-scale loss of the kelp canopy would therefore increase irradiation, consequently promoting phytoplankton production. The effect is likely to be short-term, however, as increased nutrients and light levels will in turn enhance the recruitment and growth rate of juvenile kelps leading to the recovery of the shading canopy.

In summary, several responses of the benthic communities were evident from the experiment on kelp cutting. There was an initial, rapid bleaching of coralline algal crusts. This effect is likely to be short-term, however, as removal of the kelp canopy resulted in a proliferation of foliose red and ephemeral green algae, which formed a dense secondary canopy. The faunal community structure remained relatively constant in areas were kelp had been removed, significant differences in the cover of filter-feeders being recorded at only one of the three study sites. As the holdfasts in cut areas will ultimately be lost, however, this is likely to affect the associated benthic communities. Additional effects which are likely to occur include significant changes in filter-feeder and grazer densities under conditions of repeated cutting, and increased phytoplankton production in cleared areas.

4.3 The Effects of Kelp Cutting on Rock Lobster Abundance and Distribution

Besides providing shelter for a wide diversity of marine flora and fauna, kelp beds in the central and southern Benguela region are known to serve as an essential nursery area for rock lobster and several fish species (Velimirov *et al.* 1977; Velimirov & Griffiths 1979; Carr 1989, 1994). On the southern Namibian coastline, in particular, the nearshore reefs and kelp beds are thought to play an important role in rock lobster post-puerulus settlement and juvenile recruitment (Tomalin 1995, 1996). In Namibia the depth distribution and availability of rock lobsters is strongly influenced by environmental conditions (Newman & Pollock 1971; Pollock 1978; Beyers 1979; Pollock & Beyers 1981; Bailey *et al*. 1985; Pollock & Shannon 1987; Tomalin 1993, amongst others). In the winter months the lobsters occur in deep waters, possibly seeking shelter from winter swells, or to feed and to release larvae in deeper waters (Pollock & Shannon 1987; Noli & Grobler 1998). During the summer months of upwelling (January to April) the lobsters migrate inshore again in response to the intrusion onto the shelf off low-oxygen water. This inshore migration and concentration of lobsters in shallower, better-oxygenated water coincides with the commercial fishing season (Noli & Grobler 1998). During periods of offshore hypoxia, kelp beds therefore provide an important shallow-water refuge and food source for lobsters. Consequently, any large-scale reduction or loss of these habitats may negatively affect the long-term sustainability of rock lobster resources in the area.

Kelp beds also play an important role in coastal oceanography by exerting an important breakwater effect for inshore, shallow water communities (Field *et al*. 1977; Velimirov *et al*. 1977; Løvås & Tørum 2001). The energy of the incoming swells is dissipated as they reach the edge of the kelp bed, and currents, particularly within *Laminaria* beds, are reduced relative to immediately above them (Kitching *et al*. 1934; Løvås & Tørum 2001). Bottom currents in shallow water are reduced by almost a third through bottom drag, and internal waves in cross-shore movements are dampened (Jackson 1983; Jackson & Winant 1983; Jackson 1984). Increased drag around holdfasts and stipes slows currents over a distance of about 100 m, thereby causing most of the water to flow around the kelp stand. The combination of increasing offshore depth and high kelp drag causes large current gradients, with flow being reduced by up to 75% inside the kelp bed (Jackson 1983). As lobsters are known to cease feeding in extremely turbulent water, the calmer conditions provided by kelp beds would therefore serve as a refuge not only from hypoxic conditions, but also as a shelter under high swell conditions.

In a large kelp stand, this reduced water motion and turbulence affects both the supply of nutrients and larvae to the kelp bed ecosystem. In addition to the sharp spatial gradient in shade, the resultant edge effect represents an abrupt transition from the pelagic realm and may thus have important chemical and physical cues that initiate settling behaviour in competent larvae (Dayton 1985).

The spatial and temporal variability in rock lobster abundance at the three study sites is not unexpected, and is likely attributable to a plethora of affects including differences between the sites in sediment composition and seabed structure, degree of exposure, benthic community structure, underwater visibility (affecting diver counts), amongst others. While it may be argued that removal of kelp reduces potential refugia and foraging substrata for rock lobsters (see also Bodkin 1988), the results of this study yielded no evidence that cutting kelp has any negative effect on rock lobster densities - at least at the local scale at which the experiment was conducted. Indeed, at Atlas Bay and Wolf Bay results suggested that lobsters may prefer areas where the thick kelp cover has been removed. The proliferation of red and green foliose algae in the cut lanes formed a thick algal ‘carpet’, which provided excellent shelter for small rock lobsters, which were often seen crouching under the dense fronds in reef areas where crevices are sparse. At the Jetty site, where the seabed has more structure than at the other two sites (see Section 4.5 below), rock lobsters had adequate shelter in both the cut and uncut areas and differences in densities between the treatments there were not significant. The apparent preference of lobsters for areas where the *Laminaria*-dominant canopy has been removed, may, however, also indicate an avoidance of the abrasive action of the kelp fronds. Laminar ‘sweeping’ has been demonstrated to have strong negative effects on the abundance of mobile invertebrates (e.g. holothurians and urchins: Velimirov & Griffiths 1979; urchins: Konar & Estes 2003) and the recruitment and survival of sessile invertebrates (e.g. barnacles, serpulid and spirorbid polychaetes: Jenkins *et al.* 1999; Leonard 1999; River & Edmunds 2001; Connell 2003b), and in inshore areas in Namibia where *Laminaria* densities are high, may well also negatively affect rock lobster abundances. This would also help explain why rock lobster densities are lower at Atlas Bay and Wolf Bay, these sites being characterised by relatively flat featureless seabeds and high *Laminaria* densities. The sites may therefore represent sub-optimal habitats as rock lobsters would have limited shelter from the sweeping action of kelp fronds.

Our results should, however, be interpreted with caution for two reasons. Firstly, differences among the sites indicate that the effects of kelp-cutting may be context-dependent. Secondly, the scale at which the kelp-cutting was done was probably not large enough to eliminate the break-water effect of kelps. Removal of kelps on a scale large enough to accomplish this may well yield different results. Nevertheless, on face value our results yielded no evidence that kelp-cutting is detrimental to the numbers of adult lobsters.

In their study on harvesting of *Ecklonia maxima* at Kommetjie in the Southern Benguela Region, Simons & Jarman (1981) mention anecdotal reports that rock lobster catches decreased in the cut areas. These reports were unsubstantiated, however, and possibly nothing more than public resistance to the initiation of commercial kelp harvesting in the area. Sharp & Pringle (1990), for example, noted that the harvest areas of irish moss (*Chondrus crispus*) in the western Prince Edward Islands (Gulf of St. Lawrence) were coincidental with large lobster fisheries. These areas had enjoyed extremely successful fisheries despite decades of repetitive harvesting, with catches in harvested areas increasing by 86% over a ten year period. In contrast, lobster catches in areas that were not harvested had increased by only 34% over the same period, although fishing effort remained stable.

4.4 The Effects of Kelp Cutting on Rock Lobster Puerulus Settlement

Like that of many lobster species, the life cycle of *Jasus lalandii* is relatively complicated. Adults usually move offshore to copulate in deeper waters shortly after moulting. In Namibia, egg-bearing females tend to move back into shallow water between May and July (Pollock & Beyers 1979; Beyers & Wilke 1990), where they release their eggs in early spring (September - November). The timing and magnitude of spawning in *J. lalandii* off the Namibian coast are most likely determined by environmental factors. The eggs hatch into planktic phyllosoma larvae, which are transported by offshore currents into the South Atlantic Gyre for periods of up to 12 months. After circulating in oceanic waters and undergoing 11 moults, the larvae are transported back onto the continental shelf break where they metamorphose into the post-larval puerulus phase once sufficient internal energy reserve levels are attained (McWilliam & Phillips 1997). In the absence of sufficient food or unfavourable environmental conditions, metamorphosis may, however, be delayed resulting in a slightly extended larval cycle and a shift in the time of peak puerulus settlement (Pollock 1989, 1990; Lemmens 1994). Pueruli are capable of swimming actively, and with the help of wind-induced cross-shelf currents, enter the inshore zone where they settle along the rocky coast in kelp beds and amongst benthic organisms such as sponges, mussels and various red algae species (Pollock 1987), and subsequently metamorphose into juveniles. The puerulus stage typically lasts 2-3 weeks (Lemmens 1994; Grobler, MFMR, *pers. comm*. 2006), metamorphosis being determined by the amount of stored energy, water temperature and the availability of suitable settlement habitat (Phillips & Booth 1994). Subsequent growth is influenced by temperature and food type (Haupt *et al*. 2006). Along the southern Namibian coastline, kelp beds and the structurally complex habitat they provide, are thought to play an important role in the settlement of pueruli, as sandy and muddy seabeds are avoided by both pueruli and juveniles (Keulder 2005). The availability of suitable habitat, and the necessary chemical and physical cues that initiate settling behaviour, are important for the survival and successful settlement of pueruli (Booth 1979; Herrnkind & Butler 1986; Pollock 1987; Phillips & Booth 1994; Lipcius & Eggleston 2000). Post-settlement processes may significantly disrupt the relationship between puerulus supply and adult population (Butler & Herrnkind 2000). In the central and southern Benguela, the processes that regulate the abundance of post-settled lobster are, however, poorly understood.

Various "collectors" have been developed to monitor puerulus settlement patterns in coastal waters (Heydorn 1969; Pollock 1973; Booth 1979; Phillips & Booth 1994). The designs were based on the association each species had with their preferred settlement substrates, as well as the habitat in which juveniles predominantly occur. Since the juveniles of spiny lobster species inhabit shallow waters, most of the collector designs (e.g. Booth crevice, Witham, and hogshair collectors) have been installed in coastal waters less than 15 m deep (Phillips & Booth 1994). The principle objective behind the development of the various collector designs was to gain a better understanding of the relationship between puerulus settlement and the abundance of juveniles and catch, for the purpose of future stock management predictions (Cruz *et al.* 1995; Phillips *et al.* 1994; Guzmán-Del Próo *et al.* 1996; Addison 1997; Caputi *et al.* 1997, Booth *et al*. 1999; Booth 2001; Cruz *et al.* 2001;Gardener *et al.* 2001;Yeung *et al.* 2001; Phillips *et al.* 2003; Melville-Smith *et al.* 2004). Indices of puerulus recruitment also serve as an indicator of declines in the biomass of spawning populations (Hilborn & Walters 1992). Such studies, however, require data series of at least 5 to 7 years if future "good" and "poor" years of catch, and declines in spawning biomass of commercial lobster species are to be predicted from puerulus abundance with any confidence (Booth & Kittaka 1994; Caputi *et al.* 1997; Phillips *et al.* 2000a, 2000b; Gardener *et al.* 2001). The high numbers of pueruli collected at some research sites has led researchers to investigate the viability of growing out wild-caught pueruli to marketable size on a commercial scale, thereby increasing lobster supply to the market (Chittleborough 1974, 1975; Booth & Kittaka 1994; Gardner *et al.* 2000; Greengrass 2007). Harvesting of early stage lobster for aquaculture could have significant implications for lobster populations and the management of the fishery.

Compared to the extensive research into long-term monitoring of pueruli settlement patterns conducted in New Zealand (Booth *et al.* 2001a; Booth *et al.* 2001b) and

Australia (Caputi *et al.* 1997; Gardner *et al.* 2001), knowledge of puerulus settlement and recruitment in the southern and central Benguela region is relatively sparse. Pollock (1973) reported seasonal patterns in the occurrence of *Jasus lalandii* phyllosoma larvae and pueruli and investigated the growth rate of puerulus and post-puerulus stages in this species. Peaks in puerulus abundance were reported to occur between December and April (Lazarus 1967; Heydorn 1969; Pollock 1973). Booth crevice collectors were first used for *J. lalandii* by Hazel *et al.* (2001) at various locations around the Cape Peninsula, but captured almost no pueruli. In 1999 the MFMR in Lüderitz launched a long-term program monitoring puerulus settlement on oyster crates in the bay in the hope this could be used as an indicator of future recruitment into the Lüderitz-based fishery, and to investigate the harvest of pueruli for aquaculture (Grobler & Ndjaula 2001). A study by Keulder (2005) was subsequently implemented as an extension of the MFMR's long-term puerulus monitoring program. During this project, puerulus settlement of *Jasus lalandii* was monitored on both oyster culture rafts as well as Booth crevice collectors in Lüderitz Bay, and puerulus settlement was correlated with various environmental factors (sea surface temperature, wind stress and lunar cycles).

In her study in Lüderitz, Keulder (2005) found a seasonal trend in puerulus and juvenile lobster abundance. Recruitment of transparent pueruli starts to increase from June with peak settlement during August-September. From October onwards the abundance of transparent pueruli declines again, with the lowest recruitment occurring between December and May. In Saldanha Bay, Greengrass (2007) recorded settlement later in the year, peaking in November-January. Average catches from collectors in Lüderitz was 12 pueruli/collector, although with maximum mean catches of between 18.8-21.8 per collector being recorded during the peak settlement period between August and October (Keulder 2005). Greengrass (2007) noted comparable settlement densities in Saldanha Bay. Both sets of values are similar to average puerulus catch rates reported for *J. edwardsii* (15-20 puerulus per Booth collector), during peak settlement season in Tasmania, where Booth crevice collectors were also employed (Phillips *et al.* 2003). The catches made during the present study were, however, significantly lower, with a maximum mean catch of only 5 pueruli/collector being made at the South Jetty site during April and May 2006, and September 2005 and 2006. It should be pointed out, however, that this data set is not as comprehensive as that of Keulder (2005), who monitored settlement every two weeks. It does, however, support the suggestion by Keulder (2005) that the bay at Lüderitz is a more productive habitat than juvenile nursery areas outside the bay.

Furthermore, Keulder (2005) established that peak puerulus settlement was related to below-average sea surface temperatures and moderate to strong south winds. No clear relationship was, however, established between peak puerulus abundance and moon phase. Although seasonality in settlement is relatively consistent, there is high inter-annual variability in settlement rates (see also Booth & Kittaka 2000; Jeffs & Hooker 2000), which in Namibia has been attributed by Keulder (2005) to variations in the strength of upwelling. Keulder (2005) also reports spatial variability in settlement rates, with settlement being significantly higher on collectors in close proximity to rocky outcrops and kelp beds with high density of juveniles. While the present data series is too incomplete and short to comment on inter-annual variability, significant spatial differences were observed, with settlement at the fines-impacted South Jetty site being significantly higher than at the other two sites.

During this study, juvenile recruitment on collectors was in most cases higher than that of pueruli, averaging 2.9 per collector across all sites, with a maximum of 4.3 per collector recorded at the South Jetty site, and peak abundances in January. Keulder (2005) in contrast, reports that recruitment of juvenile lobsters onto collectors was lower than for pueruli (maximum 7.32 juveniles/collector) usually lagging that of pueruli by two to three months. Juvenile abundance on collectors in Lüderitz Lagoon, peaked in November/December, but showed a second smaller peak during April (5 juveniles/collector). Depending on the water temperature, the puerulus stage lasts 2-3 weeks (Lemmens 1994; Grobler, MFMR, *pers. comm*. 2006) before metamorphosing into a juvenile; higher water temperatures increasing this processes. The higher juvenile numbers recorded in collectors during our study are thus most likely a consequence of the less frequent collector changes, with settled pueruli having already metamorphosed into juveniles whilst in the collector.

It is generally believed that pueruli and early-stage juvenile lobsters (<30 mm CL) are asocial, solitary animals (Butler 2000; MacDiarmid 2000; Yoshimura 2000; Hazell *et al.* 2001) that do not aggregate into groups. The results of our study and that of Keulder

(2005), however, suggest that high abundance of conspecifics in the surrounding natural habitat may enhances puerulus settlement at a collector site. Settlement was highest in collectors at the Jetty site, where abundances of juveniles and adults were also higher than at the other two sites.

Although kelp beds are no doubt important to the early life stages of rock lobsters, our results indicate that localised removal of kelp does not significantly affect puerulus settlement. Despite the absence of a shading canopy, the abundance of understory algae, refuges in the form of holdfasts and existing reef structure, and high juvenile kelp densities appear to provide sufficient settlement cues and shelter for settling pueruli. It is possible, however, that if kelp is repeatedly removed over large areas, thereby affecting the current regime and the structure of benthic communities in the shallow subtidal in the long-term, that puerulus settlement may also be affected. This would, however, require a large-scale removal experiment and long-term monitoring of puerulus settlement on collectors, and juvenile abundance in adjacent natural habitats, and is beyond the scope of this study.

4.5 The Effects of Seabed Topography on Kelp and Lobster Distribution

Physical factors that contribute to spatial variations in the abundance and distribution of marine benthic organisms at small and moderate spatial scales include wave exposure, natural disturbances (e.g. storm damage, floods, low oxygen events), water chemistry and substratum type (reviewed in Schoch & Dethier 1996). Consequently, changes in abundances of marine benthic organisms associated with habitats of various geo-morphological circumstances (Dethier 1990; Schoch 1994, 1996) and environmental conditions (Field *et al*. 1982; Clarke 1993; Clarke & Ainsworth 1993) conditions have been used to investigate disturbance of, and monitor changes in, biotic communities.

With very few exceptions, almost all kelps occur on hard substrata. Only large and old plants of *Laminaria sinclarii*, *L. saccharina* and *Macrocystis angustifolia* have been recorded growing on sandy bottoms (Markham 1973; North 1971; Neushul 1971). Mature populations of most other kelps, including *Ecklonia maxima* and *Laminaria pallida*, are however restricted to hard substrata. Field *et al*. (1980) in their study of numerous kelp beds along the south-western Cape Coast, reported that kelp growth is limited by transitions from rocky to sandy substratum (see also Rothman 2006).

The distribution of kelp beds on the southern African West Coast is largely determined by the coastal geomorphology. The bedrock of the inner shelf displays an irregular, erosion surface with relief of up to 15 m, generally with a thin cover of unconsolidated sediments of Orange River origin ranging in thickness from a few centimetres to tens of metres. Along the southern Namibian coastline, wave energy focused by powerful southerly winds transports the sands of the Orange River prodelta northwards, forming a predominantly sandy coast, which extends virtually continuously along the coastline till about 45 km to the north of the river mouth (Rogers 1977). Further north, the sand cover thins and increasing areas of bedrock in the nearshore region are devoid of sediment, and are exposed in the intertidal zone as scattered reefs extending offshore as shallow rocky platforms. In some areas these support small stands of kelp, while further south the reefs are strongly influenced by seasonal and/or ephemeral inundation by unconsolidated sediments. Although the duration of sand burial is variable, kelp rarely persists on such reefs, the intertidal flora of which is characterised by ephemeral algae such as *Ulva*, *Porphyra* and *Cladophora*, which rapidly colonise the bare rock on exposure. From ~95 km north of the mouth, the onshore coast is characterised by ridge and valley topography, which continues out to sea across the inner shelf as subdued submarine ridges, islands, exposed rocks and numerous small rocky headlands with north-facing bays (Murray *et al*. 1971). Depending on their exposure to wave action, the rocky outcrops and headland support variable densities of kelp, usually occurring as narrow, fringing beds dominated by *Laminaria*. In the lee of headlands, or in small bays where some shelter is provided from the prevailing swells, more extensive kelp beds occur, often including higher densities of *E. maxima.*

Sands of Orange River origin also dominate the inshore sediments of the Namaqualand coast, forming a virtually continuous, thinning sand wedge to ~250 km south of the Orange River mouth. The coastline between Alexander Bay and Port Nolloth is characterised by coastal cliffs, wave-cut platforms and exposed rocky headlands alternating with small beaches. Further south to Kleinzee the shoreline is dominated by sandy beaches with occasional rocky platforms only. Rocky shores dominate the coast for a further 250 km to the Olifants River mouth, although beaches backed by dune formations become progressively more frequent southwards. Between the Olifants River and Berg River mouths, the coastline is predominantly sandy. In all, about 48% of the coastline between Alexander Bay and Cape Columbine is composed of rocky shores (Lane & Carter 1999). Where reefs occur in this manner, they support kelp beds of variable extent, associated benthic communities and West Coast rock lobsters. In Namaqualand the kelp beds are denser than, but not as extensive as, those further south as they are comprised mainly of *Laminaria* (Rand 2006). Larger beds are found only in those areas where there are large expanses of shallow rocky reefs.

Beyers & Wilke (1990) pointed out that bottom topography, presence of bedrock and the degree of sedimentation over existing substratum influence the distribution, behaviour and availability of rock lobster off Namibia. A study conducted in 10-25 m depth at Zweispitz, just south of Lüderitz, where certain regions had been intensely mined by divers operating suction hoses for a period of over seven years, identified that the abundance and distribution patterns of benthic communities and rock lobsters were closely linked to biological and physical characteristics of the area (Pulfrich & Penney 2001). Significant associations were found between biota and seabed types, with rock and boulder habitats showing significant differences from sand and gravel areas, in terms of both the benthic community structure and rock lobster abundance. In particular, rock lobster were found to exhibit strong associations with, and preference for, creviced habitats, and clear avoidance of gravel and sand areas. This has been confirmed by data collected along diver transects in the Cape Peninsula National Park (Cape Peninsula National Park, unpublished data) (Figure 28), which clearly demonstrated preference by rock lobsters for rock and boulder habitats. The clear and strong trend in abundance of lobsters towards harder ground recorded during the recent Namibian rock lobster fisheries independent monitoring surveys (Pulfrich *et al*. 2006), is consistent with these observations.

