The effects of oxygen deficiency, hydrogen sulphide and turbidity on the rock lobster, *Jasus lalandii*: implications for diamond dredge-mining in Namibia.

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DECLARATION

I hereby declare that the work presented here is my own work and that, to my knowledge, no document similar in nature has been submitted for acknowledgement to the University of Cape Town or any other University prior to this date. Much of the information presented in this thesis represents the original work of the author. All external sources of information have been fully referenced and assistance received duly acknowledged.

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ABSTRACT

A study was initiated by the Environmental Section of the De Beers Marine Namibia group as part of a pre-feasibility study for a new dredger (a Trailing Suction Hopper Dredger) to be employed for offshore diamond mining along the Namibian south coast. The aims of this study were to assess (a) the potential mining-induced impacts on oxygen and turbidity, and (b) the behavioural responses of the rock lobster *Jasus lalandii*, to decreased oxygen, elevated hydrogen sulphide and turbidity. For the latter objective, laboratory-based experiments were designed in which juvenile lobsters were exposed to various levels of hypoxia and increased hydrogen sulphide and suspended sediment levels to determine the impacts these stressors may have on the behavioural responses of lobsters.

A literature review was conducted to determine the natural and mining-related variability of dissolved oxygen, temperature and turbidity within Chameis Bay, an area that is reserved for mining as well as being the focus of the pre-feasibility study for the suction-hopper dredger. Two information sources formed the basis of the review: the 2005-2006 Fisheries Independent Monitoring Surveys (FIMS) conducted by Pisces Environmental Services, and a turbidity and plume modeling study done by the Council for Industrial and Scientific Research (CSIR) in 2005. Natural temperature and dissolved oxygen profiles decreased with increases in depth and ranged between mean values of 12.7 and 11.4°C for the surface and bottom temperatures, respectively, and means of 3.63 and 3.20 mg/ℓ for the surface and bottom dissolved oxygen values. No overridingly significant correlations were found to exist between turbidity and temperature or turbidity and dissolved oxygen under the influence of dredge-mining. Dredge-mining had the most significant influence on turbidity. On average dredging increased turbidity values by as much as 2.5-times natural turbidity values.

Laboratory-based experiments on the effects of oxygen depletion and hydrogen sulphide exposure were conducted on a short-term (hourly) basis. These experiments followed a design of step-wise increments of stressful conditions (i.e. decreases in dissolved oxygen for the hypoxia experiments and increases in hydrogen sulphide for the hydrogen sulphide experiments), during which time the behavioural responses of the lobsters were observed and recorded. After each incremental step the stressful conditions were maintained for 10 minutes before the next incremental step was induced. At each stress level, responses were solicited from all experimental lobsters by means of two types of fright stimuli. The lobsters were more tolerant to hypoxia than to hydrogen sulphide exposure. Hypoxia-exposed lobsters did not resort to the extremely energetic evasive tactics displayed by lobsters exposed to hydrogen sulphide and were capable of strong flight responses at very low concentrations (<1 mℓ/ℓ) of dissolved oxygen.

Behavioural responses were recorded during the laboratory experiments and degree of tolerance quantified from these responses. Both male and female lobsters of 50-60 mm carapace length were used in the experiments. No obvious inter-gender differences in tolerance were evident from the results of the gas experiments. However, different types of responses were recorded as hydrogen sulphide levels rose namely: 'no movement' (0.7-1.2 mg H_2S/ℓ, when lobsters initially perceived the stressful conditions), 'prolonged movement' (1.3-2.5 mg H_2S/ℓ, which represented the levels at which lobsters actively tried to evade the stress induced), 'tail-flip' (3.9-5.6 mg H_2S/ℓ, which indicated critical levels of stress) and 'moribundity' (14.8-33.4 mg H_2S/ℓ, when the lobsters were severely incapacitated to the point that they were non-responsive and unlikely to have survived in the field). The sulphide-exposed lobsters exhibited complete recovery within 14 hours of exposure to severely toxic concentrations of hydrogen sulphide.

Laboratory-based turbidity experiments served to determine if sediment suspension, as well as related attributes such as temperature, dissolved oxygen and pH, have any adverse effects on the behaviour and feeding responses of rock lobsters. Two sediment types (collected from within Chameis Bay) were experimented with. The one sediment type comprised mainly silt and mud particles (with a mean particle size of 1.95 Phi [255 µm]) while the second type constituted a more course sediment grade (with a mean particle size of 2.5 Phi [177 µm]). The first sediment type had a high suspension time (with fines being kept in suspension for up to six hours) while the second had a settling time of under 30 minutes. No differences in temperature, dissolved oxygen or pH were detected when turbidity was experimentally increased. Lobsters showed no adverse effects from the exposure to suspended sediment over either short-term or medium-term, the latter covering a 5-day time period during which sediments were continually kept in suspension.

CHAPTER 1: INTRODUCTION, GENERAL BACKGROUND AND OBJECTIVES

Introduction

The on-shore and off-shore diamond resource on the west coast of southern Africa is considered to be the largest secondary diamond deposit in the world (Corbett 1996, Moore & Moore 2004). Estimated at a total resource potential of between 1.5 and 2 billion carats (Levison *et al.* 1992, Moore & Moore 2004), to date only about 10% of this resource has been recovered (Levison *et al.* 1992, Spaggiari *et al.* 2006). The diamond mining industry in southern Namibia has been in existence since 1908 when the first diamonds were discovered at Lüderitz, and has played a significant role in the socio-economic development of Namibia where it contributes 10% to the country's Gross Domestic Product and 30% of its total exports (Spaggiari *et al.* 2006).

However, another resource that is important to both Namibia and South Africa is the rock lobster *Jasus lalandii*, which inhabits the nearshore sub-tidal to mid-water zones along the southern African west coast. These rock lobster populations support a 100-year old, multi-million dollar fishing industry that is crucial to the fishing sectors of both countries and forms a significant part of a globally relished food source (Newmann & Pollock 1971, Brandão *et al.* 2004). However, because of the inherent geographical overlap of the mining and fishing industries, concerns have been expressed about their co-existence – since the nature of the marine diamond mining industry is considered to be in direct conflict with that of the rock-lobster fishing industry. As a result, numerous studies have been undertaken by both sectors to examine the extent of the conflict and explore ways of achieving harmony. Such is the nature of this study.

Initiated by De Beers Marine Namibia (DBMN) this study was proposed to examine the potential impacts diamond dredge-mining along the southern Namibian coast would have on rock lobster populations at depths of 30-70 m. This study – together with others such as the Chameis Bay Dredge Trial study (Pulfrich *et al.* 2004, CSIR 2005), the 2004-2005 Specialist Studies (Cownie 2004, Probyn 2004, Pulfrich 2004, Steffani & Pulfrich 2004, Pulfrich 2005), the Aerial Kelp Bed Survey (Pulfrich & Penney 2006) and the Fisheries Independent Monitoring Surveys (FIMS) (Pulfrich *et al.* 2006) – formed part of the Environmental Impact Assessment (EIA) for the Pre-feasibility phase of the Marine Dredging Project (MDP). The MDP, a project coordinated by DBMN in collaboration with the Namdeb Diamond Corporation (Pty) Ltd., was aimed at assessing the feasibility of employing Trailing Suction Hopper Dredgers to mine areas of Namdeb's Atlantic 1 Mining License Area (Pisces 2004, Pulfrich *et al.* 2004a). The MDP was designed to cover the whole process from the pre-feasibility phase (initiated in 2004) to the final commissioning of full-scale dredging operations in Atlantic 1 by 2008 – with the pre-feasibility and feasibility phases each having separate Scoping, EIA and Environmental Management Plan (EMP) components (Pulfrich *et al.* 2004a).

The EIA for the pre-feasibility phase highlighted that the re-suspension of sediment, particularly sediment <63 μm (i.e. fines) by mining constitutes one of the major concerns of environmentalists. Various aspects of mining contribute to this re-suspension, including the mechanical disturbance of sediment by the dredging process, sediment release back into the sea during the dredging process (overspill), during stock-piling of sediment in land-based paddocks, and the eventual discharge of sediment from the land-based processing plants. This concern prompted the need for understanding the physico-chemical conditions associated with the suspension, displacement and deposition of sediments (addressed by the Chameis Bay Dredge Trial studies – CSIR 2004, 2005a & 2005b). An additional issue is the impacts these attributes may have on benthic habitats and associated biota in the proposed mining region (Pulfrich *et al.* 2004a).

The selection of *Jasus lalandii* as an indicator species to monitor the impacts on benthic fauna gave rise to a number of studies – e.g. the 2005-2006 FIMS, the 2006 Kelp survey and this study. The FIMS was proposed as a response to the paucity of baseline information regarding the population dynamics, recruitment areas and migrational patterns of *J. lalandii* in Namibian waters (Pulfrich *et al.* 2006).

Besides being a food source for many species, kelp beds also provide shelter for juvenile and adult rock lobsters (Pulfrich & Penney 2006), and also play an important role during the recruitment phase of pueruli (Carr 1994, Pulfrich & Penney 2006). Mayfield & Branch (2000) and Day & Branch (2002) found that sea urchins feed on pieces of kelp fronds that break off and fall to the seabed. Sea urchins, in turn, provide shelter for juvenile abalone and both these species are preyed upon by *J. lalandii* (Mayfield & Branch 2000, Mayfield *et al.* 2001, Van Zyl *et al.* 2003). It is for these reasons that a kelp survey was conducted to obtain a general understanding of the abundance and distribution of kelp beds in the mining area, and to determine from the literature how kelp is impacted by sedimentation (Pulfrich & Penney 2006). To add to the series of studies, the current study was proposed to gauge the responses of lobsters to stressful environmental conditions considered to be directly and indirectly influenced by the dredge-mining process including suspension of sediments and gaseous perturbations.