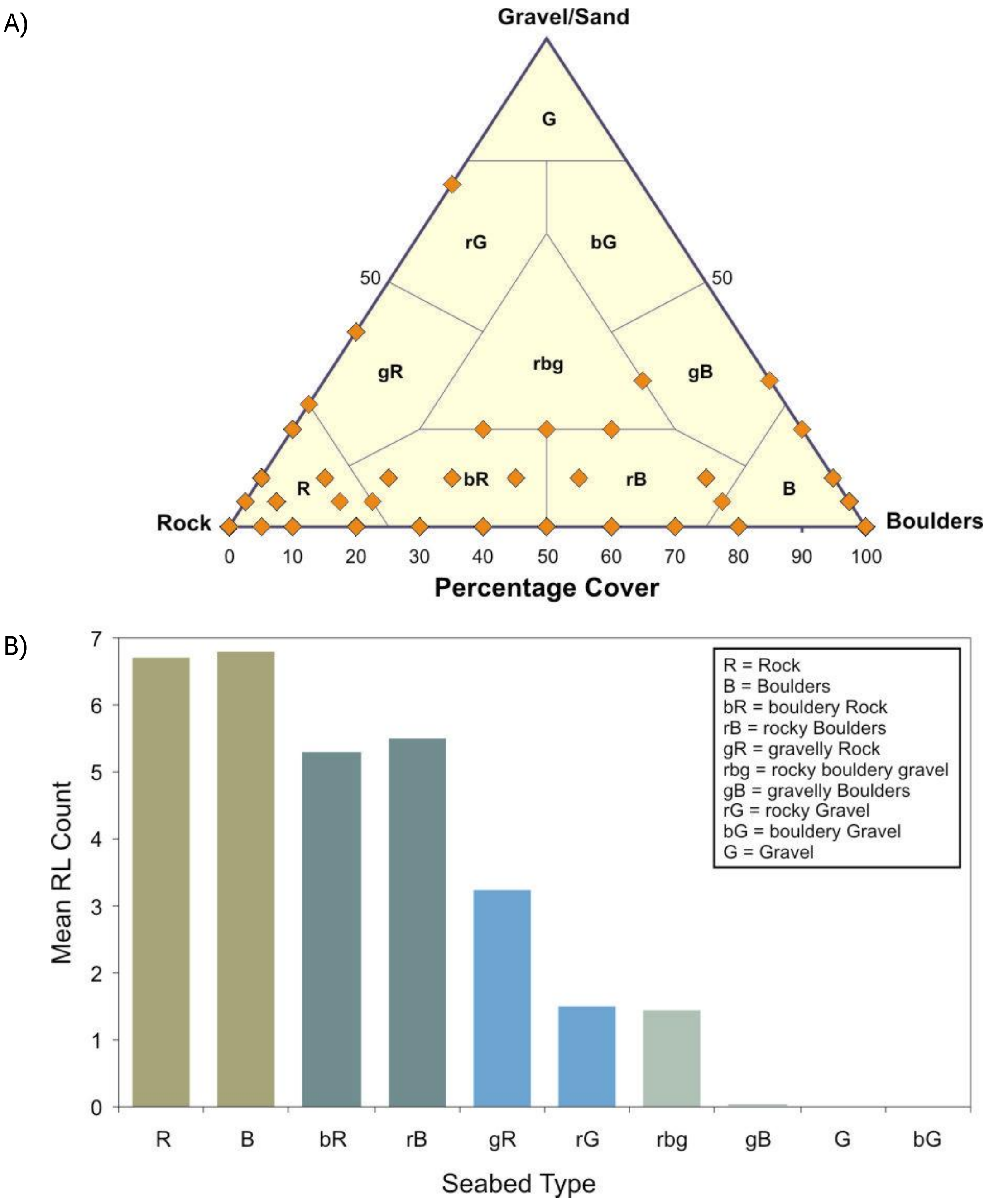


Figure 28. Seabed type associations of rock lobsters in the Cape Peninsula National Park. A) Ternary plot showing the distribution of lobsters (diamond symbols) in relation to proportions of rock, boulders and gravel/sand at 462 sites around the Cape Peninsula; B) Mean number of rock lobsters counted per site on the 10 seabed types delineated in the ternary plot (seabed types after Penney *et al.*, in prep.).

Preference by lobsters for certain seabed types has been widely discussed in the scientific literature (reviewed in Spanier 1994). The availability of shelter, in the form of highly structured reef or dense algal cover, was found to be a critical feature for all life cycle stages of the American clawed lobster, *Homarus americanus* (Cobb 1971; Wahle & Steneck 1992), with all but the large animals that have outgrown the risk of heavy predation. On the southern African West Coast, *J. lalandii* have been reported to migrate, often in very large numbers, across gravel and sandy areas in order to escape low oxygen water masses, but such migrations tend to be episodic and of short duration (Grobler, Ministry of Fisheries and Marine Resources; van Guysen, Windvogel Diamonds, pers. com.). Habitat selection in lobsters has been postulated to be the result of a complex interaction of factors including protective value of a shelter (from predators as well as detrimental abiotic forces such as strong surge), stimuli of con-specifics (Zimmer-Faust *et al*. 1985; Zimmer-Faust & Spanier 1987; Spanier 1994).

4.6 The Effects of Sediments on Benthic Communities

4.6.1 Natural Deposition

As part of a natural sedimentary cycle, the Namibian coastline is subject to gradual accumulation of sand deposits during summer, and subsequent beach erosion during winter. In other parts of the world, seasonal patterns of sand inundation have been shown to have superimposed on them bi-weekly, daily, and storm-associated sand movements, when temporary reversal in the sediment transport direction occurs in response to short-term changes in wave conditions (Daly & Mathieson 1977; Trowbridge 1996). The frequency and duration of sand burial resulting from such events are likely to be highly dependent on local topography (see also Daly & Mathieson 1977; Littler *et al.* 1983; D'Antonio 1986). Such natural short-term inundations are thought to be the reason for both the observed increase in unconsolidated sediments recorded subtidally at Atlas Bay and Wolf Bay during the current study, as well as the comparatively high proportions of sand recorded in the intertidal zone during the ongoing Elizabeth Bay monitoring study (Pulfrich 2006). However, at the South Jetty site, progressive inundation of the intertidal and subtidal reef habitats in Elizabeth Bay by discharge of mining-related sediments is superimposed on natural fluxes (Pulfrich 2006). In ongoing monitoring of rocky shore communities in the area between Bogenfels Arch and Chameis Bay, similar smothering effects of intertidal and subtidal reefs have been recorded as a consequence of severe beach accretion due to pocket beach mining operations (Clark *et al.* 2006) (see below).

Sediment movement and deposition rates are closely linked to water depth and flow speed (Hiscock 1983; Overnell & Young 1995). High levels of siltation but reduced motion of sediments may be expected in areas with low degrees of water movement. 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. As the effects of these physical variables on the benthos are different, a distinction between them should be made. The abrasive action of sand particles in suspension can cause scouring which may result in the removal of whole organisms or their parts (Van Tamelen 1996; Day *et al.* 2000). Sediment suspension also reduces light penetration thus affecting primary production of phytoplankton and seaweeds, and loads the water with inorganic suspended particles, which may affect the feeding and absorption efficiency of filter-feeders. 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 choice of settlement site (Hiscock 1983; Rodrýguez *et al.* 1993), and post-settlement survival (Hunt & Shebling 1997). The presence of suspended inorganic particles may therefore effect plants and animals directly or indirectly, either lethally or sublethally.

The effects of elevated levels of natural particulate inorganic matter (PIM) and depositions of sediment have been well studied, and are known to have marked, but relatively predictable effects in determining the composition and ecology of intertidal and shallow subtidal benthic reef communities (Zoutendyk & Duvenage 1989). It has been postulated by many authors that the distribution and diversity 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), as well as other environmental variables such as depth (light), substrate topography, wave exposure, consumption by grazers and interspecific competition. Engledow & Bolton (1994) ascertained that the diversity of intertidal seaweeds in Namibia is strongly affected by sedimentation if 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 (Daly & Mathieson 1977; Devinny & Volse 1978; Airoldi *et al.* 1995; Airoldi & Cinelli 1997; Airoldi & Virgilio 1998). For example, a number of southern African west-coast species are adapted to sand and may dominate the midlittoral region of sand-influenced rocky shores. *Aeodes orbitosa, Ulva* sp. and several species of *Gymnogongrus* have crustose holdfasts and tetrasporophytes, and are able to survive sand burial for several months (Trowbridge 1996; Stegenga *et al.* 1997). Alternatively, sediment-induced disturbance can lead to the co-existence of a diverse mosaic of sediment tolerant and sediment intolerant competitors for space (Littler 1980; Littler & Littler 1981; Seapy & Littler 1982; Littler *et al.* 1983; McQuaid & Dower 1990). In sediment influenced areas, species richness therefore appears to be controlled by the frequency, nature and scale of disturbance of the system by sedimentation (McQuaid & Dower 1990; Dethier 1984).

Macroalgae (including kelp) seem to be more extensively impacted by prolonged sediment smothering than by short-term deep sand burial (Marszalek 1981; Rogers 1990; Airoldi *et al*. 1996), and only in extreme conditions of sand movement may thalli be broken by abrasion or killed by prolonged burial (Daly & Mathieson 1977). Although playing important roles in maintaining habitat heterogeneity and determining the diversity of algal assemblages, sedimentation and scour are detrimental to kelp plants. The effects are most severe on spores, gametophytes and young plants, although susceptibility varies between species (Dayton *et al.* 1984; Dean & Deysher 1983; Devinny & Volse 1978). While kelp juveniles are tolerant of low light levels associated with elevated turbidity, even small amounts of sediment in combination with water movement can preclude the recruitment and establishment of kelps because scouring prevents spore settlement, and deposition smothers established germlings (Ebling *et al.* 1960; Foster 1975; Devinny & Volse 1978; Kendrick 1991; Chapman & Fletcher 2002; Airoldi 2003; Schiel *et al.* 2006). Sediments also interact with other factors such as light availability and grazing (Ebling *et al*. 1960; Foster 1975; Devinny & Volse 1978; Hawkins & Hartnoll 1983; Kendrick 1991; Vadas *et al.* 1992; Reed 2000). Schiel *et al.* (2006) reported that even a light dusting of sediment reduces attachment of newly settled germlings of major habitat-forming algae by up to 70%, while a complete cover of sediment prevents attachment altogether, and burial of settled germlings by sediment for even a few days can result in 100% mortality. Fucoid and laminarian algae can be particularly vulnerable to these processes during their microscopic post-settlement stages and early development (Amsler *et al.* 1992; Vadas *et al.* 1992; Airoldi 2003; Schiel *et al*. 2006). In *L. pallida* and *E. maxima* germling settlement occurs throughout the year (M. Rothman, Seaweed Unit: Marine & Coastal Management, pers. comm.), so even the ephemeral presence of sediments could have an influence on recruitment.

The amount of suspended sediment in water and the deposition rate thereof are also major physical factors limiting the distribution and abundance of benthic invertebrates. Shifting sediments and frequent sand inundation can 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 1989; Marshall & McQuaid 1993; Pulfrich *et al.* 2003a, 2003b), predators (Robles 1982), and either promoting or inhibiting the establishment of competitively dominant species (Berry 1982; Taylor & Littler 1982).

Effects of siltation and sediment movement on macrobenthic animals include abrasion (D'Antonio 1986; Slattery & Bockus 1997), smothering (Taylor & Littler 1982; Marshall & McQuaid 1989; Genovese & Witman 1999), and clogging of the respiratory and/or filtering apparatus (Reiswig 1971a, 1971b; Gerrodette & Flechsig 1979; Butler *et al.* 1995; Genovese & Witman 1999). Adult population are likely to survive severe siltation, especially in areas of moderate to high water movement, by resistance to burial (Taylor & Littler 1982; Littler *et al*. 1983), emigration to adjacent non-inundated areas (Littler *et al.* 1983), or adjustment of filtration rates (Griffiths & Griffiths 1987; Hawkins *et al*. 1999). Increased sedimentation may, however, affect the recruitment success of macrobenthic species by smothering the substrate and thereby reducing settlement stimuli and food supply for juveniles (Eggleston 1972).

Bivalves in particular show remarkable physiological adaptations to elevated suspended sediment concentrations. Suspended sediment effects on juvenile and adult bivalves occur mainly at the sublethal level with the predominant response being reduced filter-feeding efficiencies that generally occur at concentrations of about 100 mg/ℓ. Lethal effects are seen at much higher concentrations (>7 000 mg/ℓ) and with long-term (3 weeks) exposures. For bivalve egg stages, critical suspended sediment concentration range from 188 mg/ℓ for oysters to 1 000 mg/ℓ for burrowing clams and larval stages show no effects at suspended sediment concentrations <750 mg/ℓ (Clarke & Wilber 2000).

Crustaceans appear to be relatively resistant to lethal effects with 25% mortality rate reported at 10 000 mg/ℓ for >240 hr exposures (Clarke & Wilber 2000). Suspended sediment effects thus occur mainly at the sublethal level. Rock lobsters are important predators in algal bed habitats, and being highly mobile usually respond to increasing sand cover simply by emigration to adjacent unsanded areas, expanding back into formerly inundated regions once the sand retreats (see also Littler *et al.* 1983). Chronic siltation and/or sand inundation may, however, affect adult rock lobsters, as well as the post-larval settlement stage, through habitat degradation or loss, as well as reduction in the availability and suitability of food (Parrish & Polovina 1994, Hudon 1987, Wahle & Steneck 1991, 1992). Herrnkind *et al.* (1988) have shown that high siltation, both natural and man-induced, may be potentially deleterious to rock lobster recruitment. They found that despite extensive benthic algal growth, settlement of post-larval pueruli of the spiny lobster *Panilurus argus* was significantly reduced in severely silted habitats. Furthermore, as prey abundance was lower in silted algae, the juveniles of those pueruli that did manage to settle subsequently left the silted habitat to obtain adequate food in unsilted algal stands. Movement by lobsters searching for food or unsilted habitat was predicted to result in increased predation, thereby affecting recruitment success.

Some of the effects of sedimentation are indirect, however. Patellid limpets are intolerant of inundation (Marshall & McQuaid 1993), and their depletion in areas of inundation leads to a proliferation of algae. Reductions of the densities of grazers, with concomitant dominance by foliose algae, often typify rocky-shore communities disturbed by sediments (Littler & Murray 1975; Hawkins & Hartnoll 1983, Littler *et al.* 1983; Hockey & Bosman 1986; Eekhout *et al.* 1992; Pulfrich *et al.* 2003a; Pulfrich *et al.* 2003b). Inundation by sand may also promote siphonariid limpets, because they are not only tolerant of sand but inferior competitors with patellids (Marshall & McQuaid 1989; Branch *et al.* 1990).

4.6.2 Mining-Related Sediments

The magnitude of the effects of mining-related sediments on marine biotic communities is directly related to the scale of operations. These range from the small-scale and localised deposition of fines and oversize tailings by diver-assisted shore- and vessel-based operations (Plate 15), to annual discharges of millions of tons of sediments into the sea from land-based processing plants (Plate 16) (detailed in report BEHP/CEA/03/02: Pisces Environmental Services, in prep.). In South Africa, legislation prevents the discharge of process water from large gravel treatment plants directly into the sea, and consequently waste water resulting from the gravel processing is directed into slimes dams where solids settle out before the clean water is returned to the sea via a penstock system. In contrast, the fine tailings from the processing plants in Namibia are discharged as a sediment slurry through a pipeline directly into the sea. Annual total discharges from the five major plants on the southern Namibian coast currently stand at around 16 million tonnes.

Furthermore, seawall mining is practiced along a virtually uninterupted ~100-km long strip of coastline immediately to the north of the Orange River mouth. As a consequence, the shoreline has been physically and irreversibly altered to an artificially accreted shoreline backed by flooded excavations, overburden dumps and in places exposed bedrock area. The seaward slopes of the seawalls are highly vulnerable to wave erosion and despite constant maintenance during the mining operation, huge volumes of sand are perpetually washed away. Although the erosion rate of sand from seawalls depends on the orientation and seaward offset of the seawall, it is estimated that several million cubic metres of sand can be eroded from the seawalls during heavy seas (Smith *et al*. 2006). The eroded material is carried offshore by undertow and rip currents and tends to deposit in the nearshore zone down-current along the coast. On the South African West Coast beaches, the supplies of overburden and sand are insufficient to construct and maintain large-scale seawalls in the face of constant erosion by waves, and consequently large-scale seawall mining has not developed south of the Orange River mouth. Mining of beaches on a smaller scale to has only recently (since mid 1990s) proven to be feasible along the South African West Coast, using temporary cofferdams constructed from locally-sourced beach sands and rocks.

Plate 15. Typical shore-based contractor showing discard of fines across the intertidal and accumulation of tailings around the classifier. A vessel-based operation further offshore can also be seen.

Plate 16. Discharge of high volumes of sediment into the sea from a land-based gravel processing facility in southern Namibia.

Plate 17. An extreme example of oversize dumping around the classifier site above the high water mark.

Plate 18. Accumulation of overburden in a gully from a nearshore vessel-based operation.

These operations are comparatively short-term, however, as reserves are rapidly depleted and the contractors move to new areas. For all these reasons, the effects of diamond mining on sea life are more intense in Namibia than in South Africa and, as a result, most of the research into the effects of mining-related sediments on marine communities has been conducted in Namibia.

Studies on the discharge of fines and tailings from shore-and vessel-based diver operations, either across the intertidal zone or directly into the sea, have identified that the negative effects of these sediment discharges may be intense, but are extremely localised, being confined to a scale of a few 10s of metres (Barkai & Bergh 1992; Pulfrich & Penney 2001; Pulfrich *et al*. 2003a). Nevertheless, mining and sediment discharges reduces the species diversity and abundance of both intertidal and subtidal communities. During, or immediately after shore-based mining, the intertidal community becomes characterized by the near-disappearance of grazers, proliferation of fast-growing, opportunistic foliose algae and decreased cover of filter-feeders. High natural variability in benthic community structure, however, made it difficult to distinguish mining impacts even a short time after mining had ceased (Pulfrich *et al*. 2003a). Subtidally the effects of sediments discharged from shore-units are masked by rapid dilution and the natural redistribution of sediments by wave action. However, if shore-based operations discard oversize tailings above the high water mark, this precludes natural redistribution of the tailings material by waves, and results in physical degradation of the landscape if the mounds of sterile gravel are not actively removed (Barkai & Bergh 1992; Pulfrich *et al*. 2003a) (Plate 17).

Similarly, the dumping of the oversize tailings during diver-assisted vessel-based mining has been found not to have a significant negative long-term impact on nearshore biota (Pulfrich & Penney 2001). Benthic communities in areas affected by overburden dumping were, however, shown to differ significantly from those on adjacent unaffected reefs, as a result of the change in seabed type. In more sheltered gullies, in particular, accumulated overburden acts to trap quantities of particulate detritus, resulting in rapid accumulation of detritus feeders such as brittle stars and sea cucumbers. In contrast, in high-energy areas such as in the shallow subtidal zone of along the reef edge, re-distribution of overburden material into the mined-out gullies is rapid, and although initial scouring of the benthic communities occurs, recovery of impacted reefs within 1-2 years was observed (Pulfrich 1998; Pulfrich & Penney 2001).

Excessive and repetitive dumping on the same area may, however, preclude dispersion and thus induce persistent change by converting profiled reef habitat into small-boulder or gravel areas. This occurs primarily by in-filling of reef crevices and smothering of benthic organisms, thereby reducing both suitable habitat and food availability for lobsters and other consumers (Plate 18). While the spatial extent of such impacts are greater than those for shore-based discharges, they are nonetheless confined to scales below 100s of metres.

On a significantly larger scale, the discharge of sediments from the Elizabeth Bay Mine, has significantly affected the structure of the intertidal and subtidal benthic communities, both in the south and the north-west of the bay (Plates 19-22). The sediment effects are most noticeable intertidally through a reduction in algal diversity with only hardy species that can tolerate sand burial (encrusting corallines, *Aeodes orbitosa*) remaining. On the high shore, a reduction in grazer density has resulted in the development of a distinct community structure dominated by ephemeral foliose algae (primarily *Ulva* sp. and *Porphyra capensis*) (Pulfrich 2006) (Plates 21a & b). The influences of these mobile sediments on subtidal communities reflect to some extent those observed in the intertidal habitat. At impacted areas within Elizabeth Bay the covers of green foliose algae and filter-feeders such as encrusting sponges and the Cape reef worm (*Gunnarea capensis*) were significantly higher, whereas encrusting algae and red algae had a significantly lower cover than at reference sites such as Atlas Bay and Wolf Bay further north. This indicates that the discharges from the Elizabeth Bay mine have had a significant impact on the subtidal benthic communities and, as a consequence, their structure remains distinct from that at the reference sites.

Ten years after mining discharges commenced, total accretion in the western corner of Elizabeth Bay was 255 m seaward of the pre-mining 1990 shoreline, with significant changes in the nearshore profile being evident to -5 m depth (Smith *et al*. 2002). In the more exposed eastern corner, the beach had accreted by 370 m, and changes in the nearshore profile extended to -8 m depth, some 800 m offshore of the original shoreline (CSIR 2001a). It is predicted that the coarser sediments in the discharge will remain confined to the bay, and not impact seabed habitats further south or north.

Plate 19. Blanketing of the flat platform reef by sand in the south of Elizabeth Bay. Proliferation of ephemeral foliose algae can be seen.

Plate 20. Accreting sediments filling a deep gully in the south of Elizabeth Bay. Smothering of kelp in the shallow subtidal and establishment of reef worm colonies can be seen.

Plate 21a & b. North-western corner of Elizabeth Bay showing the encroaching beach. A proliferation of foliose ephemeral algae and diatoms is evident in the mid-shore, whereas an abundance of reef worm *Gunnarea capensis* in the low shore, and regression of kelp in the shallow subtidal, can also be seen.

Plate 22. The seaward side of the platform reef in the south of Elizabeth Bay showing the formation of a small beach by encroaching mine sediments. The patchy loss of mussels and scouring of foliose algae (bleached and eroded pink tips) can also be seen.

As an extension of coastal diamond mining activities on the southern Namibian coastline, some of the small Pocket Beaches in the area between Bogenfels Arch and Chameis Bay were identified for mining (Figure 29). The mining project went into full commission in January 2004 and of 12 pocket-beach sites, three will be mined progressively from south to north before the end of 2010. Mining operations involve conventional open-cast dry-mining as well as innovative wet-mining using a dredger behind a protective seawall strengthened with water-impenetrable panels (cut-off walls) inserted down to bedrock (Burke 2005). As a consequence of these mining operations, it is estimated that in the order of ~1.2 million tonnes of sediment will be discharged onto the beach from the treatment plant near Chameis Bay, with a further estimated annual discharged of 500 000 tonnes over a period of between 18 months and three years from operations near Bogenfels (CSIR 2001b; 2006). The increased sediment load in the nearshore environment as a result of discharges and seawall construction, has resulted in substantial beach accretion, and inundation of rocky shores by sand has occurred particularly to the north of the mining sites (Clark *et al.* 2006). Monitoring of rocky shore communities in the area from 2004 onwards, has provided the opportunity to observe progressive shoreline accretion, and the effects this has on the intertidal and shallow subtidal communities (Clark *et al*. 2004b, 2005, 2006).

Figure 29. Map of the area between Bogenfels Arch and Chameis Bay showing the Pocket Beaches and places mentioned in the text. Pocket beaches being mined are in italics (Site 2, Site 3/ 4 and Site 11/ 12).

Inundation of rocky shore communities by sand as a consequence of sediment discharges and seawall erosion first became noticeable at South Rock, a rocky outcrop immediately to the north of the seawall and processing plant discharge point, in 2005. Severe smothering of the rocky intertidal and shallow subtidal habitats had occurred, with a layer of sand at least 1 m thick having completely covered the shallow infratidal zone, filling gullies and intertidal rock pools. This accretion resulted in:

- Blanketing of the nearshore kelp bed habitat by sand, with abraded fronds and stipes protruding from the recently accreted beach. In some areas only eroded holdfasts remained (Plates 23-26).
- Severe scouring and polishing of *Scutellastra argenvillei* shells. Although patellid limpets are known to be intolerant of sand inundation (Marshall & McQuaid 1993), large individuals were still present between the mussel beds on the low-shore. Their shells, and the rock they were on were, however, free of any encrusting coralline algae (Plate 27).
- Smothering of extensive areas of the invasive mussels *Mytilus galloprovincialis* (Plates 28 & 29). This species is less tolerant to sand-cover than the indigenous mussel *Choromytilus meridionalis* (Griffiths *et al.* 1992), and becomes weakened or dies after prolonged burial. On exposure, large areas of the mussel bed are subsequently torn off the rocks by wave action (Plates 26 & 29).