Literature Review and Background Information

Diamond mining

Origins of Marine Diamonds

All southern African diamonds, land-based and marine, have their origin roughly 150 km within the earth's crust where they were created under extremely high temperatures and immense pressure (Moore & Moore 2004, Spaggiari *et al.* 2006). During the early Cretaceous period (approx. 120 million years ago) silica-poor magma under pressure escaped from the earth's core through weaknesses in the surrounding crust layer. As the magma flowed up towards the earth's surface it cooled down and solidified into lamprolite rock (commonly referred to as kimberlite). As a result, materials being transported by the magma (specifically temperature-resilient material like diamonds) were trapped in the solidifying rock (Moore & Moore 2004). The lamprolite rock formations generally resemble conical structures when solidified, and are thus generally referred to as kimberlite pipes (Moore & Moore 2004, Spaggiari *et al.* 2006).

Of the different types of Cretaceous kimberlite rock present in the earth's crust, diamondiferous kimberlite is only found in Archean-aged Cratons – i.e. thick layers of continental crust that are estimated to have been formed more than 2.5 billion years ago (Rogers & Li 2002, Moore & Moore 2004, Spaggiari *et al.* 2006). In southern Africa diamondiferous kimberlite pipes are found on the Kaapvaal and Zimbabwe cratons as well as on the Limpopo mobile belt (Moore & Moore 2004). It is believed that for the past 90 million years (since the Late Cretaceous) the Orange-Vaal River drainage system has been fairly stable and thus closely resembles the present-day catchment system depicted in Figure 1.1 (Moore & Moore 2004). As a result an enormous amount of rain and wind-weathered rock material from the eastern and central regions of southern Africa has been transported to the west coast (Moore & Moore 2004). It is estimated that the upper 1.4 km of the diamondiferous kimberlite pipes in the Orange-Vaal catchment have been eroded and transported westward. Thus the kimberlite pipes near Kimberly, South Africa, are considered to be the main source of the estimated 2 billion carat resource found along the west coast (Moore & Moore 2004).

Rogers & Li (2002) and Spaggiari *et al.* (2006) described the process whereby submerged marine diamond fields came to be. Elaborate as the theory is in its entirety, it is based on the simple concept of a rising and falling sea level by a range of 700 m over the last 80 million years and is accepted as the only plausible explanation for the occurrence of diamondiferous terraces as deep as 200 meters below sea level (Matthews 2000). There are, however, two theories regarding the long-shore placement of marine diamonds (Moore & Moore 2004). The reason for this is that there are two distinct areas of diamond placement off the west coast: the first being the well-known Orange River mouth area and the second the Olifants River mouth placement area (Moore & Moore 2004). (Fig. 1.1)

It is understood that after the west-coast deposits were created, the prevailing northward flowing Benguela Current displaced the diamonds from the two placement areas along the coast. For the Orange River deposits, this lead to the land-based and marine diamond placers found at regions as far north as Hottentot Bay, Namibia (Moore & Moore 2004). In the case of the Olifants River mouth deposit the diamonds are believed to have been spread out along the coastline to as far north as the Orange River mouth, thereby constituting one continuous diamond resource stretching from Cape Columbine, South Africa, all the way to Hottentot Bay, just north of Lüderitz in Namibia (Matthews 2000, Moore & Moore 2004). (Fig. 1.2)

Diamond Mining Industry

Land-based diamond mining at Lüderitz, Namibia, started in 1908 when the first diamond was discovered by a railway worker (Zacharias Lewala) who had previously worked in the diamond mines near Kimberly, South Africa, and thus was familiar with the appearance of rough diamonds (Spaggiari *et al.* 2006). In the same year the Sperrgebiet, a 26 000 km^2 area between Lüderitz and the Orange River, was proclaimed as a restricted area and since then has yielded over 75 million carats of diamonds (Spaggiari *et al.* 2006). According to Spaggiari *et al.* (2006) the Sperrgebiet is the richest placer know to exist on earth, since 95% of the diamonds recovered are of gem quality.

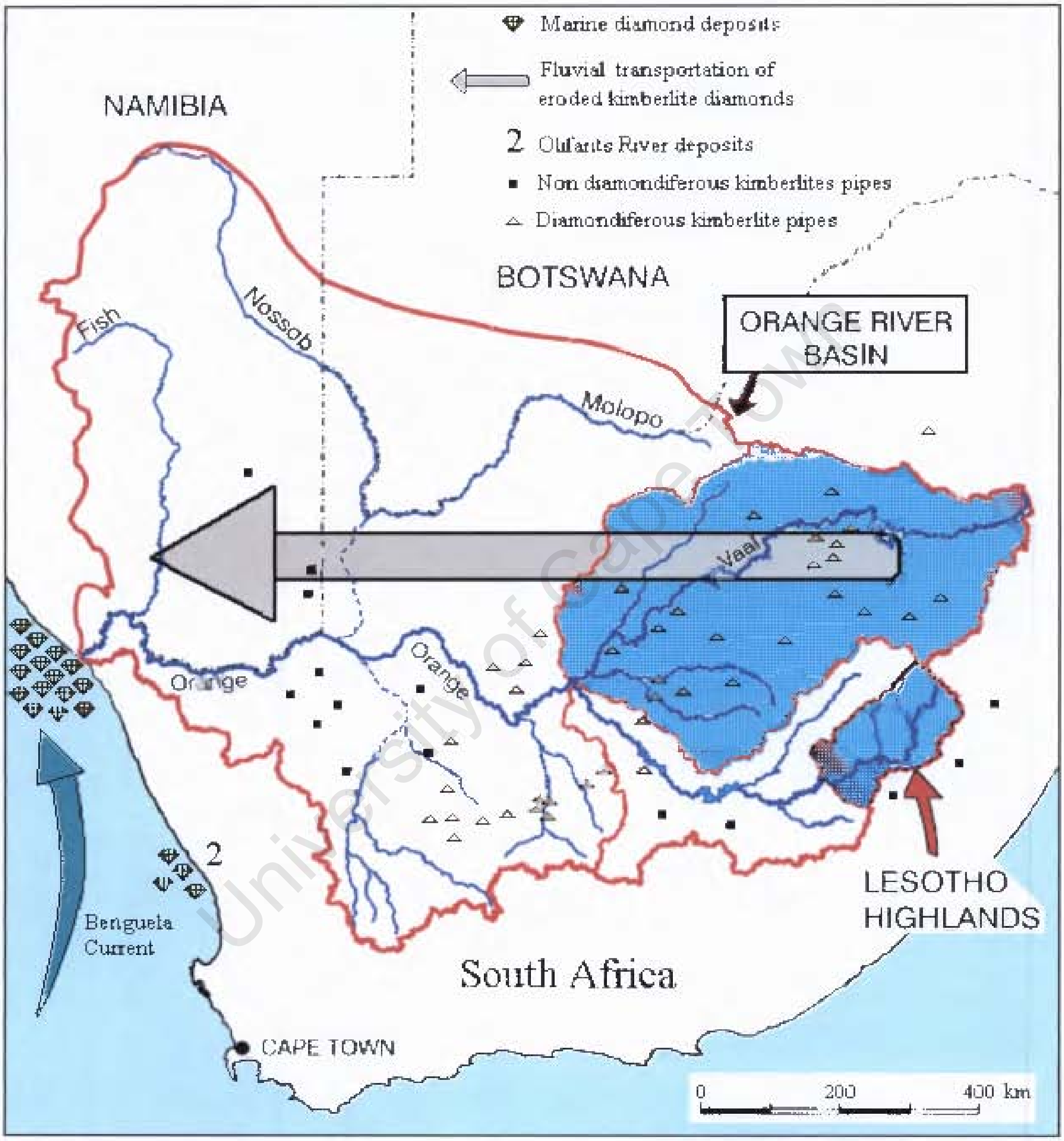


Figure 1.1: Map showing the Orange River and Vaal River drainage basins and the two marine diamond deposit sites along the west coast of southern Africa (modified from Moore & Moore 2004).

However, geological experts believe that the land-based terraces represent a small component of the actual resource and that the bulk of the west-coast diamond resource is located in the adjacent marine environment to depths potentially 300 m below the current sea level (Matthews 2000). As a consequence, over the last 30-40 years there has been an increasing focus on the marine constituent of the west coast diamond resource (Rogers & Li 2002, Moore & Moore 2004). Marine diamond mining along the Namibian coast started in the 1960s when Sam Collins, an entrepreneur from Texas, USA, utilized barges equipped with centrifugal and airlift pumps to recover diamonds in the shallow marine areas near the coast between the Orange River mouth and Hottentot Bay (Matthews 2000). In 1961 the then adjacent, land-based, diamond mining conglomerate, De Beers, bought Collins' company and in so doing started its marine branch of diamond mining along the southern African west coast (Matthews 2000).