A year later, accretion as a direct consequence of seawall erosion and sediment discharge onto the beach, had extended both southwards (Plate 30), and further northwards of the mining site, with inundation being recorded at sampling sites over 10 km, 20 km and 30 km northwards (Plates 31 - 33). Predictive beach accretion modeling conducted for the mining operation, however, estimated that acute accretion will be restricted to within 10 km of the target beaches (CSIR 2001b, 2004, 2005). While some of this sediment may therefore be the result of natural seasonal sediment fluxes, gradual and progressive inundation of rocky habitats and kelp beds as a result of pocket beach mining operations is unequivocal. The effects of sedimentation on rocky coasts are, however, notoriously temporally variable, changing with the sea-weather, the seasons and between years. Accumulation of sand in shallow-water environments can occur within weeks during periods of calm weather, with subsequent erosion occurring within days during storms. Despite the obvious accretion on rocky headlands to the north of the mining site, evidence of recent erosion was also seen (Plate 34). This was apparent from the strong smell of sulfur and remnants of anoxic sediments in small depressions in the rocks at some sites, and proliferation of fast-growing, opportunistic algae (particularly *Ulva*, *Cladophora* and *Porphyra* spp.) at others (Plates 35 & 36). Often considered as 'indicator species', the abundance of these ephemaral seaweeds was a sign that the reefs had recently become exposed after the (possibly storm-related) erosion of previously accreted sand.

Inundation of the rocky shores defining the northern and southern boundary of Pocket Beach Site 2, the first of the beaches to be mined, is further illustrated in Figures 30 and 31, which provide a comparison of aerial photographs of these shorelines taken during an Airborne Lazer Survey (ALS) during a low spring tide in 2002 (i.e. pre-mining), and the recent infra-red aerial survey of kelp beds undertaken during a low spring tide in March 2006 (Pulfrich & Penney 2006). The extent of beach accretion, particularly in the vicinity of South Rock, can be clearly seen.

Both kelp species *E. maxima* and *L. pallida* can tolerate a degree of sand scour and sedimentation, but persistent sand inundation destabilizes the holdfast, ultimately leading to detachment from the rocky substrate (R. Anderson, Seaweed Unit: Marine & Coastal Management, pers. comm.; this study). At Elizabeth Bay, for example, during March 2005, divers reported seeing the abraded stipes of adult kelp plants sticking out of a 5-10 cm thick layer of sand beyond the current edge of the kelp bed at the Jetty site. Results from the pocket beaches also confirm that the consequence of smothering and abrasion by mobile and accreting sediments is patchy loss of kelp plants.

Plate 23. Abraded kelp fronds and stipes protruding from the recently accreted beach near South Rock (March 2005).

Plate 24. Smothered shallow subtidal kelp bed and abraded fronds at South Rock (March 2006).

Plate 25. Sand inundation of intertidal rocky-shore habitat at South Rock (March 2006).

Plate 26. Eroded kelp holdfast, and mussels torn off the rocks.

Plate 27. Severe scouring and polishing of *S. argenvillei* shells.

Plate 28. Large areas of mussel bed torn off the rocks by wave action. (March 2005).

Plate 29. Smothering of extensive areas of low-shore mussels and shallow water kelp beds by accreting sediments.

Plate 30. Sand inundation of intertidal rocky-shore habitat immediately to the south of the mining site.

Plate 31.Sand inundation of intertidal rocky-shore habitat at Green Precipice, 10 km north of the mining site.

Plate 32.Sand inundation of rocky-shore habitat and fringing kelp at Kaap Dernberg, 20 km north of the mining area.

Plate 33.Sand inundation of intertidal rocky-shore habitat at Cabin, 30 km to the north of the mining site.

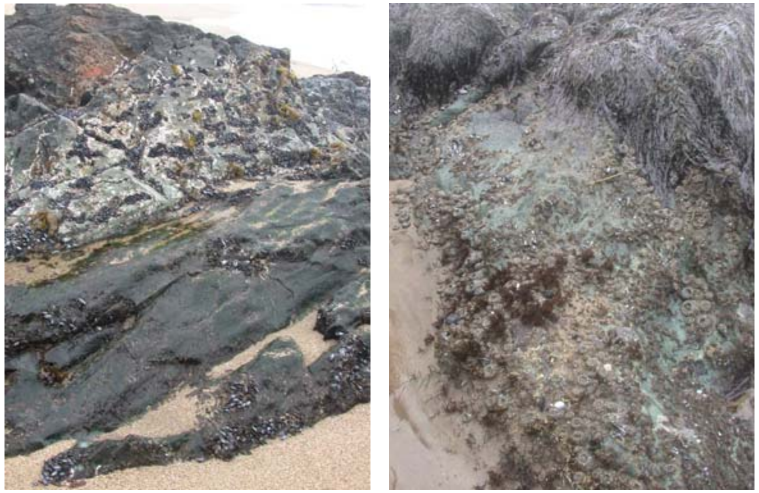


Plate 34.Exposure of patches of rocky habitat previously smothered by sand to the north of the mining site (March 2006).

Plate 35. Proliferation of green foliose algae (primarily *Ulva* sp.) at Cabin, 30 km north of the mining site (March 2006).

Plate 36. Proliferation of green foliose algae (primarily *Enteromorpha* and *Cladophora* spp.) on recently exposed rocks at Green Precipice, 10 km north of the mining site (March 2006).

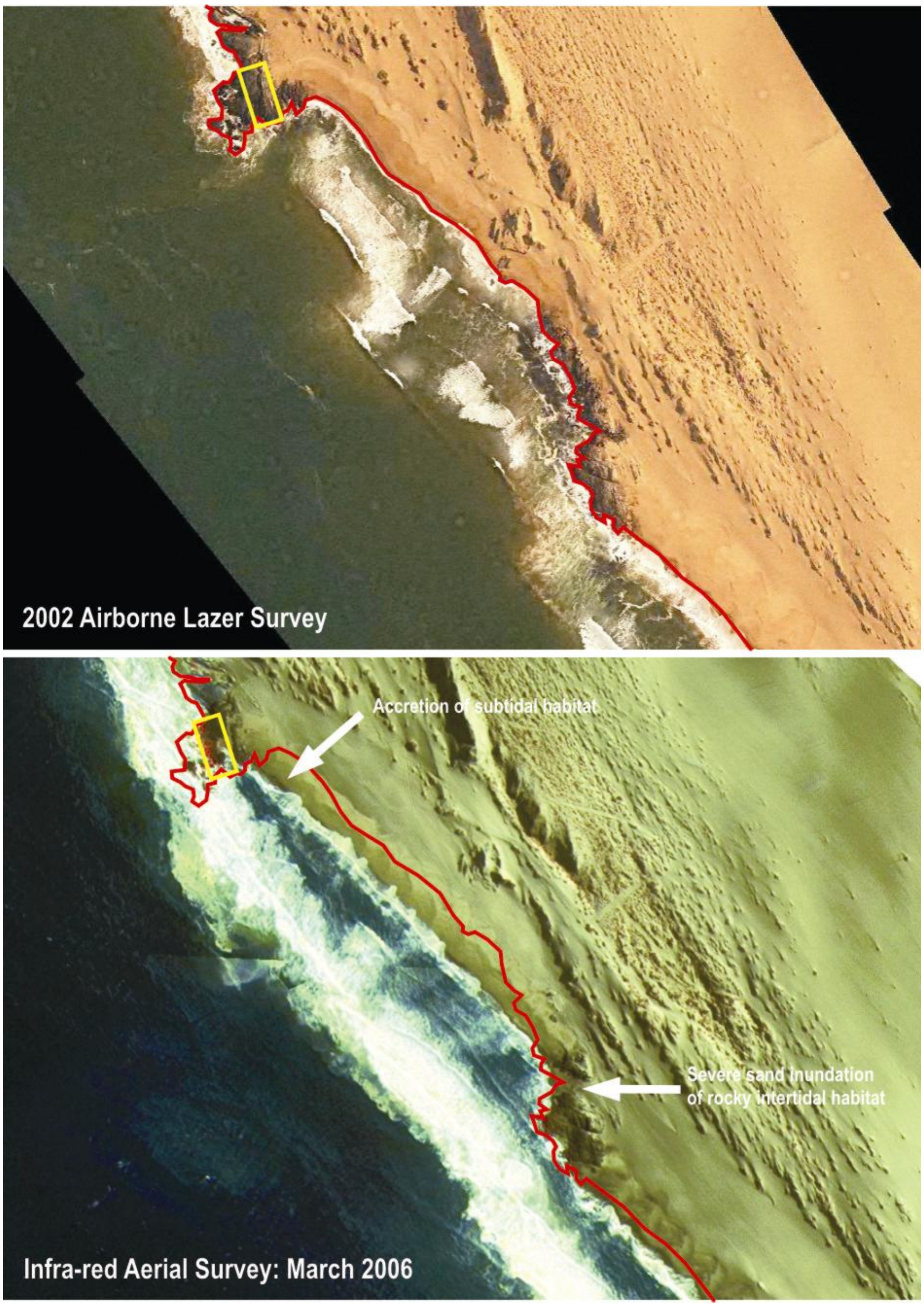


Figure 30. Aerial images of the northern portion of Pocket Beach 2, taken during the 2002 Airborne Lazer Survey (ALS), and 2006 Infra-red Aerial Survey of kelp beds, showing the beach accretion in the vicinity of the South Rock rocky intertidal sampling station (yellow rectangle). The red line represents the ALS low water mark, and is superimposed on the Infra-red image to illustrate the degree of beach accretion.

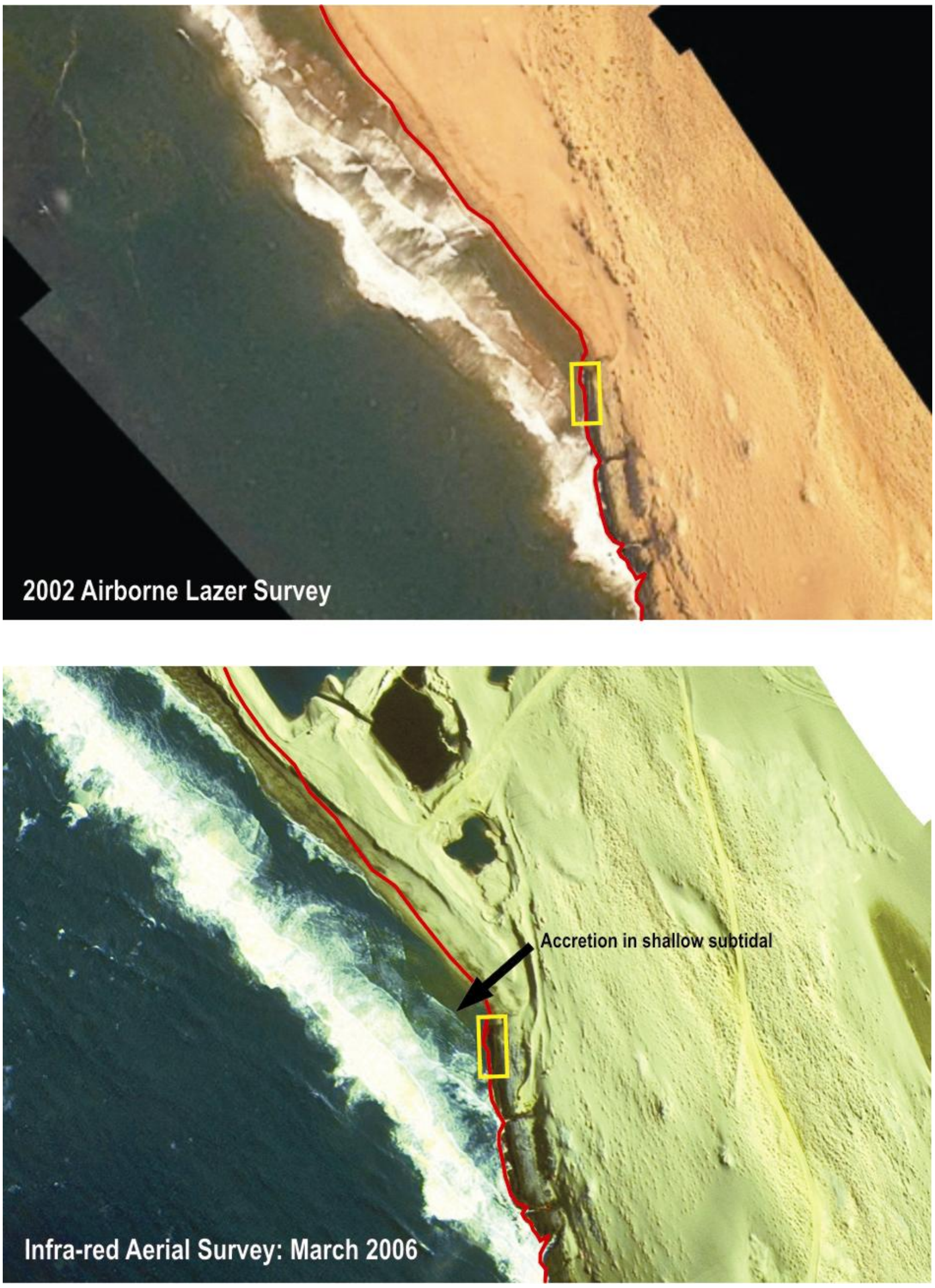


Figure 31. Aerial images of the southern portion of Pocket Beach 2, taken during the 2002 Airborne Lazer Survey (ALS), and 2006 Infra-red Aerial Survey of kelp beds, showing the beach accretion in the vicinity of the rocky intertidal sampling station (yellow rectangle). The red line represents the ALS low water mark, and is superimposed on the Infra-red image to illustrate the degree of beach accretion.

The (insignificantly) lower kelp density at the fines-impacted South Jetty site at Elizabeth Bay, and diver observations of smothered and eroded kelp plants suggest that these macrophytes are being negatively affected by the discharged sediments, either as a consequence of increased turbidity, or the high depositional environment, or both. Ebling *et al.* (1960), for example, found that the distribution of *Laminaria* sp. at Lough Ine was limited by sediments and that sediment removal by turbulent water favoured the plants. Similarly, *Macrocystis* success was reduced where layers of sediment were present on the substratum (Neushal *et al*. 1976). Recruitment of juvenile plants after the experimental removal of kelp at the Jetty site, however, suggests that in the west of Elizabeth Bay, *Laminaria* at least has not yet been significantly influenced by the deposition of mining-related sediments. Whether this recruitment reflects utilisation of available space following the loss of adult kelps and other foliose algae in response to smothering events, or adaptation of the kelp to both elevated turbidity as well as frequent sand inundation, is not known.

There will also be important differences between the smothering and abrading effects, both between the intertidal and subtidal zones, and across sites with different levels of wave exposure. There is likely to be more settlement and accretion of fine sediments in sheltered areas and more resuspension and abrasion by coarse sediments in wave-exposed places. Interaction between sedimentation and water motion is thus likely to be an important contributing factor in variations in occurrence and recruitment of the two dominant kelp species along the southern Namibian coastline.

Although no quantitative data are available for the pocket beaches area, the negative effect of accreting coarse sediments on kelp beds is clearly significantly more extensive than the beach accretion and deposition of fines on rocky shores observed at Elizabeth Bay. The effects of increased abrasion and deposition will not only lead to smothering of available rocky substrata, but result in high levels of mortality of the very young life stages, and consequently reduced recruitment, ultimately leading to reduction in kelp bed area. The greatest effects are likely to be along shores of intermediate exposure and in sheltered bays. Here wave energy is insufficient to keep the sediments in suspension and burial of kelp plants will consequently be prolonged, resulting in the progressive abrasion and smothering of fringing kelp beds along over 30 km of coastline. As it is estimated that this accretion will persist for up to 10 years after mining has ceased (C. Soltau, CSIR, pers. comm.), and this will pose a long-term threat to the future of kelp beds in the area.

4.7 The Effects of Large-scale Natural Events on Kelp Beds

The Benguela Large Marine Ecosystem displays a high degree of variability across a wide range of time and space scales. While the impacts of climate-driven interannual and decadal-scale changes on marine ecosystems are usually most apparent at the higher trophic levels, it is now recognised that major sustained events such as the *Benguela Niños* impact on the ecosystem as a whole. *Benguela Niños*, which appear to be a regional response to changes in the global atmosphere-ocean system, are characterised by a large southward displacement of the Angola-Benguela front, and anomalous large-scale flooding of the Namibian shelf by warm tropical water (Shannon & O'Toole 1989). Depending on the orientation of the flow and the amount of fresh water present from the Congo River, the warm water may be very saline (1984), and at other times may have low surface salinity (1995). They are accompanied by large-scale hypoxia in subsurface and bottom water on the continental shelf, major intrusions of cold Sub-Antarctic Water in the south, and sustained anomalous upwelling (or the absence thereof) caused by changes in regional winds (Shannon & O'Toole 1998). Although less frequent than Pacific *El Niños*, *Benguela Niños* occurred in 1934, 1949, 1963, 1984, 1995 and probably around 1910, in the mid-1920s and in 1972-74.

El Niño events are known to impact kelp forests in a number of ways. Firstly, increased irradiation leads to deterioration and increased erosion of blades, and consequently decreased formation of reproductive tissue. Secondly, as kelp sporophytes have limited temperature tolerances, entire kelp beds can be lost due to prolonged periods of increased water temperatures and low nutrients (Dayton & Tegner 1984; Ladah *et al.* 1999). Dayton *et al*. (1992) report mortalities of up to 66% of adult *Macrocystis* plants in response to low nutrient levels during the 1983/1984 Pacific ENSO. Despite increased light levels in the sub-canopy due to loss of adult plants, the limited nutrients tend to result in reduced recruitment and growth of juveniles (Schroeter *et al*. 1995; Jackson 1987; Rothman 2006). Consequently, notable changes in the canopy have be observed after an *El Niño* event (North *et al.* 1993). The effects of *Benguela Niños* on kelp beds along the southern African west coast have not been studied, however.

Unusual storms have also been reported to have long lasting effects on kelp forests. The effects of storms depends upon their frequency and magnitude as well as the depth to which their effects penetrate (Seymour *et al*. 1989). The proportion of the standing stock lost due to uprooting during storms varies tending to impact patches of the floating adult canopy more severely than the understory canopy (Dayton & Tegner 1984). Velimirov *et al*. (1977) estimate an annual 15% loss of *Ecklonia maxima* through storms in South Africa. Storm losses for *Macrocystis pyrifera* range from 46-49% (Barilotti & Zertuche-González 1990), through 82% (Gerard 1976), to as high as 100% loss of most cohorts during the extreme storm event off California in 1988 (Dayton *et al*. 1992).

Further catastrophic disturbances with long-lasting effects on kelp communities are the establishment (or disruption) of large populations of grazers, usually sea urchins. In Alaska, sea urchin barrens appear fairly persistent (Simenstad *et al*. 1978), while off Nova Scotia extensive barrens exhibit a high degree of stability for at least a decade before mass urchin mortalities led to kelp recovery (Miller & Colodey 1983; Scheibling & Stephenson 1984; Johnson & Mann 1988; Elner & Vadas 1990). According to Steneck *et al*. (2002), urchin-induced deforestation has been increasing globally over the past 2–3 decades. However, since fishing targets have recently shifted down the coastal food webs to include invertebrates such as kelp-grazing herbivores, kelp forests have returned in some locations. The kelp beds in the Benguela ecology system have a comparatively small standing crop of grazers, indicating that relatively little kelp production is passed on to debris feeders. Sea urchins (*Parechinus angulosus*) are the most abundant benthic grazers, but as the prevailing large swells knock them off large kelps, their impact is limited to grazing on kelp recruits. Estimates of the consumption of the annual production of kelps by sea urchins ranges from 5% (Field *et al*. 1977) for *Laminaria* to 14–20% for *Ecklonia* (Anderson *et al*. 1997). In southern Africa *Parechinus* does not form feeding fronts but can nonetheless produce isolated barren patches of a few to tens of metres in 8–15 m depths (Velimirov & Griffiths 1979). Although Anderson *et al.* (1997) suggest that *Parechinus* could prevent recolonization of areas cleared of kelp by harvesting or extreme storm damage, disturbance by catastrophic grazer outbreaks are not considered to be a major factor controlling the stability of kelp habitats in the Benguela region. Experimental removal of *Parechinus angulosus* has failed to increase kelp densities, probably because the urchins feed primarily on drift kelp (Day & Branch 2002).

An indirect effect of *Benguela Niños* and extreme storms is potential declines in salinity. Likewise, periodic floods issuing from the Orange River can affect coastal systems by severely depressing salinities due to elevated freshwater input into the marine system, as well as causing substantial increases in turbidity by increasing the sediment load over a wide area. Depending on the temperature of the flood water, nearshore water temperatures may also be altered. These physical affects can act alone or in synergy to have an impact on the coastal 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.

Although the effects of salinity on kelps has received little study, kelps are stenohaline and therefore do not tolerate wide fluctuations in salinity. Norton & South (1969) established that reduced salinity had adverse effects on *Saccorhiza* and *Chorda*. Depressed salinities appear to have a negative effect on zoospores, germination and early sporophyte growth in *Macrocystis integrifolia* (Buschmann *et al.* 2004) and *Ecklonia radiata* (Burridge *et al*. 1999). Alteration of salinity levels suitable for kelp growth may thus result in a decline in kelp productivity. The physiological tolerances of *Laminari pallida* and *Ecklonia maxima* to changing salinities has not been studied, but evidence suggests that, as with other coastal and marine communities on the southern African west coast, they are sufficiently robust to recover fairly rapidly from major episodic flood events in the relatively short term.

In March 1988 the Orange River experienced its most severe flood of the 20th century, discharging some 15 400 million m^3 of water, and 35.7 million m^3 (64.2 million tons) of bedload and suspended sediment (Branch *et al.* 1990; Bremner *et al.* 1990). Although the most intense effects of the flood on the rocky intertidal and subtidal biota occurred within 30 km of the mouth, consequences of the flood were evident up to 140 km south of the Orange River. Lower than usual salinities were measured as far south as Hondeklip Bay, some 200 km from the river mouth (Branch *et al.* 1990). Species resident above the mid-littoral zone were relatively unaffected by the depressed salinities, presumably as the result of higher physiological tolerances of the biota inhabiting this zone. In the sublittoral zone below about 10 m depth, mortality of *Ecklonia* and *Laminaria* plants, and other biota was negligible as the warm floodwater rode above the saltwater, establishing both a sharp halocline and thermocline. However, between 10 m depth and the mid-littoral zone, the devastation of benthic organisms was almost total, with the reefs in these depths being virtually denuded of all life. Both species of kelp suffered total mortality. Diamond divers operational in the area reported abrupt increases in water clarity and salinity (from 10-29‰), concurrent with rapid decreases in temperature between 5 and 10 m depth.

Diving surveys conducted off Port Nolloth after the March 1974 floods identified that the seabed beyond ~20 m depth was blanketed by mud. Although wave action at shallower depths will tend to keep fine sediments in suspension, flood-related sediments can be expected to also significantly affect the benthos in the inshore areas (Pollock 1982). The influence of river sediments on kelp beds and their associated benthos, following a flooding event can therefore not be excluded. While the impact on adult plants is likely to be minimal in the short-term, increased movement and deposition of flood sediments are likely to negatively affect kelp recruitment in the areas influenced by the effects of the flood.

Kelp beds may also be indirectly affected by sediment input into the coastal marine environment as a consequence of the powerful easterly “Berg winds” occurring along the South African and Namibian coastlines in autumn and winter. These offshore winds are thought to potentially contribute the same order of magnitude of sediments as the annual estimated input of sediment by the Orange River (Zoutendyk 1992, Shannon & O’Toole 1998, Lane & Carter 1999). For example, during a single berg-wind event in May 1979, it is estimated that 50 million tons of dust were transported into the sea during by extensive sandstorms along much of the coast from Cape Frio in the north to Kleinsee in the south (Shannon & Anderson 1982).