At present there are three main types of marine diamond-mining operations taking place along the southern African west coast. The first comprises inter-tidal and shallow subtidal diver-facilitated mining operations in which divers use suction hoses to extract potentially diamond-rich gravel from between rocky patches along the coast. This type of mining is implemented from the land-sea interface to a depth of 30 m, and sometimes involves the cutting of kelp to ease the movement of hoses and airlines, and overturning of boulders to gain access to the underlying sediment (Matthews 2000, Pulfrich *et al.* 2003a). In southern Namibia, coastal diamond mining is based on the principle of driving back the ocean by building extensive seawalls to reclaim submerged areas of the seabed for mining (Matthews 2000, Grundlingh *et al.* 2006). A third form of mining comprises mid- and deep-water operations that utilize remotely controlled mining vehicles (crawlers) and vertical drills to remove sediment and semi-consolidated material from the seabed (Matthews 2000). The gravels are then pumped into mined-out paddocks on land for processing and mining in-line with the conventional dry-mining process (Pulfrich 2004). These operations normally occur at 30-80 m depths for the mid-water mining, and greater than 80 m depths for the deep-water activities. Increasing focus has been placed on this sector over the last three decades as land-based resources are near depletion and thus marine mining is undoubtedly the future of the diamond mining industry along the southern African west coast (Matthews 2000).

De Beers Marine Namibia currently employs a fleet of five mining vessels (MVs) for its mid- and deep-water mining operations along the Namibian south coast. Four of these vessels (the Debmar Atlantic, Debmar Pacific, Grand Banks and !Gariep) are equipped with Wirth drill technologies, while the MV Ya Toivo utilizes a remote-controlled swivel-head crawler to mine the 6 000 km^2 Atlantic 1 Marine License Area on an exclusive mining contract to Namdeb Diamond Corporation (Pty) Ltd. (Pisces 2004, Probyn 2004). In certain areas of the Atlantic 1 area the mining procedures entail mining through thick layers of overburden (i.e. layers of overlying sediment comprising fines <63 µm and sand of 63 µm to 2 mm) to gain access to the underlying gravels (>2 mm) where most of the diamonds are found (CSIR 2005, Pisces 2004, Pulfrich *et al.* 2004a, Steffani & Pulfrich 2004). Although capable of mining to depths as deep as 150 m below mean sea level (MSL), the current DBMN drill and crawler technologies produce uneconomical yields when mining areas with overburden exceeding 5 m thickness (Pisces 2004). Thus, DBMN is considering the possibility of employing Trailing Suction Hopper Dredgers to mine these areas, as well as possibly mining areas in the biogenic mud-belt (found in the 70-120 m depth region of Atlantic 1), where the mud layer covering potentially diamond-rich gravels has an average thickness of 15 m (Rogers 1977, Pulfrich *et al.* 2004a).

Trailing Suction Hopper Dredgers use one or two trailing suction dredges, deployed from the sides of the vessel, to dredge the seabed along the path of the vessel as it slowly steams over the intended mining area. All dredged material is collected in the hopper of the mining vessel, where heavy materials (such as sand and gravel) settle in the hopper while the lighter materials in suspension flow back into the ocean through the hopper's overflow sluices. Once the hopper is full, the mining vessel deviates from its mine-path and steams inshore to connect to the sinkerline through which the contents of the hopper are pumped into mined-out paddocks on land where they are stockpiled. During subsequent dry-treatment procedures, unwanted fines are discharged back into the sea, creating turbid plumes of suspended sediment (Pulfrich 2004). The ecological impacts of this mining procedure encompass the complete destruction of benthic habitats (and associated epi- and infaunal communities) through the removal of the substrate as well as the re-suspension of sediment at the drag-head and at the sea surface (as a result of overspill). The re-suspended sediment adds to the turbidity of the water column and can potentially have secondary negative impacts on adjacent and remote benthic faunal communities as it eventually settles.

These effects must, however, be considered in the context of background suspended sediment concentrations in the area. Storms, outflow from the Orange River and Berg winds all naturally contribute to turbidity. The levels of turbidity associated with these events are substantial, often exceeding those associated with dredge-mining (Zoutendyk 1995, Rogers & Li 2002).

Since dredge-mining increases suspension of sediments in a variety of ways, one of the focuses of my study was to investigate the behavioural responses of juveniles of the rock lobster *Jasus lalandii* to turbidity. Juveniles were selected for two main reasons: (1) to assess the implications of mining-related impacts on recruitment areas (Pulfrich *et al.* 2004), and (2) juveniles are more sensitive to environmental perturbations than adults (G.M. Branch, UCT, pers. comm.).

Potential secondary effects of mining are a reduction of dissolved oxygen and an enhancement of hydrogen sulphide if mud belts laden with this gas are disturbed. Another aspect of my study was thus to examine the behavioural responses of rock lobsters to hypoxia and elevated hydrogen sulphide.