Another natural phenomenon ubiquitous to the Benguela system are red tides (dinoflagellate and/or ciliate blooms) (see Shannon & Pillar 1985; Pitcher 1998). Also referred to as Harmful Algal Blooms (HABs), these red tides can reach very large proportions, with sometimes spectacular effects. Toxic dinoflagellate species can cause extensive mortalities of fish and shellfish through direct poisoning, while degradation of organic-rich material derived from both toxic and non-toxic blooms results in oxygen depletion of subsurface water. Periodic low oxygen events associated with massive algal blooms in the nearshore can have catastrophic effects on the biota of the southern African West Coast, such as large-scale stranding of rock lobsters, and mass mortalities of white mussels, rocky shore biota and fish (Newman & Pollock 1974, Matthews & Pitcher 1996, Pitcher 1998). Anoxia following algal blooms were responsible for the large scale rock lobster 'walkout' of 1997, which lead to the death of up to 2000 tons of lobsters (Pitcher 1998). Closely associated with seafloor hypoxia, particularly off central Namibia, is the generation of toxic hydrogen sulphide and methane within the organically-rich, anoxic muds following decay of expansive algal blooms. This is periodically released from the muds as 'sulphur eruptions', causing upwelling of anoxic water and formation of surface slicks of sulphur discoloured water. These eruptions strip dissolved oxygen from the surrounding water column, resulting in mass mortalities of marine life. Satellite remote sensing has shown that eruptions off the Namibian coast occur more frequently, are more extensive and of longer duration that previously suspected (Weeks *et al*. 2002, 2004). Such events, however, occur only infrequently in the southern Benguela (eg. the so-called 'black tide' in St Helena Bay in 1994) (Pitcher 1998).

The direct effects of HABs and sulphur eruptions on kelps have not been well studied. Extensive blooms have been reported to lower light availability resulting in the shading-out and decimation of eelgrass (*Zostera marina*) beds and perrenial seaweeds such as the kelps (*Laminaria saccharina* and *L. digitata)* (Dennison *et al*.1980; McLain & McHale 1996). Elevated concentrations of sulfide in the sediments of eelgrass beds has been reported to result in reduced growth rates of the seagrasses, which draw most of their nutrients from the sediments rather than the water column, or even poison them and lead to their decline. Eelgrass also loses the ability to oxygenate the sediments under high sulfide concentrations (Howarth *et al*. 2000).

Harmful algal blooms have become recognised as a multi-faceted "ecosystem" problem. As their impacts are often subtle and difficult to document, their true extent and significance is often not established. However, HABs are known to affect all trophic compartments of marine foodweb, with chronic sublethal impacts probably being more significant than acute impacts in altering foodwebs and causing trophic dysfunction (Pitcher 1998; Gastrich 2000).

5. CONCLUSIONS

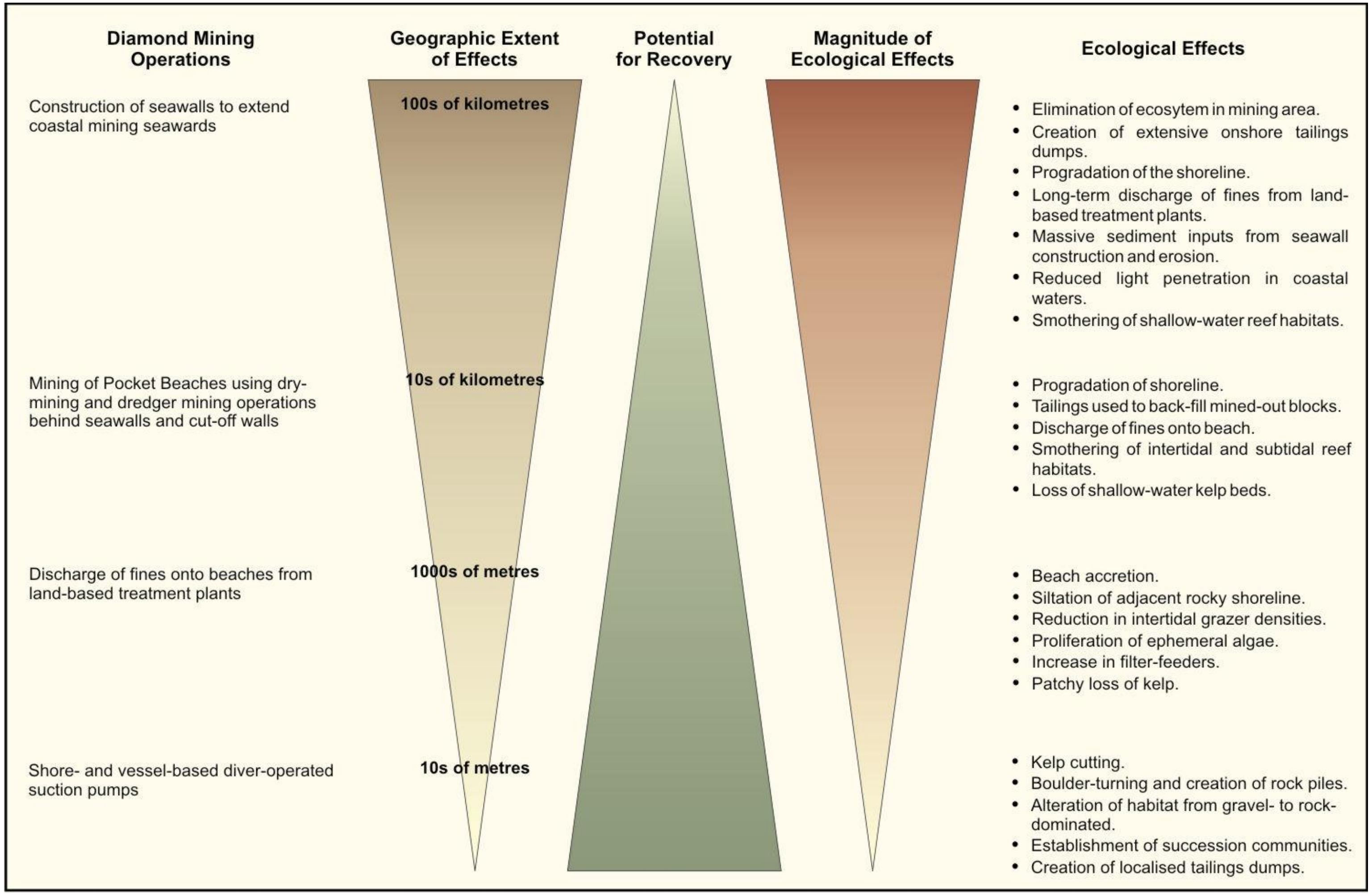
Coastal and nearshore diamond mining on the Southern African West Coast can be broadly categorised into four operations. At increasing scales and severity of impacts to the coastal ecosystems these are:

- Diver-based operations either from the shore or from small vessels;
- Fines deposition from land-based processing plants;
- Pocket beach mining involving wet- and dry-mining behind comparatively localised seawalls; and
- Large-scale land reclamation and opencast mining behind extensive seawalls as practiced in southern Namibia.

The geographic extent and magnitude of their ecological impacts, and potential for recovery are summarise in Figure 32, and recapped briefly below.

Shallow-water diamond mining is opportunistic in nature, being highly dependent on weather and sea conditions. These circumstances effectively limit the periods in which mining can take place, with five days of mining per month by shore units being considered a good average for the South African west coast. Sea conditions also dictate where safe operations can be conducted, as these often have to be in areas with some wave shelter. In addition, the topography of the shoreline will determine where operators can place their mining equipment. Furthermore, swell conditions and underwater visibility can vary enormously over small spatial and temporal scales, making it necessary to choose appropriate mining sites on specific days rather than sequentially mining a concession from one end to the other.

When seen in context with the wide distribution range of the two kelp species in question, and the effects of large-scale natural disturbances characteristic of the Benguela ecosystem, the cutting of kelp and discharge of sediments by diamond divers, creates an extremely localised impact. Their method of operation does not permit a



Diamond Mining Operations	Geographic Extent of Effects	Potential for Recovery	Magnitude of Ecological Effects	Ecological Effects
Construction of seawalls to extend coastal mining seawards	100s of kilometres			• Elimination of ecosytem in mining area. • Creation of extensive onshore tailings dumps. • Progradation of the shoreline. • Long-term discharge of fines from land-based treatment plants. • Massive sediment inputs from seawall construction and erosion. • Reduced light penetration in coastal waters. • Smothering of shallow-water reef habitats.
Mining of Pocket Beaches using dry-mining and dredger mining operations behind seawalls and cut-off walls	10s of kilometres			• Progradation of shoreline. • Tailings used to back-fill mined-out blocks. • Discharge of fines onto beach. • Smothering of intertidal and subtidal reef habitats. • Loss of shallow-water kelp beds.
Discharge of fines onto beaches from land-based treatment plants	1000s of metres			• Beach accretion. • Siltation of adjacent rocky shoreline. • Reduction in intertidal grazer densities. • Proliferation of ephemeral algae. • Increase in filter-feeders. • Patchy loss of kelp.
Shore- and vessel-based diver-operated suction pumps	10s of metres			• Kelp cutting. • Boulder-turning and creation of rock piles. • Alteration of habitat from gravel- to rock-dominated. • Establishment of succession communities. • Creation of localised tailings dumps.

Figure 32. Summary of the geographic extent, potential for recovery and magnitude of the ecological effects of the four main categories of coastal and shallow-water marine diamond mining. See text for details.

phased approach to mining of the surf-zone and shallow water areas, and consequently the areas of kelp removed, and volumes of sediment discharged, remain small and focussed. Compared to the areas of kelp harvested commercially, or damaged during storm events, the area affected through kelp cutting is insignificant, even when seen on a cumulative scale. Only in extremely isolated and highly productive "hotspots" do diver operations remain in an area for an extended period of time. This is both a function of sea conditions and the natural re-deposition of unconsolidated sediments into mined out areas. Despite rapid recovery of cut lanes, repetitive removal of kelp will therefore seldom occur. If areas are repeatedly cut, this is likely to occur on a scale of no more than a few 10s of square metres at any one mining site.

It is possible that when these small-scale, localised impacts occur in synergy with large-scale events such as *Benguela Niños*, extreme floods or chronic deposition of sediments in coastal areas as a consequence of mining-related activities, recovery of the disturbed areas may take longer. As kelp forests are highly dynamic ecosystems, showing considerable temporal and spatial variation, the persistence of such disturbance effects is, however, likely to vary on both local and regional scales (Dayton 1975a, 1985a; Schiel & Foster 1986; Harrold & Pearse 1987; Tegner & Dayton 1987; Johnson & Mann 1988; Kennelly 1989). The results of this and previous studies indicate that kelp beds appear relatively resilience to small-scale disturbances, showing considerable stability on a time scale of years. Furthermore, localised disturbances to kelp beds, either in the form of kelp cutting or diver-assisted diamond mining operations, do not have significant effects on the abundance and distribution of commercially important species such as rock lobsters. In fact, Steneck *et al*. (2002) in their study on the stability, resilience and future of kelp forest ecosystems, conclude that it is rather the overfishing of apex predators (such as rock lobsters, sea otters and finfish) in kelp forests that remains the greatest threat to these ecosystems, and suggest that management remain focussed on minimizing fishing impacts and restoring populations of functionally important species in these systems.

Other disturbances that are also known to have had measurable impacts on kelp forest ecosystems and marine biodiversity over the past few decades at a global scale are climate changes (Steneck *et al*. 2002), and the world-wide increase in water turbidity and sediment deposition in coastal areas as a consequence of anthropogenic activities (Airoldi 2003). The control of these should be a global priority. When evaluated in context with these and other large-scale natural disturbances such as severe storms, grazer outbreaks and *El Niño* events, the effects of kelp cutting and mining activities by shore-based diamond divers is insignificant, even when considered on a cumulative scale.

Of far greater concern is the continual and increasing discharges of fines from land-based processing plants and sediment inputs through seawall erosion from coastal and pocket beach mining operations. The results of ongoing monitoring studies on the discharge of fines at Elizabeth Bay have shown that the sediment slurries discharges from land-based processing plants have a measurable affect on intertidal and subtidal habitats and their associated benthic communities in the immediate vicinity of the outfall point. The impacted communities are characterised by low grazer densities, a proliferation of ephemeral foliose algae and a high cover of filter-feeders. Progressive beach accretion and the concomitant smothering of reef habitats has also resulted in the patchy loss of kelp plants (Pulfrich 2006).

On a significantly larger scale is the effect of increased sediment load in the nearshore environment as a result of seawall construction and erosion, and fines discharges from onshore gravel treatment facilities between Bogenfels Arch and Chameis Bay. These pocket beach mining operations are having a significant impact on rocky shore communities and shallow-water kelp beds in the area by progressive shoreline accretion and inundation of rocky shores by sand, both to the north and to the south of the mining target area. In rock lobster Zone 15, the Kerbe Huk area is the main southern commercial fishing ground (see Figure 26, pg 58). The dominance of sandy coastline in the area, however, implies that recruitment onto the fishing grounds must occur primarily from the kelp beds and shallow water reefs to the north (i.e. Zones 13 and 14, where smothering of kelp beds is currently occurring as a result of mining activities). Consequently, any reduction or loss of kelp beds in these zones by mining–related suspended and/or depositing sediments may have potentially important implications for the long-term sustainability of rock lobster resources on the southern fishing grounds.

Further south towards the Orange River mouth, extensive seawall mining has been in progress along the coast of Mining Area 1 for the past sixty years. As a result, the shoreline, and even the landscape south of Chameis Head has been significantly altered from the original. As mining activities began before the development of environmental concerns, no records of faunal/floral distribution prior to seawall mining are available. Consequently the magnitude of the biological impact that these mining operations have had have not been quantified, but can safely be assumed to be significant.

In addition to the coastal and nearshore diamond mining operations mentioned above, diamonds are also extracted from the seabed further offshore, at depths down to 120 m. Deep-water mining is currently conducted by airlifts, large-diameter drills, seabed crawlers and dredgers. Of these, however, only dredging operations affect coastal habitats, as dredged sediments are pumped ashore for stockpiling before being processed by land-based treatment plants. The fines from these plants are discharged across the beach into the sea. The magnitude, extent and severity of the impacts associated with deep-water mining are discussed in detail in the project BEHP/CEA 03/02: Data Gathering and Gap Analysis for Assessment of Cumulative Effects of Marine Diamond Mining Activities on the BCLME Region, and they are mentioned here only for the sake of completion. Mining rates of dredgers are orders of magnitude greater than those for the nearshore operations. It is predicted that during full-scale mining operations using dredgers between 2 and 4 km^2 of the Atlantic 1 Mining Licence Area will be dredged each year. Whilst taking only three month to mine this area, it is anticipated that sufficient gravel will be produced to keep a land-based production facility in operation for one year (Pisces Environmental Services 2004).

6. RECOMMENDATIONS

6.1 Management Recommendations for Kelp Cutting

Macrophyte communities appear to be relatively resistant to long-term damage by harvesting or removal. Nonetheless, a number of management recommendations relating to the removal of kelp during shore-based diamond mining activities can be made. Although kelp cutting is not universally practiced by diamond divers working in surf-zone and shallow water concessions on the South African and Namibian coasts, it would be prudent for diamond companies that sub-contract diamond divers to include these management recommendations in their Environmental Management Programme Reports, and Management Plans, and regularly ensure that they are being implemented.

The following management recommendations should be considered:

1. Do not cut kelp unless diver safety is at stake or it is essential for the operation.
2. Where the removal of kelp is necessary, the following guidelines should be applied:
    - If kelp has to be thinned or removed, cut individual plants with a sharp knife just above the holdfast. Retain the holdfast to maintain (temporary) refuges for benthic fauna and settlement substratum for kelp recruits. Do not drag or tear the kelp out or damage the holdfast.
    - To facilitate rapid kelp re-establishment do not cut plants with a stipe length of less than 10 cm.
    - Cut the kelp in lanes to promote re-colonisation from mature plants on the edges of the lanes.
    - Where fairly extensive kelp removal is required, notify the relevant kelp harvesting permit holders to collect the kelp once it has been cast onto the shore.
    - Allow at least 18 months for the kelp community to recover before re-cutting an area.

6.2 Management, Mitigation and Monitoring Recommendations

Effective management and mitigation of seawall construction and erosion, and the large volumes of tailings discharged annually from the processing plants on the southern Namibian coastline should become a priority to the diamond companies concerned, as well as the Government of Namibia. While it can be argued that these sediment inputs occur on a sediment-dominated coastline frequently influenced by natural large-scale sediment inputs from Orange River floods and “Berg wind” events, the threat to biodiversity and the structure and functioning of rocky shore assemblages as a consequence of the world-wide increase in sediment deposition in coastal areas must always be kept in mind.

The following management recommendations should be considered and effectively implemented for all current and future coastal diamond mining operations:

- The duration of mining operations at each site should be kept to a minimum to keep the cumulative impacts associated with mining at acceptable levels.
- Active rehabilitation of the marine environment below the high water mark is not required, as rehabilitation within this highly dynamic nearshore area is a natural process. Nonetheless, restoration and rehabilitation activities of coastal impacts should become an integral part of the mine plan, and should be run concurrently with mining activities, especially when they are expected to be operational in the medium to long term.
- As extensive stretches of coastline have been reclaimed by the construction of seawalls and are backed by flooded excavations, back-filling such mined-out areas with overburden from tailings heaps and plant discharges should be investigated and implemented wherever feasible.
- The extent of seawalls should be limited only to that necessary for the protection and safety of mining operations, and the future implementation of alternative techniques such as cut-off walls should become a priority. Seawalls should be breached after the termination of mining to allow more rapid natural rehabilitation of the beach by wave action.
- The discharge of fines should be kept to a minimum, but where unavoidable, discharge points should be positioned to facilitate rapid mixing of fines and re-distribution of tailings in the longshore drift, thereby minimising impact on rocky shore communities.
- Marine monitoring programmes should be implemented for each mining operation to establish the effect of sediment discharge on sandy beach and rocky shore communities, and kelp beds, and to ascertain the rate of recovery of impacted habitats. These monitoring programmes should include at least one year of pre-mining baseline data, and continue on a regular basis for the duration of the life of the mine, and for at least 5 years after mine closure. Monitoring of ‘control’ or ‘reference’ areas that are not impacted by mining must take place in parallel.
- Implementation of Marine Protected Areas (MPAs) where no mining or any other extractive harvesting is permitted. Such areas must be large enough to be unaffected by mining-related silt plumes and have representative examples of all

habitat types. Such areas could subsequently also serve as true ‘control’ or ‘reference’ areas against which to assess the impacts of mining.

As emphasised by Airoldi (2003), there is a lack of information on the spatial and temporal variability in both the patterns of deposition of sediments and benthic assemblages. The effects of changes in sediment regimes over large spatial scales have seldom been addressed, and most past studies on sediment effects have been carried out over very short periods only. As part of the BCLME initiative, three projects were proposed to investigate the cumulative impacts of mining in the Benguela region:

- BEHP/CEA/03/02: Data Gathering and Gap Analysis for Assessment of Cumulative Effects of Marine Diamond Mining Activities on the BCLME Region.
- BEHP/CEA/03/03: Assessment of the Cumulative Effects of Sediment Discharges from On-shore and Near-shore Diamond Mining Activities on the BCLME.
- BEHP/CEA/03/04: Assessment of cumulative impacts of scouring of sub-tidal areas and kelp cutting by diamond divers in near-shore areas of the BCLME region.

While these have improved understanding of the cumulative effects of mining-related sediments and their spatial and temporal scales (Smith *et al*. 2006; Pisces Environmental Services, in prep.), effective future monitoring of mining impacts, implemented as part of the Environmental Monitoring Plans, provides excellent opportunity to gain a greater insight into trends of changes in mining-related sediment loads and the responses of biotic communities to these. It is important, however, that mining companies adopt an integrated approach to both management and monitoring of the nearshore marine environment rather than focussing efforts on assessing and quantifying project-specific impacts only.

A problem that remains is putting the observed trends of changes from mining impacts in context relative to the often extreme natural fluctuations characterising the Benguela ecosystem. Monitoring of mining impacts alone is insufficient if one wants to confidently identify whether the causes of change are due to human activities, or are attributable to natural processes. This, however, requires the implementation of large-scale and long-term monitoring of ‘baseline’ biophysical factors. Monitoring programmes in this regard that would enable a more accurate assessment of the impacts of mining related sediments include:

- long-term, large-scale monitoring of sediment deposition in the coastal environment using sediment traps and other means to record seasonal and longer-term shifts in sedimentation in both intertidal and subtidal locations.
- Regular, georeferenced airborne lazer surveys to monitor changes in the coastline as a consequence of accretion or erosion.
- Monitoring of kelp bed areas using aerial photography techniques as indicators of climate change, *Benguela Niños* and/or increases in sediment deposition.

Such monitoring is, however, labour intensive and costly and should be conducted under the auspices of multi-national research programmes rather than be the responsibility of individual mining companies. The frequency and intensity of sampling need careful planning to ensure detection of effects while minimising cost and effort.

6.3 Recommendations for Monitoring of Rock Lobster Recruitment

Public misconceptions regarding the impacts of marine diamond mining continue to fuel conflicts that arose between the marine diamond mining and rock lobster fishing industries in the early 1990s. As the impacts of long-term increases in sediment deposition on kelp beds and marine assemblages in southern Namibia have not been adequately quantified, and information on recovery of marine habitats following cessation of large-scale sediment discharges and accumulation is lacking, it is recommended that effort be made to obtain indices of rock lobster puerulus and juvenile abundances, in areas heavily influenced by mining sediments as well as in unimpacted or lightly impacted areas. While for the purposes of predictive fisheries management of *J. lalandii* it would be prudent to monitor both puerulus settlement and juvenile abundances, Butler & Herrnkind (2000) suggest that in cases where the relationship between puerulus supply and adult population size may be disrupted by post-settlement processes, juvenile abundance indices are often more appropriate for stock prediction. Such research should be partly funded by the diamond companies concerned, and partly by MFMR and carried out in close collaboration with the rock lobster section of the MFMR.

6.4 BEHP/CEA/04/04 Project Recommendations

To date four major surveys have been undertaken during which quantitative data on rock lobster density, benthic cover and kelp biomass were collected. These were conducted in April and September 2005, and April and October 2006 (Pisces Environmental Services 2005a, 2005b, 2006a, 2006b). Maintenance surveys, during which puerulus collectors were serviced and settlement rates determined, were undertaken in September 2005, and January, February, March, May, July and October 2006. The results of these surveys have been presented to BCLME in the form of three Progress Reports (submitted May 2005, October 2005 and June 2006) an Interim Progress Report (submitted October 2006), and this Final Report (February 2007). Therefore, despite substantial delays in the initial establishment of the field experiment and subsequent cancellation of several of the scheduled interim surveys due to delays in receiving suitable long-term work visas, the outcome of this project has been generally successful.

Since the initiation of the underwater experiment in April 2005, the galvanised steel puerulus collectors originally obtained for the project from the MFMR have required constant maintenance, repair and replacement, as they suffer from severe corrosion, even over short periods in the sea. Unfortunately, due to budget constraints and the high costs of manufacturing stainless steel collectors, not all collectors initially installed were stainless. Consequently there was occasionally some loss of data as galvanised collectors broke apart in the period between maintenance surveys. However, following interest shown by MFMR in the results of the project, and their support in principle in continuing with the collection of data, additional stainless steel components were purchased and manufactured. This was possible because of savings made on some budget items, and by October 2006 all galvanised components on existing collectors had been replaced with stainless steel. All three study sites therefore now have stainless collectors, and further loss of data due to loss of collectors is unlikely.

Despite the achievements of the project to date, only a relatively short data series on puerulus settlement is available, and is somewhat incomplete as some surveys had to be cancelled due to delays in obtaining new work visas. Furthermore, differences in community structure between experimentally cut and uncut control lanes are only now becoming evident. Following interest by MFMR in the outcome of the project, Ms

Kolette Grobler of the rock lobster section of MFMR in Lüderitz, has recommended the study be continued for at least another year, to provide a two-year data set. A letter of motivation from Ms Grobler in this regard has been forwarded to the BCLME offices.

It is therefore proposed that consideration be given to extending the project to January 2008. This would substantially improve the data set and provide a greater insight into the longer-term effects of kelp cutting, and the subsequent recovery of the benthic communities, as well as temporal and spatial variability in rock lobster puerulus settlement in habitats influenced by mining-related suspended sediments. Furthermore, recommendation have been made through the Namdeb Elizabeth Bay Biological Monitoring Programme to install sediment traps at some of the kelp experiment sites. Obtaining puerulus settlement data concurrent with sedimentation rates would provide extremely valuable quantitative information on the effects of sedimentation and tailings disposal from the mine, on rock lobster puerulus settlement and subsequent recruitment.

7. ACKNOWLEDGMENTS

This project was funded by the Global Environment Facility (GEF) through the United Nations Development Programme (UNDP). The success of this project is largely attributable to the members of the project field team. In particular, Mark Noffke and Pieter Truter, without whose inputs into the design and setting up of the underwater experiment, intensive underwater work (often under harsh and uncomfortable conditions), and untiring dedication, this project would not have been possible. Hundreds of man hours of underwater work were required in the installation, data collection, kelp cutting and changing of puerulus collectors, and consequently the efforts and extensive scientific diving experience of the team members were paramount to the success of the project. Thanks also go to the management and staff of the Elizabeth Bay Mine and Namdeb's Northern Areas for their interest and support of the project, in providing access to the mining licence area, accommodation on the mine for the project team, access to their mechanical workshop and use of their portable compressor to fill aqualungs. The use of the MFMR puerulus collectors, and the interest in the project by Kolette Grobler and the technical staff of the Rock Lobster Section in Lüderitz, is acknowledged. Sincere thanks go to Professor George Branch from the University of Cape Town for providing constructive criticism and advice throughout the course of the project and for reviewing the final document.

8. REFERENCES

ADDISON, J.T., 1997. Lobster stock assessment: report from a workshop; I. *Mar. Freshw. Res.,* **48**: 941-944.

AIROLDI, L., 2003. The effects of sedimentation on rocky coast assemblages. *Oceanogr. Mar. Biol. Ann. Rev.*, **41**: 161-236.

AIROLDI, L. & M. VIRGILIO, 1998. Responses of turf-forming algae to spatial variations in the deposition of sediments. *Mar. Ecol. Prog. Ser.*, **165**: 271-282.

AIROLDI, L. & F. CINELLI, 1997. Effects of sedimentation on subtidal macroalgal assemblages: an experimental study from a Mediterranean rocky shore. *J. Exp. Mar. Biol. Ecol.,* **215**: 269-288.

AIROLDI, L., FABIANO, M. & F. CINELLI, 1996. Sediment deposition and movement over a turf assemblage in a shallow rocky coastal area of the Ligurian Sea. *Mar. Ecol. Prog. Ser.*, **133(1-3)**: 241-25.

AIROLDI, L., RINDI, F. & F. CINELLI, 1995. Structure, seasonal dynamics and reproductive phenology of a filamentous turf assemblage on a sediment influenced, rocky subtidal shore. *Bot. Mar.*, **38(3)**: 227-237.

ALLEN, J.C. & C.L. GRIFFITHS, 1981. The fauna and flora of a kelp bed canopy. *S. Afr. J. Zool.*, **16**: 80-84.

AMSLER, C.D., REED, D.C. & M. NEUSHUL, 1992. The microclimate inhabited by macroalgal propagules. *Br. Phycol. J.*, **27**: 253– 270.

ANDERSON, R.J., CARRICK, P., LEVITT, G.J. & A. SHARE, 1997. Holdfasts of adult *Ecklonia maxima* provide refuges from grazing for recruitment of juvenile kelps. *Mar. Ecol. Prog. Ser.*, **159**: 265-273.

ANDREWS, W.R.H. & L. HUTCHINGS, 1980. Upwelling in the southern Benguela Current. *Prog. Oceanogr.*, **9(1)**: 1-81.

ARMSTRONG, S.L., 1989. The behavior in flow of the morphologically variable seaweed *Hedophyllum sessile* C. Ag. Setchell. *Hydrobiologia*, **183**: 115–122.

BAILEY, G.W., BEYERS, C.J. DE B. & S.R. LIPSCHITZ, 1985. Seasonal variation of oxygen deficiency in waters off southern South West Africa in 1975 and 1976 and its relation to catchability and distribution of the Cape rock lobster *Jasus lalandii*. *S. Afr. J. mar. Sci.*, **3**: 197-214.

BARILOTTI, D.C. & J.A. ZERTUCHE-GONZÁLEZ, 1990. Ecological effects of seaweed harvesting in the Gulf of California and the Pacific Ocean off Baja California and California. *Hydrobiologia*, **204/205**: 35-40.

BARKAI, A. & M.O. BERGH, 1992. *The effects of marine diamond pumping operations on the littoral and shallow sublittoral benthos along the South African west coast, Namaqualand region, with special attention to possible effects on the rock lobster resource: a pilot study.* UCT Unpublished Report. 43 pp.

BARKAI, A. & G.M. BRANCH, 1988. Contrasts between the benthic communities of subtidal hard substrata at Marcus and Malgas Islands: a case of alternative stable states? *S. Afr. J. Mar. Sci.*, **7**: 117-137.

BERRY, P.F., 1982. Biomass and density of detritivores on a rocky littoral reef on the Natal coast, with an estimate of population production for the ascidian, *Pyura stolonifera*. *Invest. Rep. Oceanogr. Res. Inst.*, **53**:1-12

BEYERS, C.J. DE B., 1979. Stock assessment and some morphometric and biological characteristics of the rock lobster *Jasus lalandii* on Marshall Rocks, its main commercial fishing area off South West Africa, 1971-1974. *Investl. Rep. Sea Fish. Brch S. Afr.*, **117**: 1-26.

BEYERS, C.J. DE B. & C.G. WILKE, 1990. The biology, availability and exploitation of rock lobster *Jasus lalandii* off South West Africa/Namibia, 1970-1980. *Investl. Rep. Sea Fish. Brch S. Afr.*, **133**: 1-56.

BEYERS, C.J. DE B., WILKE, C.G. & P.C. GOOSEN, 1994. The effects of oxygen deficiency on growth, intermoult period, mortality and ingestion rates of aquarium-held juvenile rock lobster *Jasus lalandii*. *S. Afr. J. mar. Sci.*, **14**: 79-88.

BODKIN, J.L., 1988. Effects of kelp forest removal on associated fish assemblages in central California. *J. Exp. Mar. Biol. Ecol.*, **117**: 227-238.

BOLTON, J.J. & G.J. LEVITT, 1985. Light and temperature requirements for growth and reproduction in gametophytes of *Ecklonia maxima* (Alaraiceae, Laminariales), *Mar. Biol.*, **87**: 131-135.

BOOTH, J.D., 1979. Settlement of the rock lobster, *Jasus edwardsii* (Decapoda: Palinuridae), at Castlepoint, New Zealand. *N. Z. J. Mar. Freshw. Res.,* **13(3)**: 395-406.

BOOTH, J.D., 2001. Habitat preferences and behaviour of newly settled *Jasus edwardsii* (Palinuridae). *Mar. Freshw. Res.*, **52**: 1055-1065.

BOOTH, J.D., FOREMAN, J.S., STROTTER, D.R., BRADFORD, E., RENWICK, J. & S.M. CHISWELL, 1999. Recruitment of the red rock lobster, *Jasus edwardsii*, with management implications. *New Zealand Fisheries Assessment Research Document* **99(10)**: 102pp.

BOOTH, J.D. & J. KITTAKA, 1994. Grow out of juvenile spiny lobster. In: PHILLIPS, B.F., COBB, J.S. & J. KITTAKA (Eds), *Spiny Lobster Management.*. Oxford, UK, Fishing News Books, Blackwell Scientific Publications: 424-445.

BOOTH, J.D & J. KITTAKA, 2000. Spiny lobster growout. In: PHILLIPS, B.F., COBB, J.S. & J. KITTAKA (Eds), *Spiny Lobster Management.*. Oxford, UK, Fishing News Books, Blackwell Scientific Publications: 556-585.

BOOTH, J.D., FOREMAN, J.S., STROTTER, D.R. & E. BRADFORD, 2001a. Settlement indices for 1999 and 1999-2000 juvenile of the red rock lobster *Jasus edwardsii*. *New Zealand Fisheries Assessment Report.* **2001(28)**: 34pp.

BOOTH, J.D., STROTTER, D.R., FOREMAN, J.S. & E. BRADFORD, 2001b. Juvenile abundance both mirrors and masks a settlement pulse of the rock lobster *Jasus edwardsii* (Palinuridae). *Mar. Freshw. Res.* **52**: 1067-1075.

BORCHERS, P. & J.G. FIELD, 1981. The effect of kelp shading on phytoplankton production. *Bot. Mar.*, **24**: 89-91.

BRANCH, G.M. & M. BRANCH, 1981. *The Living Shores of Southern Africa*. C. Struik Publishers, Cape Town, pp 272.

BRANCH, G.M., EEKHOUT, S. & A.L. BOSMAN, 1990. Short-term effects of the 1988 Orange River floods on the inter-tidal rocky-shore communities of the open coast. *Trans. Roy. Soc. S. Afr.,* **47**: 331-354.

BRANCH, G. M., GRIFFITHS, C. L., BRANCH, M. L. & L. E. BECKLEY, 1994. *Two Oceans - A Guide to the Marine Life of Southern Africa*. David Philip, Cape Town, pp 353.

BRANDT, R.P., 1923. Potash from kelp: early development and growth of the giant kelp *Macrocystis pyrifera*. *U.S. Dept. Agr. Bull*., **1191**: 40pp.

BRAY, J.R. & J.T. CURTIS, 1957. An ordination of the upland forest communities of Southern Wisconsin. *Ecol. Monogr.*, **27**: 325-349.

BREMNER, J.M., ROGERS, J. & J.P. WILLIS, 1990. Sedimentological aspects of the 1988 Orange River floods. *Trans. Roy. Soc. S. Afr.*, **47**: 247-294.

BUSCHMANN, A.H., VÁSQUEZ, J.A., OSORIO, P., REYES, E., FILÚN, L., HERNÁNDEZ-GONZÁLEZ M.C. & A. VEGA, 2004. The effect of water movement, temperature and salinity on abundance and reproductive patterns of *Macrocystis* spp. (Phaeophyta) at different latitudes in Chile. *Mar. Biol*., **145(5)**: 849-862.

BUSTAMANTE, R.H., BRANCH, G.M. & S. EEKHOUT, 1997. The influences of physical factors on the distribution and zonation patterns of South African rocky-shore communities. *S. Afr. J. Mar. Sci.*, **18**: 119-136.

BUSTAMANTE, R.H. & J.C. CASTILLA, 1990. Impact of Human Exploitation on Populations of the Intertidal Southern Bull-kelp *Durvillaea antarctica* (Phaeophyta, Durvilleales) in Central Chile. *Biological Conservation*, **52**: 205-220.

BUTLER, M., 2000. Recruitment of Caribbean spiny lobster in a dynamic nursery. In: FRUSHER, S., PRESCOTT, J., HOBDAY, D. and C. GARDNER (Eds), *Workshop on Post-settlement Processes Affecting the Southern Rock Lobster, Jasus edwardsii, in Southern Australia* University of Tasmania, Tasmanian Aquaculture and Fisheries Institute Report **10**: 28pp.

BUTLER IV, M.J. & W.F. HERRNKIND, 2000. Puerulus and juvenile ecology. In: PHILLIPS, B.F. & J. KITTAKA (Eds), *Spiny Lobsters Fisheries and Culture*. Oxford, UK. Fishing News Books, Blackwell Scientific Publications: 276-301.

BUTLER, M.J., HUNT, J.H., HERRNKIND, W.F., CHILDRESS, M.J., BERTELSEN, R.D., SHARP, W. MATTHEWS, T., FIELD, J.M. & H.G. MARSHALL, 1995. Cascading disturbances in Florida Bay, USA: Cyanobacteria blooms, sponge mortality, and implications for juvenile spiny lobsters *Panulirus argus. Mar. Ecol. Prog. Ser*., **129**: 119-125.

BURKE, A., 2005. *Pocket Beach Areas: Amendment to the Environmental Assessment and Environmental Management Plan.* Prepared for NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia, by EnviroScience. 28pp.

BURRIDGE, T. R., KARISTIANOS, M. & J. BIDWELL, 1999. The use of aquatic macrophyte ecotoxicological assays in monitoring coastal effluent discharges in southern Australia. *Mar. Poll. Bull*., **1-12**: 89-96.

BUXTON, C.D. & J.G. FIELD, 1983. Feeding, defaecation and absorption efficiency in the sea urchin, *Parechinus angulosus*, Leske. *S. Afr. J. Zool*., **18**: 11-14.

CAPUTI, N., CHUBB, C.F., HALL, N. & A. PEARCE, 1997. Relationship between different life history stages of the western rock lobster, *Panulirus cygnus*, and their implications for management. In: HANCOCK, D.A., SMITH, D.C. & J.P. BEUMER (Eds), *Developing and Sustaining World Fisheries Resources: The State of Science and Management.* The Second World Fisheries Conference: 579-585.

CARR, M.H., 1989. Effects of macroalgal assemblages on the recruitment of temperate zone reef fishes. *J. Exp. Mar. Biol. Ecol.*, **126**: 59-76

CARR, M.H., 1994. Effects of macroalgal dynamics on recruitment of a temperate reef fish. *Ecology*, **75(5)**: 1320-1333.

CHAPMAN, A.R.O. 1986. Population and community ecology of seaweeds. *Adv. Mar. Biol.*, **23**: 1-161.

CHAPMAN, A.S. & R.L. FLETCHER, 2002. Differential effects of sediments on survival and growth of *Fucus serratus* embryos (Fucales Phaeophyceae). *J. Phycol.*, **38**: 894–903.

CHAPMAN, A.R.O. & J.E. LINDLEY, 1981. Productivity of *Laminaria solidungula* J.AG. in the Canadian High Arctic: A year round study. *Proc. int. Seaweed Symp.*, **10**: 247-252.

CHESHIRE, A.C. & N.D. H ALLAM, 1988. Morphology of the southern Bull-Kelp (*Durvillaea potatorum*, Durvilleales, Phaeophyta) from King Island (Bass Strait Australia). *Bot. Mar*. **31**: 139–148.

CHITTLEBOROUGH, R.G., 1974. Western rock lobster reared to maturity. *Aust. J. Mar. Freshw. Res.,* **25**: 221-225.

CHITTLEBOROUGH, R.G., 1975. Environmental factors affecting growth and survival of juvenile western rock lobsters *Panulirus longipes* (Milne-Edwards). *Aust. J. Mar. Freshw. Res.,* **26**: 177-196.

CHRISTIE, H., FREDRIKSEN, S. & E. RINDE, 1998. Regrowth of kelp and colonization of epiphyte and fauna community after kelp trawling at the coast of Norway. *Hydrobiologia*, **375/376**: 49–58.

CHRISTIE, H., RINDE, E., FREDRIKSEN, S. & A. SKADSHEIM, 1994. Ecological consequences of kelp trawling: re-establishment of kelp forest, epiphytes and holdfast fauna after kelp trawling at the Rogaland coast. *NINA Oppdragsmelding*, **295**: 1–29 (in Norwegian with English summary).

CLARK, K.R., 1993. Non-parametric multivariate analyses of changes in community structure. *Aust. J. Ecol.*, **18**: 117-143.

CLARKE, K.R. & M. AINSWORTH, 1993. A method of linking multivariate community structure to environmental variables. *Mar. Ecol. Prog. Ser.*, **92**: 205-219.

CLARK, B.M., ATKINSON, L.J. & A. PULFRICH, 2005. Sandy Beach and Rocky Intertidal Monitoring Studies in the Bogenfels Mining Licence Area, Namibia. *Monitoring Report 2005. Report to NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia.*

CLARK, B.M., ATKINSON, L.J., STEFFANI, N. & A. PULFRICH, 2004b. Sandy Beach and Rocky Intertidal Baseline Monitoring Studies in the Bogenfels Mining Licence Area, Namibia. Monitoring Report 2004. *Report to NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia.*

CLARK, B.M., PULFRICH, A. & L.J. ATKINSON, 2006. Sandy Beach and Rocky Intertidal Monitoring Studies in the Bogenfels Mining Licence Area, Namibia. Monitoring Report 2006. *Report to NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia*. 133pp

CLARK, B.M., PULFRICH, A. & L.J. ATKINSON, 2006. Sandy Beach and Rocky Intertidal Monitoring Studies in the Bogenfels Mining Licence Area, Namibia. Monitoring Report 2006. *Report to NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia.* 133pp.

CLARK, R.P., EDWARDS, M.S. & M.S. FOSTER, 2004a. Effects of shade from multiple kelp canopies on an understorey algal assemblage. *Mar. Ecol. Prog. Ser.*, **267**:107–119

CLARKE, K.R. & R.M. WARWICK 1994. *Change in Marine Communities – An approach to statistical analysis and interpretation.* Natural Environment Research Council, U.K.

CLARKE, D.G. & D.H. WILBER, 2000. Assessment of potential impacts of dredging operations due to sediment resuspension. DOER Technical Notes Collection (ERDC TN-DOER-E9), U.S. Army Engineer Research and Development Cnter, Vicksburg, MS. www.wes.army/mil/el/dots/doer.

COBB, J.S., 1971. The shelter related behavior of the lobster *Homarus americanus*. *Ecology*, **52**: 108-115.

CONNELL, S.D., 2003a. The monopolization of understorey habitat by subtidal encrusting coralline algae: a test of the combined effects of canopy-mediated light and sedimentation. *Mar. Biol.*, **142**:1065–1071.

CONNELL, S.D., 2003b. Negative effects overpower the positive of kelp to exclude invertebrates from the understorey community. *Oecologia*, **137**: 97–103.

COUSENS, R., 1982. The effect of exposure to wave action on the morphology and pigmentation of *Ascophyllum nodosum* (L.) Le Jolis in South-Eastern Canada. *Bot. Mar*. **25**: 191–195.

CRUZ, R., DΊAZ, E., BÁEZ, M. & R. ADRIANO, 2001. Variability in recruitment of multiple life stages of the Caribbean spiny lobster, *Panulirus argus*, in the Gulf of Batabanó, Cuba. *Mar. Freshw. Res.,* **52:** 1263-70.

CRUZ, R., LEÓN, M.E. & R. PUGA, 1995. Prediction of commercial catches of the spiny lobster *Panulirus argus* in the Gulf of Batabanó. *Crustaceana,* **68**: 238-244.

CSIR, 2001a. Elizabeth Bay monitoring project: 2000 review. *CSIR Report ENV-S-C 2001-037*.

CSIR, 2001b. Chameis Pocket Beaches: Coastal Evolution Predictions to Address Critical Issues. *CSIR Report ENV-S-C 2001-054*.

CSIR, 2002. The effects of Marine Sediment Discharge from Extended Mining Operations at Elizabeth Bay. *CSIR Report ENV-S-C 2002-056*.

CSIR, 2004. Preliminary modelling of suspended sediment and shoreline effects from Marine Dredging at Chameis Bay. *CSIR Report ENV-S-C 2004-110*.

CSIR, 2005. Assessment of the Physical Effects of Sediment Discharged from Marine Dredging at Chameis Bay : Interim Report. *CSIR Interim Report to De Beers Marine Namibia and Namdeb Diamond Corporation (Pty) Ltd.*

D'ANTONIO, C.M.D., 1986. Role of sand in the domination of hard substrata by the intertidal alga *Rhodomela larix*. *Mar. Ecol. Prog. Ser.*, **27**: 263-275.

DALY, M.A. & A.C. MATHIESON, 1977. The effects of sand movements on intertidal seaweeds and selected invertebrates at Bound Rock, New Hampshire. *Mar. Biol.*, **43**:45-55.

DAY, E. & G.M. BRANCH, 2002. Effects of sea urchins (*Parechinus angulosus*) on recruits and juveniles of abalone (*Haliotis midae*). *Ecol. Monogr.*, **72(1)**: 133-149.

DAY, E.G., BRANCH, G.M. & C. VILJOEN, 2000. How costly is molluscan shell erosion ? A comparison of two patellid limpets with contrasting shell structures. *J. Exp. Mar. Biol. Ecol.*, **243**: 185-208.

DAYTON, P.K., 1975. Experimental evaluation of ecological dominance in a rocky intertidal algal community. *Ecol. Monogr.*, **45**: 137-159.

DAYTON, P.K., 1985. The structure and regulation of some South American kelp communities. *Ecol. Monogr.*, **55(4)**: 447-468.

DAYTON, P.K., 1985. Ecology of Kelp Communities. *Ann. Rev. Ecol. Syst.*, **16**: 215-245.

DAYTON, P.K., CURRIE, V., GERRODETTE, T., KELLER, B.D., ROSENTHAL, R. & D. VEN TRESCA, 1984. Patch dynamics and stability of some California kelp communities. *Ecol. Monogr.*, **54**: 253–290.

DAYTON, P.K. & M. TEGNER, 1984. Catastrophic storms. El Niño, and patch stability in a southern Californian kelp community. *Science*, **224**: 283-284.

DAYTON, P.K., TEGNER, M.J., EDWARDS, P.B. & K.L. RISER, 1998. Sliding baselines, ghosts, and reduced expectations in kelp forest communities. *Ecological Applications*, **8**: 309– 322.

DAYTON, P.K., TEGNER, M.J., PARNELL, P.E. & P.B. EDWARDS, 1992. Temporal and spatial patterns of disturbance and recovery in a kelp forest community. *Ecol. Monogr.,* **62**: 421–445.

DEAN, R.L. & J.H. CONNELL, 1987. Marine invertebrates in algal succession. I. Variations in abundance and diversity with succession. *J. Exp. Mar. Biol. Ecol*., **109**: 196–215.

DEAN, T.A. & L.E. DEYSHER, 1983. The effects of suspended solids and thermal discharges on kelp. In: BASCOM, W. (ed.), *The Effects of Waste Water Discharges on Kelp in Southern California*. Southern California Coastal Water Research Project, Long Beach, California.: 114-135.

DENNISON, W.C., MARSHALL, G.J. & C. WIGAND, 1989. Effect of "brown tide" shading on eelgrass (*Zostera marina* L.) distributions. In: COSPER, E.M., BRICELJ, V.M. & E.J. CARPENTER (Eds). Novel Phytoplankton Blooms: Causes and Impacts of Recurrent Brown tides and Other Unusual Blooms. Springer-Verlag, Berlin : 675-692.

DETHIER, M.N., 1984. Disturbance and recovery in intertidal pools: maintenance of mosaic patterns. *Ecol. Monogr.*, **54**: 99-118

DETHIER, M.N., 1990. *A marine and estuarine habitat classification system for Washington State.* Washington Natural Heritage Program, Department of Natural Resources, Olympia, WA, 56pp.