Jasus lalandii

Biology and life cycle of Jasus lalandii

Due to its economic significance, *Jasus lalandii* has been extensively studied. It is endemic to the southern African west coast and is an opportunistic predator and scavenger, feeding on a range of benthic and non-benthic organisms such as mussels, abalone, urchins, whelks, barnacles, mysids and fish (Heydorn 1969, Branch *et al.* 1987, Barkai & Branch 1988a, Pollock *et al.* 1982, Mayfield *et al.* 1999, Mayfield *et al.* 2001, Van Zyl *et al.* 2003). Numerous studies have shown that prey size is dependent on lobster size and that lobsters can be selective, seeking out food that yield the highest return per energy investment (Heydorn 1969, Mayfield & Branch 2000, Mayfield *et al.* 2001, Van Zyl *et al.* 2003).

Jasus lalandii has an elaborate life cycle, with the most radical morphogenesis occurring during the planktonic larval stages. The mating season differs slightly between the northern and southern Benguela populations but coincides with the moulting of sexually mature females (Silberbauer 1971) at the beginning of April (mid Autumn) and continuing until the end of July (Heydorn 1969, Dubber *et al.* 2004). Delayed internal fertilization of the eggs occurs within 15 days of copulation (Silberbauer 1971). Fertilized eggs are released, attached to setae on the female's pleopods (Silberbauer 1971) and carried around for the duration of the three-month embryonic phase in Winter. The planktonic phase is initiated in Spring by the hatching and release of the first naupliosomal larvae into the water column. The larvae undergo a total of 14 phyllosomal stages (Lazarus 1966) before they finally moult into the first benthic stage, which is termed a puerulus and resembles the adult, but is colourless (Heydorn 1969). This stage lasts roughly 25 days after which the juvenile lobster undergoes ecdysis and sheds its first calcareous exoskeleton. Rate of growth of puerulus is influenced by temperature and food type (Dubber *et al.* 2004). The whole process from hatching to the first instar stage takes about nine months (Silberbauer 1971).

Ecdysis takes on average 26-32 days, depending on the temperature (Heydorn 1969, Silberbauer 1971, Hazell *et al.* 2002), and the lobsters are extremely vulnerable during this time. High mortalities occur as a result of predation, blood loss due to external damage and other unknown factors (Heydorn 1969, Lipschitz 1983, Hazell *et al.* 2001, 2002). No food is consumed during this period, as the mandibles of the lobsters are soft (Lipschitz 1983). Mature lobsters moult annually, with males moulting during late Winter to Spring (August to November), while the females moult from April to July (Heydorn 1969, Silberbauer 1971, Hazell *et al.* 2002). Juveniles moult roughly three times per year, and reach sexual maturity at carapace lengths of 60 mm for the males and 68 mm for the females in the southern Benguela (Pollock 1973).

Although lobsters reach sexual at relatively the same age, lobsters in the southern Benguela are much larger than northern Benguela lobsters at the onset of sexual maturity. This difference in size is considered to be a result of reduced growth rates (i.e. limited and sometimes negative moult increments) in the northern Benguela lobster populations that is believed to cover both the juvenile and adult phases (Cockcroft & Goosen 1995, Hazell *et al.* 2002). Size at sexual maturity is, however, not a fixed population attribute as it is dependent on a number of factors (such as temperature, dissolved oxygen levels and the nature and amount of food available during the time preceding the moulting phase), and also varies geographically and inter-annually

(Cockcroft & Goosen 1995, Dubber *et al.* 2004). To give an example, Grobler & Noli-Peard (1997) have shown that at Hottentot Point, an area which has been under continuous fishing pressure since the 1980s, females reach sexual maturity at carapace lengths (CL) of 51 mm, while those from Black Rock, roughly 15 km north of Hottentot Point, reach sexual maturity at 55 mm. This is equivalent to a six- to eight-month time difference, given a standard moult increment of 2.5 mm (Grobler & Ndjaula 2001) and three juvenile moults per year.

Distribution and Migration

Populations of *J. lalandii* are found between Cape Cross on the west coast of Namibia (21° 43' S, 13° 58' E) and Port Elizabeth (33° 50' S, 25° 50' E) on the southeast coast of South Africa (Fig 1.2). They generally inhabit rocky outcrops and have a depth range from just below the low-water mark down to 135 meters (Heydorn 1969).

Seasonal migratory patterns have been observed mainly for adult rock lobsters in both the northern and southern Benguela populations (Pollock *et al.* 1982, Tomalin 1993, Grobler & Noli-Peard 1997, Atkinson & Branch 2003). Inshore and offshore migrations coincide with particular life cycle stages, with the lobsters tending to migrate inshore at the start of the moulting and mating seasons (Pollock *et al.* 1982), but differ in extent depending on the concentration of dissolved oxygen levels (Tomalin 1993, Grobler & Noli-Peard 1997).

Atkinson & Branch (2003) have considered whether *J. lalandii* exhibits contranatent long-shore migration (against the current) as an instinctive measure to counter-displacement of larvae and maintain its geographical distribution – a trait exhibited by other palinurid species such as *J. edwardsii*, *J. verreauxi* and *Palinurus gilchristi* (Groeneveld & Branch 2002). Atkinson & Branch (2003) found no evidence of adult long-shore movements by *J. lalandii*, although they did note that their study was incomplete as the data used were obtained mainly from adult males of carapace lengths greater than 70 mm – excluding adult females and juveniles.