DETHIER, M.N., GRAHAM, E.S., COHEN, S. & L.M. TEAR, 1993. Visual versus random-point percentage cover estimations: 'objective' is not always better. *Mar. Ecol. Prog. Ser.*, **96**: 93-100.

DEVINNY, J.S. & L.A. VOLSE, 1978. Effects of sediments on the development of *Macrocystis pyrifera* gametophytes. *Mar. Biol.*, **48**: 343-348.

DIECHMANN, G.S., 1978. *Aspects of growth and reproduction of Laminaria pallida (Grev.) J. Ag. off the Cape Peninsula*. MSc Thesis, University of Cape Town, 98pp.

DIECHMANN, G.S., 1980. Aspects of the ecology of *Laminaria pallida* (Grev.) J. Ag. off the Cape Peninsula (South Africa). I. Seasonal Growth. *Bot. Mar.*, **23**: 579-585.

EBLING, F.J., SLEIGH, M.A., SLOANE, J.F. & J.A. KITCHING, 1960. The Ecology of Lough Ine: VII. Distribution of some common plants and animals of the littoral and shallow sublittoral. *J. Ecol.*, **48**: 25-53.

EDWARDS, M.S., 1998. Effects of long-term kelp canopy exclusion on the abundance of the annual alga *Desmarestia ligulata* (Light F). *.J Exp. Mar. Biol. Ecol.,* **228**: 309–326.

EEKHOUT, S., RAUBENHEIMER, C.M., BRANCH, G.M., BOSMAN, A.L. & M.O. BERGH, 1992. A holistic approach to the exploitation of intertidal stocks: limpets as a case study. *S. Afr. J. Mar. Sci*., **12**: 1017-1029.

EGGLESTON, D., 1972. Factors influencing the distribution of sub-littoral ectoprocts of the south of the Isle of Man (Irish Sea). *J. Nat. Hist.*, **6**: 247-260.

ELNER, R.W. & R.L. VADAS Sr, 1990. Inference in ecology: the sea urchin phenomenon in the northwestern Atlantic. *Am. Nat*., **136**: 108–125.

EMANUEL, B.P., BUSTAMANTE, R.H., BRANCH, G.M., EEKHOUT, S. & F.J. ODENDAAL, 1992. A zoogeographic and functional approach to the selection of marine reserves on the west coast of South Africa. *S. Afr. J. Mar. Sci*., **12**: 341-354.

ENGLEDOW, H.R. & J.J. BOLTON, 1994. Seaweed alpha-diversity within the lower eulittoral zone in Namibia: The effects of wave action, sand inundation, mussels and limpets. *Botanica Mar.*, **37**:267-276.

FIELD, J.G., CLARKE, K.R. & R.M. WARWICK, 1982. A practical strategy for analysing multispecies distribution patterns. *Mar. Ecol. Prog. Ser.*, **8**: 37-52.

FIELD, J.G. & C.L. GRIFFITHS, 1991. Littoral and sublittoral ecosystems of southern Africa. In: MATHIESON, A.C. & P.H. NIENHUIS (Eds), *Ecosystems of the world*. Elserivier, Amsterdam, **24**: 323-346.

FIELD, J.G., JARMAN, N.G., DIECKMANN, G.S., GRIFFITHS, C.L., VELIMIROV, B. & P. ZOUTENDYK, 1977. Sun, Waves, Seaweed and Lobsters: The Dynamics of a West Coast Kelp-bed. *S. Afr. J. Sci.*, **73**: 7-10.

FIELD, J.G., LINLEY, E.A. CARTER, R.A. & P. ZOUTENDYK, 1980. Upwelling in a nearshore marine ecosystem and its biological implications. Estuar. coast. mar. Sci., 11: 133-150.

FIGUEIREDO, M.A.de O., KAIN, J.M. & T.A. NORTON, 2000. Responses of crustose corallines to epiphyte and canopy cover. *J. Phycol.,* **36**: 17–24.

FOSTER, M.S., 1975. Algal succession in a *Macrocystis pyrifer*a forest. *Mar. Biol.*, **32**: 313-329.

FOSTER, M.S. & D.C. BARILOTTI, 1990. An approach to determining the ecological effects of seaweed harvesting: a summary. *Hydrobiologia*, **204/205**: 15-16.

FRICKE, A.H., 1979. Kelp grazing by the common sea urchin *Parechinus angulosus* Leske in False Bay, Cape. *S. Afr. J. Zool*., **14(3)**: 143-148.

GARDNER, C., FRUSHER, S.D., KENNEDY, R.B. & A. CAWTHORN, 2001. Relationship between settlement of southern rock lobster pueruli, *Jasus edwardsii*, and recruitment to the fishery in Tasmania, Australia. *Mar. Freshw. Res.,* **52**: 1271-1275.

GARDNER, C., MILLS, D., IBBOTT, WILCOX, S & B. CREAR, 2000. Preliminary investigation towards ongrowing puerulus to enhance rock lobster stocks while providing animals for commercial culture. *Fish. Res. Rep. - FRDC Project 1999/314* **June 2000**: 57pp.

GASTRICH, M.D., 2000. Harmful Algal Blooms in coastal waters of New Jersey. New Jersey Department Of Environmental Protection, Division Of Science, Research And Technology. pp34.

GENOVESE, S.J. & J.D. WITMAN, 1999. Interactive effects of flow speed and particle concentration on growth rates of an active suspension feeder. *Limnol. Oceanogr*., **44(4)**: 1120-1131.

GERARD, V.A., 1976. *Some aspects of material dynamics and energy flow in a kelp forest in Monterey Bay, Calirornia*. PhD Thesis, University of California, Santa Cruz, 173pp.

GERARD, V.A. & K.H. MANN, 1979. Growth and production of *Laminaria longicruris* (Phaeophyta) populations exposed to different intensities of water movement. *J. Phycol*., **15**: 33–41.

GERRODETTE, T. & A.O. FLECHSIG, 1979. Sediment-induced reduction in the pumping rate of the tropical sponge *Verongia lacunosa*. *Mar. Biol*., **55**: 103-110.

GRIFFITHS, C.L. & R.J. GRIFFITHS, 1987. Bivalvia. pp 1-88. *In:* PANDIAN, T.J. & F.J. VERNBERG (ed.) *Animal Energetics*, Academic Press, New York, NY.

GRIFFITHS, C.L., HOCKEY, P.A.R., VAN ERKOM SCHURINK, C. & P.J.L. ROUX, 1992. Marine invasive aliens on South African shores: implications for community structure and trophic functioning. *S. Afr. J. mar. Sci*., **12**: 713-722.

GREENGRASS, C., 2007. Settlement patterns, biology and collection ofpueruli and early juvenile stages of the West Coast rock lobster Jasus lalandii. MSc Thesis, University of Cape Town.

GROBLER, C.A.F. & H.O.N. NDJAULA, 2001. Namibian *Jasus lalandii* recruitment: size, weight and growth of pueruli and early juveniles. *Mar. Freshw. Res.,* **52**: 1271-1281.

GUZMÁN-DEL PRÓO, S.A., CARRILO-LAGUNA, J., BELMAR-PÉREZ, J., DE LE CAMPA, S. & A.B. VILLA, 1996. -The puerulus settlement of red spiny lobster (*Panulirus interruptus*) in Bahía California, Mexico. *Crustaceana,* **69(8)**: 949-957.

HARROLD, C. & J.S. PEARSE, 1987. The ecological role of echinoderms in kelp forests. In: JANGOUX, M. & J.M. LAWRENCE (Eds), Echinoderm Studies. A.A. Balkema, Rotterdam. **2**: 137–233.

HAUPT, P., BROUWER, S.L., BRANCH, G.M. & G. GÄDE, 2006. Effects of exposure to air on the escape behaviour and haemolymph chemistry of the South African Cape lobster, *Jasus lalandii*. *Fisheries Research*, **81**: 210–218.

HAWKINS, S.J. & R.G. HARTNOLL, 1983. Grazing of intertidal algae by marine invertebrates. *Oceanogr. Mar. Biol. Ann. Rev.*, **21**: 195-282.

HAWKINS, A.J.S., JAMES, M.R., HICKMAN, R.W., HATTON, S. & M. WEATHERHEAD, 1999. Modelling of suspension-feeding and growth in the green-lipped mussel *Perna canaliculus* exposed to natural and experimental variations of seston availability in the Marlborough Sounds, New Zealand. *Mar. Ecol. Prog. Ser*., **191**: 217-232.

HAY, C.H. & G.R. SOUTH, 1981. Some implications of field experiments on proposed commercial harvesting of *Durvillaea* in New Zealand. *Proc. int. Seaweed Symp.*, **8**: 699-712.

HEYDORN, A.E.F., 1969. The rock lobster of the South African West Coast *Jasus lalandii* (H. Milne-Edwards). 2. Population studies, behaviour, reproduction, moulting, growth and migration. *Investl. Rep. Div. Sea Fish. S. Afr.*, **71**: 1-52.

HAZELL, R.W.A, COCKCROFT, A.C., MAYFIELD, S. & M. NOFFKE, 2001. Factors influencing the growth rate of juvenile rock lobsters, *Jasus lalandii*. *Mar. Fresw. Res.,* **52**: 1367-1373.

HERNÁNDEZ-CARMONA, G., 1996. Frond elongation rates of *Macrocystis pyrifera* (L.) Ag. at Bahía Tortugas, Baja California sur, Mexico. *Ciencias Marinas*, **22(1)**: 57-72.

HERRNKIND, W.F. & M.J. BUTLER IV, 1986. Factors regulating postlarval settlement and juvenile microhabitat use by spiny lobsters, *Panulirus argus*. *Mar. Ecol. Prog*. *Ser*., **34**: 23-30.

HERRNKIND, W.F., M.J. BUTLER IV & R.A. TANKERSLEY, 1988. The effects of siltation on recruitment of spiny lobsters, *Panulirus argus*. *Fish. Bull.*, **86(2)**: 331-338.

HILBORN, R. & C.J. WALTERS, 1992. *Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty*. New York; Chapman & Hall: xv + 570 pp.

HISCOCK, K., 1983. Water movement. In: EARLL, R. & D.G. ERWIN (Eds). *Sublittoral Ecology: the Ecology of the Shallow Sublittoral Benthos*. Clarendon Press, Oxford, pp. 58-96.

HOCKEY, P.A.R. & A.L. BOSMAN, 1986. Man as an intertidal predator in Transkei: disturbance, community convergence and management of a natural food resource. *Oikos,* **46**: 3-14.

HOWARTH, R., ANDERSON, D., CLOERN, J., ELFRING, C., HOPKINSON, C., LAPOINTE, B., MALONE, T., MARCUS, N., McGLATHERY, K., SHARPLEY, A. & D. WALKER, 2000. Nutrient Pollution of Coastal Rivers, Bays, and Seas. *Issues in Ecology*, **7**: 1-15.

HUDON, C., 1987. Ecology and growth of post-larval and juvenile lobster, *Homarus americanus*, off isle de la Madeleine (Quebec). *Can. J. Fish. Aquat. Sci.*, **44**: 1855-1869.

HUNT, H.L., & R.E. SHEBLING, 1997. Role of post-settlement mortality in recruitment of benthic marine invertebrates. *Mar. Ecol. Prog. Ser*., **155**: 269–301.

IRVING, A.D. & S.D. CONNELL, 2006. Physical disturbance by kelp abrades erect algae from the understorey. *Mar. Ecol. Prog. Ser.*, **324**: 127–137.

IRVING, A.D., CONNELL, S.D., JOHNSTON, E.L., PILE, A.J. & B.M/ GILLANDERS, 2005. The response of encrusting coralline algae to canopy loss: an independent test of predictions on an Antarctic coast. *Mar. Biol.*, **147(5)**: 1075-1083.

IRVING, A.D., CONNELL, S.D. & T.S. ELSDON, 2004. Effects of kelp canopies on bleaching and photosynthetic activity of encrusting coralline algae. *J. Exp. Mar. Biol. Ecol.*, **310(1)**: 1-12.

JACKSON, G.A. 1983. The physical and chemical environment of a kelp community. In: BASCOM, W. (ed.). The effects of waste disposal on kelp communities. Southern California Coastal Water Research Project, Long Beach. pp 11–37.

JACKSON, G.A., 1984. Internal wave attenuation by coastal kelp stands. *J. Phys. Oceanogr.*, **14**: 1300-1306.

JACKSON, G.A., 1987. Modelling the growth and harvest yield of giant kelp *Macrocystis pyrifera*. *Mar. Biol.*, **95**: 611-624.

JACKSON, G.A. & C.D. WINANT, 1983. Effect of a kelp forest on coastal currents. *Cont. Shelf Rep.*, **2**: 75-80.

JARMAN N.G. & R.A. CARTER, 1981. The primary producers of the inshore regions of the Benguela. *Trans. Roy. Soc. S. Afr.*, **44(3)**: 321-325.

JEFFS, A. & S. HOOKER, 2000. Economic feasibility of aquaculture of spiny lobsters *Jasus edwardsii* in temperate waters. *J. World Aquac. Soc.,* **31**(1): 30-41.

JENKINS, S.R., NORTON, T.A. & S.J. HAWKINS, 1999. Settlement and post-settlement interactions between *Semibalanus balanoides* (L.) (Crustacea: Cirripedia) and three species of fucoid canopy algae. *J. Exp. Mar. Biol. Ecol.,* **236**: 49–67.

JOHNSON, C.R. & K.H. MANN, 1988. Diversity, patterns of adaptation, and stability of Nova Scotia kelp beds. *Ecol. Monogr.*, **58**: 129-154.

JOSKA, M.A.P. & J.J. BOLTON, 1987. *In situ* measurement of zoospore release and seasonality of reproduction in *Ecklonia maxima* (Alariacea, Laminariales). *British Phycological Journal*, **22**: 209-214.

KAIN, J.M., 1977. The biology of *Laminaria hyperborea* X. The effect of depth on some populations. *J. mar. Biol. Assoc. U.K.*, **57**: 587-607.

KENNELLY, S.J., 1989. Effects of kelp canopies on understorey species due to shade and scour. *Mar. Ecol. Prog. Ser.*, **50**: 215–224.

KEULDER, F.J., 2005. *Puerulus and early juvenile recruitment of the rock lobster Jasus lalandii in relation to the environment at Lüderitz Bay, Namibia.* MSc Thesis, Rhodes University, 125pp.

KONAR, B. & J.A. ESTES, 2003. The stability of boundary regions between kelp beds and deforested areas. *Ecology*, **84**: 174–185.

KENDRICK, G.A., 1991. Recruitment of coralline crusts and filamentous turf algae in the Galapagos Archipelago: effect of simulated sand scour, erosion and accretion. *J. Exp. Mar. Biol. Ecol.,* **147**: 47-63.

KENT, M. & P. COKER, 1992. *The description of vegetation in the field. In: Vegetation Description and Analysis*. Belhaven Press, London: p45.

KIMURA, R.S. & M.S. FOSTER, 1984. The effects of harvesting *Macrocystis pyrifera* on the algal assemblage in a giant kelp forest. *Hydrobiologia*, **116/117**: 425-428.

KITCHING, J.A., MACAN, T.T. & H.C. GILSON, 1934. Studies in sublittoral ecology: I. A submarine gully in Wembury Bay, South Devon. *J. mar. Biol. Assoc. U.K.,* **19**: 677– 705.

LADAH, L.B., J.A. ZERTUCHE-GONZÁLEZ & G. HERNÁNDEZ-CARMONA, 1999. Giant kelp (*Macrocystis pyrifera*, Phyophyceae) recruitment near its southern limit in Baja Calafornia after disappearance during ENSO 1997-1998. *J. Phycol.*, **35**: 1106-1112.

LANE, S.B. & R.A. CARTER, 1999. *Generic Environmental Management Programme for Marine Diamond MIning off the West Coast of South Africa.* Marine Diamond Mines Association, Cape Town, South Africa. 6 Volumes.

LAZARUS, B.I., 1967. The occurrence of phyllosoma larvae off the Cape, with particular reference to *J. lalandii*. *Investl. Rep. Div. Sea. Fish. S. Afr.,* **63**: 38pp.

LEMMENS, J.W.T.J., 1994. Biochemical evidence for absence of feeding in puerulus larvae of the western rock lobster (*Panulirus cygnus*; Decapoda: Palinuridae). *Mar. Biol. (Berlin)* **118**: 383-391.

LEONARD, G.H., 1999. Positive and negative effects of intertidal algal canopies on recruitment and survival of barnacles. *Mar. Ecol. Prog. Ser.*, **178**: 241–249.

LEVITT, G.J., ANDERSON, R.J., BOOTHROYD, C.J.T. & F.A. KEMP, 2002. The effects of kelp harvesting on its regrowth and the understorey benthic community at Danger Point, South Africa, and a new method of harvesting kelp fronds. *S. Afr. J. Mar. Sci*., **24**: 71-85.

LIPCIUS, R.N. & D.B. EGGLESTON, 2000. Ecology and fishery biology of spiny lobster. In: PHILLIPS, B.F & J. KITTAKA (Eds), *Spiny Lobsters: Fisheries and Culture*. Oxford, Fishing News Books, Blackwell Science Publications: 1-41.

LITTLER, M.M., 1980. The effects of recurrent sedimentation on rocky intertidal macrophytes. *J. Phycol.*, **16**: 26.

LITTLER, M. M. & D.S. LITTLER, 1981. Intertidal macrophyte communities from Pacific Baja California and the upper Gulf of California: relatively constant vs. environmentally fluctuating systems. *Mar. Ecol. Prog. Ser.*, **4**: 145-158.

LITTLER, M.M., MARTZ, D.R. & D.S. LITTLER, 1983. Effects of recurrent sand deposition on rocky intertidal organisms: importance of substrate heterogeneity in a fluctuating environment. *Mar. Ecol. Prog. Ser.*, **11**: 129-139.

LITTLER, M.N. & S.N. MURRAY, 1975. Impact of sewage on the distribution, abundance and community structure of rocky intertidal macro-organisms. *Mar. Biol.,* **30**: 277-291.

LØVAS, S.M. & A. TØRUM, 2001. Effect of the kelp *Laminaria hyperborea* upon sand dune erosion and water particle velocities. *Coastal Engineering*, **44**: 37–63.

MACDIARMID, A., 2000. Enhanced ecology of *Jasus edwardsii*. In: *Workshop on Post-settlement Processes Affecting the Southern Rock Lobster, Jasus edwardsii, in Southern Australia*. TAFI, University of Tasmania: 4-5.

MANN, K.H., 1972. Ecological energetics of the seaweed zone in a marine bay on the Atlantic Coast of Canada. I. Zonation and biomass of seaweeds. *International Journal on Life in Oceans and Coastal Waters*. **12(1)**: 1-10.

MANN, K.H., 1973. Seaweeds: Their productivity and strategy for growth. *Science*, **182(4116)**: 975-981.

MANN, K.H., JARMAN, N. & G. DIECKMANN, 1979. Development of a method for measuring the productivity of the kelp *Ecklonia maxima* (Osbeck) Papenf. *Trans. Roy. Soc. S. Afr.*, **44(3)**: 27-41.

MARKHAM, J.W., 1973. Observations of the ecology of *Laminaria sinclarii* on three northern Oregon beaches. *J. Phycol.*, **9**: 336-341.

MARSHALL, D.J. & C.D. MCQUAID, 1989. The influence of respiratory responses on the tolerance to sand inundation of the limpets *Patella granularis* L (Prosobranchia) and *Siphonaria Capensis* Q et G. (Pulmonata). *J. Exp. Mar. Biol. Ecol*., **128**: 191-201.

MARSHALL, D.J. & C.D. MCQUAID, 1993. Differential physiological and behavioural responses of the intertidal mussels, *Choromytilus meridionalis* (Kr.) and *Perna perna* L., to exposure to hypoxia and air: a basis for spatial separation. *J. Exp. Mar. Biol. Ecol*., **171**: 225-237.

MARSZALEK, D.S., 1981. Impact of dredging on a subtropical reef community, Southeast Florida, USA. *Proc. 4th Int. Coral Reefs Congr.*, **1**:147-153.

McCLENEGHAN, K. & J.L. HOUK, 1985. The effect of canopy removal on holdfast growth in *Macrocystis pyrifera* (Phaeophyta: Laminariales). *Calif. Fish and Game*, **71(1)**: 21-27.

McLAIN, P. & M. McHALE, 1996. Barnegat Bay eelgrass investigation 1995-96. *Barnegat Bay Ecosystem Workshop. Toms River, NJ. 11/14/96.*

McQUAID, C.D. & DOWER, K.M., 1990. Enhancement of habitat heterogeneity and species richness on rocky shores inundated by sand. *Oecologia,* **84**: 142-144.

McWILLIAM, P.S. & B.F. PHILLIPS, 1997. Metamorphosis of the final phyllosoma and secondary lecithotrophy in the puerulus of *Panulirus cygnus* George: a review. *Mar. Freshw. Res*., **48**: 783-790.

MELVILLE-SMITH, R., THOMPSON, A.W. & N. CAPUTI, 2004. Improved forecasts of recreational western rock lobster (*Panulirus cygnus*) catches in Western Australia, by predicting licence usage. *Fish. Res.,* **68**: 203-208.

MILLER, D.J. & J.J. GEIBEL, 1973. Summary of blue rockfish and lingcod life histories: a reef ecology study; and giant kelp, *Macrocystis pyrifera*, experiments in Monterey Bay, California. *Fish. Bull*., **158**: 137pp.

MILLER, R.J. & A.G. COLODEY, 1983. Widespread mass mortalities of the green sea urchin in Nova Scotia, Canada. *Mar. Biol*., **73**: 263-267.

MOLLOY, F.J., 1992. *Studies on the Ecology and Production of Seaweeds of Economic and Potentially Economic Importance on the Namibian Coast*. PhD Thesis, University of Cape Town, pp253.

MOLLOY, F.J. & J.J. BOLTON, 1996. The effects of wave exposure and depth on the morphology of inshore populations of the Namibian kelp I Foslie. *Bot. Mar*., **39**: 525–531.

MURRAY, L.G., JOYNT, R.H., O'SHEA, D.O'C., FOSTER, R.W. & L. KLEINJAN, 1971. The geological environment of some diamond deposits off the coast of South West Africa. In: DELANY, F.M. (Ed.), The geology of the East Atlantic continental margin. *Rep. Inst. Geol. Sci*., **70(13)**: 119-141.

NEUSHAL, M., 1971. The species of *Macrocystis* with particular references to those of North and South America. *Nova Hedwigia Beihefte,* **32**: 211-222.

NEUSHAL, M., FOSTER, M.S., COON, D.A. WOESSNER, J.W. & B.W. HARGER, 1976. An *in situ* study of recruitment, growth and survival of subtidal marine algae: techniques and preliminary results. *J. Phycol.*, **12**: 397-408.

NEWELL, R.C., M.I. LUCAS, B. VELIMIROV & L.J. SEIDERER, 1980. Quantitative significance of dissolved organic losses following fragmentation of kelp (*Ecklonia maxima* and *Laminaria pallida*). *Mar. Ecol. Prog. Ser*., **2**: 45-49.

NEWMAN, G.G. & D.E. POLLOCK, 1971. Biology and migration of rock lobster *Jasus lalandii* and their effect on availability at Elands Bay, South Africa. *Investl. Rep. Div. Sea Fish. S. Afr.*, **94**: 1-24.

NOLI, K.R. & C.A.F. GROBLER, 1998. Seasonal migration of Namibian Rock-lobster. *Namibia Brief*, **20**: 126-128.

NORDERHAUG, K.M., CHRISTIE, H. & E. RINDE, 2002. Colonization of kelp imitations by epiphyte and holdfast fauna; a study of mobility patterns. *Mar. Biol.*, **141**: 965–973.

NORTH, W.J., 1957. Cutting of kelp. In: NORTH, W.J. (ed.), *Kelp Investigations Program Annual Report for the Period 1 July 1956-30 June 1957*. Univ. Calif. Inst. Mar. Resour., La Jolla, pp20-24.

NORTH, W.J., 1971. The biology of giant kelp beds (*Macrocystis*) in California. *Nova Hedwigia Beihefte,* **32**: 1-60.

NORTH, W.J., JAMES, D.E. & L.G. JONES, 1993. History of kelps (*Macrocystis*) in Orange and San Diego Counties, California. *Hydrobiologia*, **260/261**: 277-283.

NORTON, T.A., 1994. Essay: The rocky intertidal zone. In: LOBBAN, C.S. & P.J. HARRISON (Eds), Seaweed Ecology and Physiology. Cambridge University Press.

NORTON, T.A., HISCOCK, K. & J.A. KITCHING, 1977. The ecology of Lough Ine. XX. The *Laminaria* forest at Carrigathorna. *J. Ecol.*, **65**: 919-941.

NORTON, T.A., MATHIESON, A.C. & M. NEUSHUL, 1982. A review of some aspects of form and function in seaweeds. *Bot. Mar.*, **25**: 501-510.

NORTON, T.A. & G.R. SOUTH, 1969. Influence of reduced salinity on the distribution of two laminarian algae. *Oikos*, **20**: 320-326.

OVERNELL, J. & S. YOUNG, 1995. Sedimentation and carbon flux in a Scottish sea lough, Loch Linnhe. *Estuar. Coast. Shelf. Sci.*, **41**: 361–376.

PARRISH, F.A. & J.J. POLOVINA, 1994. Habitat thresholds and bottlenecks in production of the spiny lobster (*Panulirus marginatus*) in the northwestern Hawaiian Islands. *Bull. Mar. Sci*., **54(1)**: 151-163.

PHILLIPS, B.F., CRUZ, R., CAPUTI, N. & R.S. BROWN, 2000a. Predicting the catch of spiny lobster fisheries. In: PHILLIPS, B.F. & J. KITTAKA (Eds.), *Spiny Lobsters Fisheries and Culture*. Oxford, Blackwell Scientific Publications: 357–375.

PHILLIPS, B.F., PEARCE, A.F., LITCHFIELD, R. & S. GUZMAN DEL PROO, 2000b. Spiny lobster catches and the ocean environment. In PHILLIPS, B.F. & J. KITTAKA (Eds.), *Spiny Lobsters Fisheries and Culture*. Oxford, Blackwell Scientific Publications: 321–333.

PHILLIPS, B.F., MELVILLE-SMITH, R., ROSSBACH, M., CHENG, Y.W., CAPUTI, N., THOMPSON, A.W., MILLS, D. & B. CREAR, 2003. Towards establishing techniques for large scale harvesting of pueruli and obtaining a better understanding of mortality rates. *Fish. Res. Rep. - FRDC Project 1998/302* **144**: 140pp.

PHILLIPS, B.F. & J.D. BOOTH, 1994. Design, use and effectiveness of collectors for catching the puerulus stage of spiny lobsters. *Rev. Fish. Sci.,* **2(3)**: 255-289.

PISCES ENVIRONMENTAL SERVICES (PTY) LTD, 2004. *Marine Dredging Project Pre-Feasibility Study, Environmental Impact Assessment, Volume I: Scoping Report, Environmental Impact Report & EMP*. Prepared for De Beers Marine (Pty) Ltd. and Namdeb Diamond Corporation (Pty) Ltd.

PISCES ENVIRONMENTAL SERVICES (PTY) LTD., 2005a. BEHP/CEA/03/04: Assessment of cumulative impacts of scouring of sub-tidal areas and kelp cutting by diamond divers in near-shore areas of the BCLME region. Progress Report I. May 2005, pp40.

PISCES ENVIRONMENTAL SERVICES (PTY) LTD., 2005b. BEHP/CEA/03/04: Assessment of cumulative impacts of scouring of sub-tidal areas and kelp cutting by diamond divers in near-shore areas of the BCLME region. Progress Report II. October 2005, pp28.

PISCES ENVIRONMENTAL SERVICES (PTY) LTD., 2006a. BEHP/CEA/03/04: Assessment of cumulative impacts of scouring of sub-tidal areas and kelp cutting by diamond divers in near-shore areas of the BCLME region. Progress Report III. June 2006, pp37.

PISCES ENVIRONMENTAL SERVICES (PTY) LTD., 2006b. BEHP/CEA/03/04: Assessment of cumulative impacts of scouring of sub-tidal areas and kelp cutting by diamond divers in near-shore areas of the BCLME region. Progress Report III. October 2006, pp7.

PISCES ENVIRONMENTAL SERVICES (PTY) LTD., in prep. BEHP/CEA/03/02: Data Gathering and Gap Analysis for Assessment of Cumulative Effects of Marine Diamond Mining Activities on the BCLME Region.

PITCHER, G.C., 1998. *Harmful algal blooms of the Benguela Current.* IOC, World Bank and Sea Fisheries Research Institute Publication. 20 pp.

POLLOCK, D.E., 1973. Growth of juvenile rock lobster *Jasus lalandii* (H. Milne Edwards). *Investl. Rep. Div. Sea Fish. S. Afr.*, **106**: 1-16.

POLLOCK, D.E., 1978. *Growth and production rates of the rock lobster Jasus lalandii (H. Milne Edwards).* Ph.D thesis, University of the Witwatersrand, pp 1-226.

POLLOCK, D.E., 1982. The fishery for and population dynamics of West Coast rock lobster related to the environment in the Lambert's Bay and Port Nolloth areas. *Investl. Rep. Div. Sea Fish. S. Afr.*, **124**: 1-57.

POLLOCK, D.E., 1987. Simulation models of rock-lobster populations from areas of widely divergent yields on the Cape west coast. In: PAYNE, A.I.L., GULLAND, J.A. & K. H. BRINK (Eds). *The Benguela and Comparable Ecosystems*. *S. Afr. J. Mar. Sci*., **5**: 531-545.

POLLOCK, D.E., 1989. Spiny lobsters. In: PAYNE, A.I.L. & R.J.M. CRAWFORD (Eds), *Oceans of life off Southern Africa*. CapeTown; Vlaeberg: 70-80.

POLLOCK, D.E., 1990. Palaeoceanography and speciation in the spiny lobster genus *Jasus*. *Bull. Mar. Sci.,* **46(2)**: 387-405.

POLLOCK, D.E. & C.J. DE B. BEYERS, 1979. Trap selectivity and seasonal catchability of rock lobster *Jasus lalandii* at Robben Island Sanctuary, near Cape Town. *Fish. Bull. S. Afr.*, **12**: 75-79.

POLLOCK, D.E. & C.J. DE B. BEYERS, 1981. Environment, distribution and growth rates of West Coast rock lobster *Jasus lalandii* (H. Milne Edwards). *Trans. Roy. Soc. S. Afr.*, **44**: 379-400.

POLLOCK, D.E. & L.V. SHANNON, 1987. Response of rock lobster populations in the Benguela ecosystem to environmental change – A hypothesis. *S. Afr. J. mar. Sci.*, **5**: 887-899.

PULFRICH, A., 1998. *Assessment of the impact of diver-assisted nearshore diamond mining on marine benthic communities in the Kerbe Huk Area, Namibia*. Report to NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia, 29 pp.

PULFRICH, A., 2006. Survey of intertidal and subtidal rocky shore communities at Elizabeth Bay: Intertidal and subtidal monitoring report – 2006. *Report to NAMDEB Diamond Corporation (Pty) Ltd., Oranjemund, Namibia*, 39pp.

PULFRICH, A., C.A. PARKINS & G.M. BRANCH, 2003a. The effects of shore-based diamond-diving on intertidal and subtidal biological communities and rock-lobsters in southern Namibia. *Aquatic Conserv: Mar Freshw. Ecosyst.*, **13**: 257-278.

PULFRICH, A., C.A. PARKINS, G.M. BRANCH, R.H. BUSTAMANTE & C.R. VELÁSQUES, 2003b. The effects of sediment deposits from Namibian diamond mines on intertidal and subtidal reefs and rock-lobster populations. *Aquatic Conserv: Mar Freshw. Ecosyst.,* **13**: 233-255.

PULFRICH, A. & A.J. PENNEY, 2001. Assessment of the impact of diver-operated nearshore diamond mining on marine benthic communities near Lüderitz, Namibia. Phase III. *Report to NAMDEB Diamond Corporation (Pty) Ltd*, 50pp.

PULFRICH, A. & A.J. PENNEY, 2006. Marine Dredging Project: Kelp Bed Aerial Survey. Infra-red Aerial Survey of the Kelp Beds along the Southern Namibian Coast. *Prepared for De Beers Marine Namibia, July 2006*. 37pp.

PULFRICH, A., PENNEY, A.J., BRANDÃO, A., BUTTERWORTH, D.S. & M. NOFFKE, 2006. Marine Dredging Project: FIMS Final Report. Monitoring of Rock Lobster Abundance, Recruitment and Migration on the Southern Namibian Coast. *Prepared for De Beers Marine Namibia, July 2006*. 149pp.

RALPH, P.J., MORRISON, D.A. & A. ADDISON, 1998. A quantitative study of the patterns of morphological variation within *Hormosira banksii* (Turner) Decaisne (Fucales: Phaeophyta) in south-eastern Australia. *J. Exp. Mar. Biol. Ecol*., **225**: 285–300.

RAND, A.M., 2006. *Using Geographic Information Systems and Remote Sensing to imporove the management of kelp resources in South Africa.* MSc Thesis, University of Cape Town

REED, D.C., 2000. Microecology of macroalgal blooms. *J. Phycol.*, **36**: 1–2.

REED, D.C. & M.S. FOSTER, 1984. The effects of canopy shading on algal recruitment and growth in a giant kelp forest. *Ecology*, **65**: 937–948.

REISWIG, H.M., 1971a. *In situ* pumping activities of tropical Desospongiae. *Mar. Biol.*, **9**: 38-50.

REISWIG, H.M., 1971b. Particle feeding in natural populations of three marine desmosponges. *Biol. Bull.*, **141**: 568-591.

RINDE, E., CHRISTIE, H., FREDRIKSEN S. & A. SIVERTSEN, 1992. Ecological consequences of kelp trawling: Importance of the structure of the kelp forest for abundance of fauna in the kelp holdfasts, benthic fauna and epiphytes. *NINA Oppdragsmelding*, **127**: 1–37. (in Norwegian with English summary).

RIVER, G.F. & P.J. EDMUNDS, 2001. Mechanisms of interactions between macroalgae and scleractinians on a coral reef in Jamaica. *J. Exp. Mar. Biol. Ecol.,* **261**: 159–172.

ROBERSON, L.M. & J.A.COYER, 2004. Variation in blade morphology of the kelp *Eisenia arborea*: incipient speciation due to local water motion? *Mar. Ecol. Prog. Ser*., **282**: 115–128.

ROBLES, C., 1982. Distribution and predation in an assemblage of herbivorous diptera and algae on rocky shores. *Oecologia*, **54**: 23-31.

RODRÝGUEZ, S.R., OJEDA, F.P. & N.C. INESTROSA, 1993. Settlement of benthic marine invertebrates. *Mar. Ecol. Prog. Ser.*, **97**: 193–207.

ROGERS, C.S., 1990. Responses of coral reefs and reef organisms to sedimentation. *Mar. Ecol. Prog. Ser.*, **62**: 185-202.

ROGERS, J., 1977. *Sedimentation on the continental margin off the Orange River and the Namib Desert.* Unpubl. Ph.D. Thesis, Geol. Dept., Univ. Cape Town. 212 pp.

ROTHMAN, M.D., 2006. *Investigations into the Harvesting Ecology of the South African kelp Ecklonia maxima (Alariacea, Laminariales).* MSc Thesis, University of Cape Town, pp112.

RØV, N., CHRISTIE, H., FREDRIKSEN, S., LEINAAS, H.P. & S.H. LORENTSEN, 1990. Biologiske forundersøkelser i forbindelse med planer om taretråling i Sør-Trøndelag. *NINA Oppdragsmelding*, **52**: 1-20. (in Norwegian with English summary).

SANTOS, C., 1993. A multivariate study of biotic and abiotic relationships in a subtidal algal stand. *Mar. Ecol. Prog. Ser.*, **94**: 181-190.

SCHEIBLING, R.E. & R.L. STEPHENSON, 1984. Mass mortality of *Strongylocentrotus droebachiensis* (Echinodermata: Echinoidea) off Nova Scotia, Canada. *Mar. Biol.*, **78**: 153-164.

SCHIEL, D.R., 1988. Algal interactions on shallow subtidal reefs in northern New Zealand: a review. *New Zealand J. mar. Fresh. Res.*, **22**: 481-489.

SCHIEL, D.R. & W.A. NELSON, 1990. The harvesting of macroalgae in New Zealand. *Hydrobiologia*, **204/205**: 25-33.

SCHIEL, D.R. & M.S. FOSTER, 1986. The structure of subtidal algal stands in temperate waters. *Oceanogr. Mar. Biol. Ann. Rev.*, **24**: 265-307.

SCHIEL, D.A., WOOD, S.A., DUNMORE, R.A. & D.I. TAYLOR, 2006. Sediment on rocky intertidal reefs: Effects on early post-settlement stages of habitat-forming seaweeds. *J. Exp. Mar. Biol. Ecol.*, **331**: 158– 172

SCHOCH, G.C., 1994. Geomorphological shoreline classificationand habitat sensitivity analysis for Katmai National Park and Preserve, Alaska. *1994 Proceedings, The Coastal Society, The 4th International Conference*, Charleston, SC, pp 4.

SCHOCH, G.C., 1996. *The classification of nearshore habitats: a spatial distribution model*. PhD thesis, College of Oceanic and Atmospheric Sciences, Oregon State University, pp 1-146.

SCHOCH, G.C. & M.N. DETHIER, 1996. Scaling up: the statistical linkage between organismal abundance and geomorphology on rocky intertidal shorelines. *J. Exp. Mar. Biol. Ecol.*, **201**: 37-72.

SCHROETER, S.C., DEAN, T.A., THIES, K. & J.D. DIXON, 1995. Effects of shading by adults on the growth of blade-stage *Macrocystis pyrifera* (Phaeophyta) during and after the 1982-1984 El Niño. *J. Phycol.*, **31**: 697-702.

SEAPY, R.R. & M.M. LITTLER, 1982. Population and species diversity fluctuations in a rocky intertidal community relative to severe aerial exposure and sediment burial. *Mar. Biol.*, **71**: 87-96.

SEYMORE, R.J., TEGNER, M.J., DAYTON, P.K. & P.E. PARNELL, 1989. Storm wave induced mortality of giant kelp, *Macrocystis pyrifera*, in southern California. *Estuar. Coast. Shelf Sci.*, **28**: 277–292.

SHANNON, L.V., 1985. The Benguela Ecosystem. Part 1. Evolution of the Benguela, physical features and processes. *Oceanogr. Mar. Biol. Ann. Rev.*, **23**: 105-182.

SHANNON, L.V. & M.J. O'TOOLE, 1998. *Intergrated overview of the oceanography and environmental variability of the Benguela Current region. Synthesis and Assessment of Information on the BCLME: Thematic Report 2*. UNDP/GEF (RAF/96/G43), pp. 58.

SHANNON L.V. & S. PILLAR. 1985. The Benguela Ecosystem III. Plankton. *Oceanogr. Mar. Biol. Ann. Rev.,* **24**: 65-170.

SHARP, G.J. & J.D. PRINGLE, 1990. Ecological impact of marine plants harvesting in the northwest Atlantic: a review. *Hydrobiologia*, **204/205**: 17-24

SIMENSTAD, C.A., ESTES, J.A. & K.W. KENYON, 1978. Aleuts, sea otters, and alternative stable state communities. *Science*, **200**: 403-411.

SIMONS, R.H. & N.G. JARMAN, 1981. Subcommercial harvesting of a kelp on a South African shore. *Proc. int. Seaweed Symp.*, **10** pp. 731-736.

SJØTUN, K., FREDRIKSEN, S., LEIN, T.-E., RUENESS, J. & K. SIVERTSEN, 1993. Population studies of *Laminaria hyperborea* from its northern range of distribution in Norway. *Hydrobiologia*, **260/261**: 215-221.

SLATTERY, M. & D. BOCKUS, 1997. Sedimentation in McMurdo Sound, Antarctica: A disturbance mechanism for benthic invertebrates. *Polar Biol.*, **18(3)**: 172-179.

SMITH, G.G., MOCKE, G.P. VAN BALLEGOOYEN, R., & C. SOLTAU, 2002. Consequences of Sediment Discharge from Dune Mining at Elizabeth Bay, Namibia. *J. Coastal Res.*, **18(4)**: 776-791.

SMITH, G., WIETZ, N., SOLTAU, C. & A. VILJOEN, 2006. Assessment of the Cumulative Effects of Sediment Discharges from On-shore and Near-shore Diamond Mining Activities on the BCLME. *CSIR Report No. CSIR/NRE/ECO/ER/2006/0093/C*.

SPANIER, E., 1994. What are the characteristics of a good artificial reef for lobsters? *Crustaceana*, **67**: 173-186.

STEGENGA, H., BOLTON, J.J. & R.J. ANDERSON, 1997. Seaweeds of the South African West Coast. *Contributions from the Bolus Herbarium, No. 18*. Creda Press, Cape Town. 655 pp.

STENECK, R.S., GRAHAM, M.H., BOURQUE, B.J., CORBETT, D., ERLANDSON, J.M., ESTES, J.A. & M.J. TEGNER, 2002. Kelp forest ecosystems: biodiversity, stability, resilience and future. *Environmental Conservation*, **29**: 436-459.

STEWART, J.G., 1983. Fluctuations in the quantity of sediments trapped among algal thalli on intertidal rock platforms in Southern California. *J. Exp. Mar. Biol. Ecol.*, **73**:205-211.

SVENDSEN, P., 1972. Noen observasjoner over taretråling og gjenvek av stortare, *Laminaria hyperborea*. *Fiskets Gang*, **22**: 448-460. (in Norwegian with English summary).

TARR, R.J.Q., 1993. *Stock assessment, and aspects of the biology of the South African abalone, Haliotis midae.* MSc Thesis, University of Cape Town, pp 156.

TAYLOR, D.I. & D.R. SCHIEL, 2003. Wave-related mortality in zygotes of habitat-forming algae from different exposures in southern New Zealand: the importance of "stickability". *J. Exp. Mar. Biol. Ecol.*, **290**: 229–245.

TAYLOR, P.R. & M.M. LITTLER, 1982. The roles of compensatory mortality, physical disturbance, and substrate retention in the development and organization of a sand-influenced, rocky-intertidal community. *Ecology,* **63**: 135-146.

TEGNER, M.J. & P.K. DAYTON, 1987. El Niño effects on Southern California kelp forest communities. *Adv. Ecol. Res*., **17**: 243–279.

TOMALIN, B.J., 1993. *Migrations of spiny rock lobsters,* Jasus lalandii*, at Lüderitz : Environmental causes, and effects on the fishery and benthic ecology.* MSc thesis, University of Cape Town, pp 1-99.

TOMALIN, B.J., 1995. Severity of potential impacts of mid-water diamond mining on the rock-lobster (*Jasus lalandii*) population and fishery at Marshall Reef, Lüderitz. *C.S.I.R. Unpublished Report*. 47 pp.

TOMALIN, B.J., 1996. Specialist study on impacts of diamond mining on rock-lobster populations and the fishery. *In*: Environmental Impact Assessment of Marine Diamond Mining in the Namibian Islands Concession, *CSIR Report EMAS-C96023*, 33 pp.

TROWBRIDGE, C.D., 1996. Demography and phenology of the intertidal green alga *Codium setchellii*: the enigma of local scarcity on sand-influenced rocky shores. *Mar. Biol.*, **127**: 341-351.

VADAS, R., JOHNSON, S. & T. NORTON, 1992. Recruitment and mortality of early post-settlement stages of benthic algae. *Br. Phycol. J.*, **27**: 331–351.

VAN TAMELEN, P.G., 1996. Algal zonation in tidepools: experimental evaluation of the roles of physical disturbance, herbivory and competition. *J. Exp. Mar. Biol. Ecol.*, **201**: 197-231.

VASQUEZ, J.A. & B. SANTELICES, 1990. Ecological effects of harvesting Lessonia (Laminariales, Phaeophyta) in central Chile. *Hydrobiologia*, **204/205**: 41-47.

VELIMIROV, B., FIELD, J.G., GRIFFITHS, C.L. & P. ZOUTENDYK, 1977. The ecology of kelp bed communities in the Benguela upwelling system. *Helgoländer Meeresunters*, **30**: 495-518.

VELIMIROV, B. & C.L. GRIFFITHS, 1979. Wave induced kelp movement and its importance for community structure. *Bot. Mar.*, **22**: 169-172.

WAAGE-NIELSEN, E., CHRISTIE, H. & E. RINDE, 2003. Short-term dispersal of kelp fauna to cleared (kelp-harvested) areas. *Hydrobiologia*, **503**: 77–91.

WAHLE, R.A. & R.S. STENECK, 1991. Recruitment habitats and nursery grounds of the American lobster (*Homarus americanus* Milne Edwards): a demographic bottleneck ?, *Mar. Ecol. Prog. Ser.*, **69**: 231-243.

WAHLE, R.A. & R.S. STENECK, 1992. Habitat restrictions in early benthic life: experiments on habitat selection and *in situ* predation with the American lobster. *J. Exp. Mar. Biol. Ecol.*, **157**: 91-114.

WERNBERG, T. & M.S. THOMSEN, 2005. The effect of wave exposure on the morphology of *Ecklonia radiata*. *Aquatic Botany*, **83**: 61–70.

WHITTICK, A., 1983. Spatial and temporal distribution of dominant epiphytes on the stipes of *Laminaria hyperborea* (Gunn.) Fosl. (Phaeophyta: Laminariales) in S. E. Scotland. *J. Exp. Mar. Biol. Ecol.*, **73**: 1–10.

YEUNG, C., JONES, D.L., CRIALES, M.M., JACKSON, T.L. & W.J. RICHARDS, 2001. Influence of coastal eddies and counter-currents on the influx of spiny lobster, *Panulirus argus*, postlarvae into Florida Bay. *Mar. Freshw. Res.*, **52**: 1217-1232.

YOSHIMURA, T., 2000. An experimental approach to post-settlement processes of *P. japonicus* using artificial structures. In: *Workshop on Post-settlement Processes Affecting the Southern Rock Lobster, Jasus edwardsii, in Southern Australia*. TAFI, University of Tasmania: 4-5.

ZIMMER-FAUST, R.K. & E. SPANIER, 1987. Gregariousness and sociality in spiny lobsters: implications for den habitation. *J. Exp. Mar. Biol. Ecol.*, **105**: 57-71.