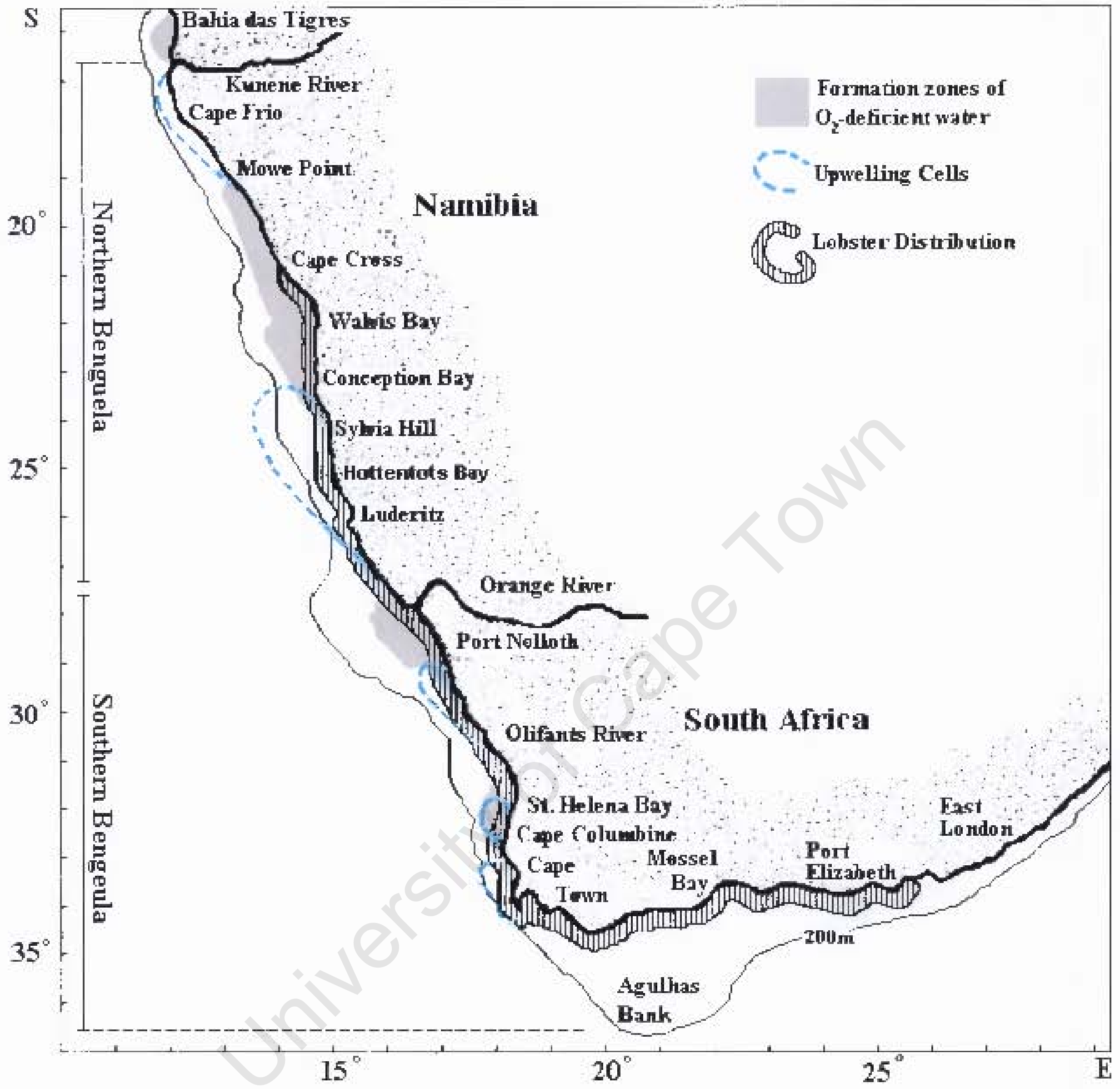


Figure 1.2: Map showing the main upwelling areas and distribution of *Jasus lalandii* along the west coast of southern Africa as well as the two oceanographic sub-divisions (modified from Beyers *et al.* 1994).