ZIMMER-FAUST, R.K., TYRE, J.E. & J.F. CASE, 1985. Chemical attraction causing aggregation in the spiny lobster, *Panulirus interruptus*, and its probable ecological significance. *Biol. Bull. (Woods Hole, Mass.)*, **169**: 106-118.

ZOUTENDYK, P. & I.R. DUVENAGE, 1989. Composition and biological implications of a nepheloid layer over the inner Agulhas Bank near Mossel Bay, South Africa. *Trans. Roy. Soc. S. Afr.*, **47**: 187-197.

APPENDIX I CAPACITY BUILDING AND TRAINING

Capacity building, training and technology transfer among regions is a high priority in the BCLME Programme. A student scholarship at the MSc level was incorporated into this project, and after award of the contract, suitable students were actively sought by advertising the project on the SANCOR and BENEFIT mailing lists in April 2004, as well as at the various Universities. Only one response was received by an applicant who did not have the necessary BSc degree required to register for an MSc, and was therefore unsuitable.

During preparation of the project proposal, contact was made with potential academic supervisors at various Universities (UNam, UCT and UWC), and enquiries made as to availability of prospective students. During this initiative, Ms Janine Basson, who was employed as an intertidal researcher in the Environmental Section at MFMR in Swakopmund in February 2004, was identified as the most promising candidate. Contact was therefore made with Dr Ben van Zyl and Mr Chris Bartholomae at MFMR Swakopmund to negotiate the possibilities of Ms Basson's involvement in the Project. In subsequent discussions with Mr Bartholomae and in an e-mail received from him on 2 August 2004, we were informed that it would not be possible for MFMR to let Ms Basson take part in the project at an MSc level. The reasons given were that she was on probation until the end of 2004, had many other obligations towards the Ministry, and not being stationed in Lüderitz would find it difficult to meet both the MFMR and BCLME project commitments. Furthermore, it was pointed out that the MFMR would not be in a position to cover the travel expenses between Swakopmund and Lüderitz.

The project was also discussed telephonically with potential supervisors at the various universities and the proposal e-mailed to them with a request to inform us should they identify a suitable candidate amongst their students. No further responses were received. A possible reason for this is that the academic year was already well advanced and prospective students had already secured a position. It was also extremely difficult to secure prospective students without knowing whether or not the necessary long-term multiple entry work visas for the project team would be issued, and the project could be successfully initiated. Following the substantial delays associated with the application for work visas for the project team, and the uncertainty of how this would affect the successful initiation of the project, no further attempts were made to source a student.

During the initial survey in which the experiment was set up, it was, however, confirmed that substantial diving experience and competency is fundamental for the success of the experiment, as it requires extensive and intensive underwater work using heavy equipment and machinery, often under harsh and hazardous conditions. It would therefore have been crucial that the applicant have suitable scientific diving experience, and be capable of working proficiently and independently under harsh conditions. As a result, it was unfortunately not feasible to give consideration to a student who may require substantial supervision in the field. Due to the remoteness of the study area, and the frequency of the required field trips, constant guidance by either an academic supervisor or ourselves was not possible within the financial constraints of the project.

Furthermore, as field survey logistics require the survey team and/or student to visit the proposed survey area regularly, it was not feasible to involve anyone from Angola in this work due to visa constraints. Kelp bed habitats also do not extend beyond Rocky Point in northern Namibia. However, the results of the project, including descriptions of survey design, and recommendations for future monitoring programmes of this nature, will be made available should any nearshore marine mining be initiated in Angola, and similar capacity need to be developed in that country.

APPENDIX II MINISTRY AND INDUSTRY CO-OPERATION

As part of the tender proposal, several assumptions were made about the conditions necessary for the successful completion of the project. The most important of these was the prerequisite that the Namibian Ministry of Home Affairs grant multiple-entry work visas, valid for one year to all non-Namibian nationals on the project team. Despite letters of motivation from BCLME, as well as other supporting documentation accompanying the application, the Ministry did not meet our request. Substantial delay in the issuing the initial visas resulted in the postponement of the setting up of the field experiment, and subsequently postponement of all parts of the work plan. There were also delays in the issuing of mine clearance for the project team by the Ministry of Mines and Energy with the result that the interim surveys scheduled for the period between April and September could not be undertaken as planned.

Furthermore, as only 6-month visas were initially issued, it was necessary to re-apply for further visas, and consequently the interim surveys scheduled for between October and December could not be undertaken. Work visas valid until December 2006 were finally re-issued to the survey team on 14 December 2005.

A large number of puerulus collectors was required for the project, and their construction is associated with relatively high costs which could not be fully covered by the project budget. The Ministry of Fisheries and Marine Resources in Lüderitz, however, have numerous puerulus collectors which had been constructed and used during a previous study on rock lobster settlement in Lüderitz Bay, and these were made available to the project and installed together with newly-constructed collectors, at the three experimental sites.

A further assumption made in the project proposal was that assistance would be sought from MFMR for the supply of diving equipment and a portable compressor. As this equipment is, however, in frequent use by MFMR scientists and therefore critical to the success of numerous MFMR projects, alternative arrangements had to be made by the project.

During both the July and October 2006 surveys, Ms Kolette Grobler of the rock lobster section of Ministry of Fisheries and Marine Resources in Lüderitz, and/or her technicians visited the study sites and assisted with collector cleaning and re-installation. Ms Grobler has expressed an interest in the study and its results, and has recommended the study be continued for at least another year, to provide a two-year data set. She has indicated that a letter of motivation in support of the extension of the project will be drafted by the MFMR and submitted to the BCLME offices in Luanda and Windhoek.

The study sites are located within Namdeb Diamond Corporation's Elizabeth Bay Mine. Namdeb, and the managers and staff of the Northern Areas in particular, have been extremely forthcoming with their support for the project in providing access to the mining licence area, accommodation on the mine for the project team (often at short notice), access to their mechanical workshop and use of their portable compressor to fill aqualungs. The industry must be complimented for their support of this BCLME project.

APPENDIX III FINANCIAL REPORT

An itemised breakdown of income and costs incurred to date are provided in Tables III.1 to III.4 below.

Table III.1. Itemised breakdown of income as at 10 February 2007 on the project BCLME BEHP/CEA/03/04.

Date	Memo	US$ Amount	Rand Amount	Rate
2004-09-27	Submission of Inception Report	$21,000.00	R 131,934.60	R 6.28260
2005-06-08	Submission of First Progress Report	$17,000.00	R 112,616.50	R 6.62450
2005-11-03	Submission of Second Progress Report	$12,117.00	R 79,900.71	R 6.59410
2005-07-19	Submission of Third Progress Report	$6,000.00	R42,225.00	R 7.03750
Total		**$56,117.00**	**R 366,676.81**	**R 6.63468**

Table III.2. Itemised breakdown of costs (in US$) as at 10 February 2007 on the project BCLME BEHP/CEA/03/04. Average Forex rate: 1US$ = 6.63468.

Budget Item	Sub-Item	US$ Amount
Personnel	Project Management	$15,426.53
	Technical Supervision	$5,846.48
	Diver Hire	$9,921.69
	Student Assistant	$271.30
	Report Writing	$0.00
	Professional Review	$0.00
Total Personnel Costs		$31,466.00
Capacity Building	MSC Student Scholarship	
	Scientific Dive Course	
	Diving Equipment Purchase	
Total Capacity Building		$0.00
Equipment Purchase & Hire	Equipment Purchase	$2,977.54
	Equipment Hire	$1,567.39
Total Equipment Costs		$4,544.93
Survey Costs	Survey Travel Costs	$9,528.35
	Survey Subsistence Costs	$1,676.97
Total Survey Costs		$11,205.32
Project Administration		$1,693.21
Total Expenditure		$48,909.45
Net Income		$7,207.55

Table III.3. Detailed and itemised breakdown of costs (in ZAR) as at 10 February 2007 on the project BCLME BEHP/CEA/03/04.

Budget Item	Date	Expenditure Item	Rand Amount
Personnel			
Project Management			
	2005-05-05	Pisces Project management & reporting	R 7,500.00
	2005-11-01	Pisces Project management & reporting	R 7,500.00
	2006-06-01	Pisces Project management & reporting	R 7,500.00
	2005-05-05	Pisces Project co-ordinator	R 22,000.00
	2005-11-01	Pisces Project co-ordinator	R 18,000.00
	2006-01-30	Pisces Project co-ordinator	R 6,100.00
	2006-02-29	Pisces Project co-ordinator	R 4,250.00
	2006-04-30	Pisces Project co-ordinator	R 6,750.00
	2006-05-31	Pisces Project co-ordinator	R 4,900.00
	2006-07-11	Pisces Project co-ordinator	R 3,600.00
	2006-10-30	Pisces Project co-ordinator	R 6,750.00
	2006-10-30	Pisces Project management & reporting	R 7,500.00
Total Project Management			R 102,350.00
Technical Supervision			
	2005-07-07	Diving supervisor	R 11,000.00
	2005-11-09	Diving supervisor	R 7,500.00
	2006-02-15	Diving supervisor	R 5,000.00
	2006-06-10	Diving supervisor	R 4,000.00
	2006-11-07	Diving supervisor	R 5,789.47
	2006-10-30	Diving supervisor	R 5,500.00
Total Technical Supervision			R 38,789.47
Diver Hire			
	2005-05-05	Diver hire	R 3,800.00
	2005-07-11	Diver hire	R 9,000.00
	2005-07-22	Diver hire	R 7,894.71
	2005-12-05	Diver hire	R 6,800.00
	2005-12-05	Diver hire	R 6,800.00
	2006-04-12	Diver hire	R 2,850.00
	2006-04-25	Diver hire	R 4,550.00
	2006-05-10	Diver hire	R 6,096.49
	2006-06-15	Diver hire	R 6,885.96
	2006-07-07	Diver hire	R 3,200.00
	2006-08-10	Diver hire	R 3,000.00
	2006-10-30	Diver hire	R 4,950.00
Total Diver Hire			R 65,827.16
Student Assistance			
	2006-11-16	Literature search	R 1,800.00
Total Student Assistance			R 1,800.00

Budget Item	Date	Expenditure Item	Rand Amount
Report Writing			R 0.00
Professional Review			R 0.00
Total Personnel Costs			R 208,766.63
Equipment Purchase and Hire			
Equipment Hire			
	2005-07-22	Compressor hire	R 4,719.30
	2005-12-05	Compressor hire	R 1,750.00
	2006-05-10	Compressor hire	R 2,035.09
	2006-06-15	Compressor hire	R 973.68
	2006-11-07	Compressor hire	R 921.05
Total Equipment Hire			R 10,399.12
Equipment Purchase and Expenses			
	2004-04-01	Puerulus collector bolts	25.11
	2004-04-02	Puerulus collector wood	1,476.00
	2005-04-20	Rope	R 196.44
	2005-04-30	Compressor fuel	R 11.50
	2005-04-30	Compressor fuel	R 55.00
	2004-05-24	Puerulus collector welding	9,814.86
	2005-07-07	Drill bits and Stainless steel anchor bolts	R 1,826.48
	2005-07-07	Compressor fuel	R 182.35
	2005-07-11	Puerulus collector manufacture	R 2,668.44
	2005-09-20	Puerulus collector netting	R 207.00
	2005-09-21	Puerulus collector bolts	R 80.62
	2005-09-27	Compressor fuel	R 61.00
	2005-09-27	Survey gas	R 228.07
	2005-11-09	Survey equipment	R 327.01
	2006-01-06	BCLME Survey bolts	R 362.85
	2006-02-08	Underwater Slate Ozatex	R 75.00
	2006-02-15	Collector threaded rod	R 263.45
	2006-03-23	U/W slate surgical tubing	R 139.65
	2006-09-04	BCLME 04 collector frames	R 438.60
	2006-09-07	BCLME 04 collector frames	R 957.43
	2006-10-11	Compressor fuel	R 169.06
	2006-10-11	Rope	R 40.37
	2006-10-11	25 ltr containers	R 92.20
	2006-10-13	Rope	R 56.51
Total Survey Equipment			R 19,755.00
Total Equipment Costs			R 30,154.12

Budget Item	Date	Expenditure Item	Rand Amount
Survey Expenses			
Survey Accommodation			
	2005-09-26	Accommodation	R 614.04
	2006-01-02	Accommodation	R 263.16
	2006-02-19	Accommodation	R 458.50
	2006-05-23	Accommodation	R 330.00
	2006-07-18	Accommodation	R 280.70
	2006-10-10	Accommodation	R 380.00
Total Survey Accommodation			R 2,326.40
Survey Meals			
	2005-04-12	Survey Food	R 23.89
	2005-04-15	Survey Food	R 457.25
	2005-04-18	Survey Food	R 497.00
	2005-04-18	Survey Drinks	R 117.60
	2005-04-20	Survey Food	R 172.50
	2005-04-20	Survey Food	R 23.70
	2005-04-22	Survey Food	R 292.20
	2005-05-01	Survey Drinks	R 29.60
	2005-07-07	Survey Meals	R 258.09
	2005-09-22	Survey Food	R 549.87
	2005-09-24	Survey Food	R 61.90
	2005-09-24	Survey Food	R 5.85
	2005-09-25	Survey Meals	R 192.98
	2005-09-26	Survey Food	R 19.25
	2005-09-27	Survey Food	R 762.11
	2005-11-09	Survey Food	R 70.25
	2006-01-02	Survey Meals	R 110.09
	2006-01-02	Survey Drinks	R 14.39
	2006-01-03	Survey Food	R 69.85
	2006-01-03	Survey Food	R 930.32
	2006-01-06	Survey Drinks	R 28.70
	2006-01-06	Survey Food	R 315.90
	2006-01-07	Survey Meals	R 208.00
	2006-02-19	Survey Food	R 27.40
	2006-02-19	Survey Food	R 21.89
	2006-02-20	Survey Food	R 15.61
	2006-02-20	Survey Drinks	R 18.20
	2006-02-20	Survey Food	R 854.98
	2006-04-03	Survey Food	R 264.82
	2006-04-04	Survey Food	R 38.60
	2006-04-04	Survey Food	R 496.38
	2006-04-04	Survey Food	R 5.07
	2006-04-04	Survey Food	R 135.23
	2006-04-08	Survey Food	R 583.00
	2006-04-10	Survey Food	R 37.74
	2006-04-10	Survey Food	R 22.00

Budget Item	Date	Expenditure Item	Rand Amount
	2006-05-23	Survey Food	R 497.46
	2006-05-23	Survey Meals	R 64.65
	2006-05-24	Survey Food	R 69.80
	2006-07-15	Survey Food	R 459.65
	2006-07-18	Survey Meals	R 136.84
	2006-07-21	Survey Food	R 151.32
	2006-10-10	Survey Meals	R 124.56
	2006-10-11	Survey Food	R 102.42
	2006-10-11	Survey Food	R 1,016.30
	2006-10-11	Survey Drinks	R 42.00
	2006-10-13	Survey Food	R 608.95
	2006-10-16	Survey Meals	R 119.96
Total Survey Meals			R 11,126.12
Fuel Expense			
	2005-04-20	Bakkie Fuel: Luderitz	R 81.00
	2005-04-26	Bakkie Fuel: Luderitz	R 150.02
	2005-04-30	Bakkie Fuel: Luderitz	R 169.53
	2005-05-01	Bakkie Fuel	R 120.30
	2005-05-01	Bakkie Fuel: Keetmanshoop	R 159.71
	2005-05-01	Bakkie Fuel: Van Rijnsdorp	R 200.00
	2005-07-07	Bakkie Fuel	R 740.06
	2005-09-25	Bakkie Fuel: Springbok	R 289.38
	2005-09-25	Bakkie Fuel: Bitterfontein	R 273.01
	2005-09-26	Bakkie Fuel: Keetmanshoop	R 149.24
	2005-09-27	Bakkie Fuel: Luderitz	R 257.72
	2005-10-03	Bakkie Fuel: Aus	R 283.05
	2005-10-03	Bakkie Fuel: Van Rhynsdorp	R 402.14
	2005-10-03	Bakkie Fuel: Grunau	R 224.19
	2005-10-07	Bakkie Fuel: Tokai	R 249.03
	2006-01-02	Bakkie Fuel - Clanwilliam	R 110.00
	2006-01-02	Bakkie Fuel: Springbok	R 203.05
	2006-01-03	Bakkie Fuel: Aus	R 185.03
	2006-01-03	Bakkie Fuel: Grunau	R 125.45
	2006-01-07	Bakkie Fuel: Luderitz	R 106.04
	2006-01-09	Bakkie Fuel: Van Rhynsdorp	R 206.00
	2006-01-09	Bakkie Fuel: Noordoewer	R 174.29
	2006-01-09	Bakkie Fuel: Keetmanshoop	R 206.33
	2006-04-03	Bakkie Fuel: Luderitz	R 312.01
	2006-04-04	Bakkie Fuel: Cape Town	R 86.67
	2006-04-04	Bakkie Fuel: Grunau	R 165.20
	2006-04-04	Bakkie Fuel: Springbok	R 207.05
	2006-04-04	Bakkie Fuel: Keetmanshoop	R 164.00
	2006-04-10	Bakkie Fuel: Luderitz	R 282.48
	2006-04-10	Bakkie Fuel	R 195.00
	2006-04-10	Bakkie Fuel: Springbok	R 83.14
	2006-04-10	Bakkie Fuel: Keetmanshoop	R 181.40
	2006-04-10	Bakkie Fuel: Noordoewer	R 151.32
	2006-04-11	Bakkie Fuel: Tokai	R 420.00

Budget Item	Date	Expenditure Item	Rand Amount
	2006-04-11	Bakkie Fuel: Springbok	R 441.00
	2006-04-11	Bakkie Fuel: Klawer	R 147.50
	2006-05-23	Bakkie Fuel: Springbok	R 438.80
	2006-05-24	Bakkie Fuel: Keetmanshoop	R 300.62
	2006-05-24	Bakkie Fuel: Luderitz	R 224.40
	2006-05-27	Bakkie Fuel: Springbok	R 317.00
	2006-05-27	Bakkie Fuel: Keetmanshoop	R 361.57
	2006-05-27	Bakkie Fuel Tokai	R 383.03
	2006-06-12	Bakkie Fuel: Luderitz	R 189.02
	2006-07-15	Bakkie Fuel: Tokai	R 216.76
	2006-07-16	Bakkie Fuel: Namibia	R 266.00
	2006-07-16	Bakkie Fuel: Keetmanshoop	R 421.00
	2006-07-16	Bakkie Fuel: Garies	R 345.70
	2006-07-18	Bakkie Fuel: Springbok	R 379.05
	2006-07-18	Bakkie Fuel: Keetmanshoop	R 369.91
Total Fuel Expense			R 12,114.20
Vehicle Hire			
	2005-05-05	Vehicle hire	R 4,859.38
	2005-07-07	Vehicle hire	R 1,260.00
	2005-11-09	Vehicle hire	R 1,680.00
	2005-11-09	Vehicle hire	R 7,778.70
	2006-01-15	Vehicle hire	R 4,683.81
	2006-02-15	Vehicle hire	R 1,162.00
	2006-04-16	Vehicle hire	R 3,163.23
	2006-04-12	Vehicle hire	R 7,560.00
	2006-06-12	Vehicle hire	R 6,469.78
	2006-07-07	Vehicle hire	R 246.00
	2006-10-30	Vehicle hire	R 9,114.00
			R 47,976.90
Total Survey Costs			R 73,543.62
Project Administration			
	07/15/2004	Police clearance courier	R 175.44
	07/15/2004	Inception Report copies	R 112.28
	07/16/2004	Inception report courier	R 236.84
	07/23/2004	Invoice courier	R 236.84
	10/07/2004	Work visa courier	R 276.32
	10/15/2004	Fee Teletransmission Inwards	R 285.09
	12/15/2004	Work Visa application copying	R 18.86
	04/04/2005	Work visas	R 3,145.00
	05/27/2005	1st Progress Report Courier	R 517.54
	05/31/2005	Mine Clearance courier	R 187.72
	05/31/2005	Progress Report Copies	R 17.98
	06/22/2005	Fee Teletransmission In: BCLME 04	R 400.00
	08/15/2005	Police clearance courier	R 175.44
	08/23/2005	Mine clearance courier	R 315.79
	10/24/2005	Work visa applications courier	R 187.72

Budget Item	Date	Expenditure Item	Rand Amount
	10/26/2005	Namibia cross border charge	R 120.00
	11/03/2005	2nd Progress Report courier	R 741.14
	11/16/2005	Fee - Teletransmission inward	R 319.60
	12/17/2005	Mine clearance courier	R 214.00
	12/17/2005	Mine clearance copies	R 14.91
	12/19/2005	Work Visas	R 2,000.00
	12/19/2005	Fee: Teletransmission Outward	R 145.00
	01/09/2006	Namibia cross border charge	R 120.00
	02/14/2006	Mine clearance copies	R 18.42
	04/04/2006	Namibia cross border charge	R 140.00
	05/23/2006	Namibia cross border charge	R 140.00
	06/12/2006	3rd Progress Report courier	R 742.00
	07/16/2006	Namibia cross border charge	R 140.00
	10/10/2006	Work visa	R 750.00
	10/10/2006	Namibia cross border charge	R 140.00
Total Administration Costs			R 12,033.93
TOTAL EXPENSE			**R 324,498.30**
NET INCOME			**R 42,178.51**

Table III.4. Breakdown of costs (in US$) as at 10 February 2007 on the project BCLME BEHP/CEA/03/04, illustrating the budget items on which overspending has occurred. Average Forex rate: 1US$ = 6.63468.

Budget Item	Sub-Item	Approved Budget	Expenditure to Date	Remaining Balance
Income		$62,117.00		$6,000.00
Personnel				
	Project Management	$19,000.00	$15,426.53	$3,573.47
	Technical Supervision	$9,000.00	$5,846.48	$3,153.52
	Diver Hire	$13,000.00	$10,192.99	$2,807.01
	Report Writing	$2,500.00	$0.00	$2,500.00
	Professional Review	$1,400.00	$0.00	$1,400.00
TOTAL PERSONNEL		$44,900.00	$31,466.00	$13,434.00
Capacity Building				
	MSc Scholarship	$0.00	$0.00	$0.00
TOTAL CAPACITY BUILDING		$0.00	$0.00	$0.00
Equipment Purchase and Hire				
	Equipment Hire	$1,560.00	$2,977.54	-$1,417.54
	Equipment Purchase	$2,790.00	$1,567.39	$1,222.61
TOTAL EQUIPMENT COSTS		$4,350.00	$4,544.93	-$194.93

Survey Costs			
Subsistence Costs	$1,500.00	$1,676.97	-$176.97
Vehicle Expenses	$11,500.00	$9,528.35	$1,971.65
TOTAL SURVEY COSTS	$13,000.00	$11,205.32	$1,794.68
TOTAL ADMINISTRATION COSTS	$5,647.00	$1,693.21	$3,953.79
TOTAL EXPENSE	$62,117.00	$48,909.45	$13,207.54

Table III.5 provides a breakdown of expenditure to date for the various budget items compared to the budget amended in June 2005. As the costs for maintenance surveys have proven to be considerably higher than anticipated, the budget allocated to subsistence and equipment hire have again been exceeded. As the project was extended to October 2006 in order to obtain a more sound data base, funds from budget items, which had not previously been utilized can be redistributed to cover these expenses. It is also recommended that funds be redistributed to allow for additional time for the write-up of the final report, and professional review thereof. The redistribution of funds will not, however, influence the overall contract amount, or the milestone payments as stipulated in section 3.4 in the UNOPS contract for this project.