Jasus lalandii Fishing Industry

In Namibia, lobsters are found in commercially exploitable densities between Meob Bay (24° 31' S, 14° 35' E) and Kerbe Huk (28° 13' S, 16° 28' E) just north of the Orange River mouth (Bailey *et al.* 1985, Grobler & Noli-Peard 1997). The history of the fishery has been reviewed in numerous studies since its inception in 1922 (Grobler 1994) and is known to have undergone a severe collapse during the 1960s – believed to have been brought on by a combination of stressful environmental conditions and excessive fishing pressure (Mathews & Smith 1978, Beyers & Wilke 1990, Tomalin 1993, Grobler 1994, Grobler & Noli-Peard 1997). However, after the collapse strict control – entailing annually reviewed Total Allowable Catch (TAC) quotas, minimum catch-size (65 mm CL) limitation, a male-only policy for commercial catches, and a closed fishing season (1 May to 31 October) which envelopes the mating and moulting seasons (Tomalin 1993, Grobler & Noli-Peard 1997) – has ensured conservative harvesting of the resource, and signs of stock recovery, although subtle, are evident (Grobler 1994).

As the lobsters are concentrated nearshore because of seasonal oxygen-poor water, the Namibian lobster fishing industry operates at much shallower depths than the South African fishing industry – i.e. from the inter-tidal zone to a depth of 30 m below MSL (Tomalin 1993, Grobler & Noli-Peard 1997).

The Benguela Ecosystem

The Benguela system dynamics

As one of the four major upwelling systems in the world, the Benguela upwelling system is a highly dynamic (Fig. 1.3) and productive ecosystem (Cushing 1971, Jarre-Teichmann *et. al.* 1998, Shillington 1998, Shannon *et al.* 1992, Shannon & O'Toole 1999, Shannon 2000). Due to the importance of its marine resources for Angola, Namibia and South Africa, the system has been extensively studied with the earliest records dating back to the 1830s (Hart & Currie 1960, Cushing 1971, Bailey *et al.* 1985, Jarre-Teichmann *et. al.* 1998). The Benguela system is made up of a main offshore, equator-ward flowing cold current, roughly 200 km wide in the southern regions (Shannon 1985) and approximately 750 km wide in the northern regions (Wedepohl *et al.* 2000). The current is believed to originate in the South Atlantic and forms part of the global oceanic circulatory system known as the Great Oceanic Conveyor (Shillington 1998, Hardman-Mountford *et al.* 2003).

The offshore Ekman transport of coastal water, resulting from the combined effects of the prevailing southeast trade winds, shelf bathymetry and the earth's rotation, induces the upwelling (Fig. 1.3b) of cold water from areas well beyond the 200 m isobath (Hardman-Mountford *et al.* 2003). These upwelling events result in nutrient pulses into the warmer, better-illuminated coastal areas and sustain the high levels of productivity found in these regions (Cushing 1971, Boyer *et al.* 2000, Shannon 2000, Hardman-Mountford *et al.* 2003, Van der Lingen *et al.* 2006). Most of the upwelling occurs along the coast between 16°S and 34°S (Shannon 1985). However, the upwelling regime is not continuous or homogenous along the coast, but is defined by distinct zones (cells) with varying degrees of upwelling (Cushing 1971, Lutjeharms & Meeuwis 1987) – with the largest and most constant cell being near Lüderitz, Namibia (Lutjeharms & Meeuwis 1987, Hardman-Mountford *et al.* 2003) (Fig. 1.2).

Due to a number of inconsistencies in system dynamics within the Benguela system, the system has been subdivided into two oceanographic regions (Fig. 1.2), the northern and southern Benguela (Jarre-Teichmann *et. al.* 1998, Shannon 2000). The exact boundaries of these regions are not well defined, especially in the case of the southern limit of the southern Benguela. However, numerous authors have designated the southern-most limit of the Lüderitz upwelling cell as the boundary between the northern and southern regions because of the evident barrier brought on through the strong geophysical and biological features of this upwelling cell

(Shannon 1985, Hardman-Mountford *et al.* 2003). To the south, upwelling is spatially and temporally more variable, being pulsed and concentrated at the Cape (Fig. 1.2).

Wind anomalies and upwelling

The prevailing southeastern winds along the west coast of southern Africa are controlled by an anticlockwise (anti-cyclonic) South Atlantic high-pressure system, the seasonal low-pressure system that develops over the land, and eastward-moving cyclones, which cross the southern part of the subcontinent (Shannon & O'Toole 1999, Chang 2003). The South Atlantic high-pressure system is part of a discontinuous belt of high pressure that encircles the southern hemisphere and prevails throughout the year. The pressure over the subcontinent alternates between a well-developed low-pressure system during Summer and a weak high-pressure system during Winter, which causes the seasonal variability in the atmospheric pressure and wind fields (Shannon & O'Toole 1999). In the Benguela ecosystem, wind plays a very important role in that it is the primary factor responsible for upwelling (Shannon & O'Toole 1999, Hardman-Mountford *et al.* 2003, Monteiro *et al.* 2006).

In the northern Benguela wind fields are most pronounced during Autumn and Spring (Shannon & O'Toole 1999), while in the southern Benguela increased wind activity is generally observed during late Spring to Summer (Andrews & Hutchings 1980, Roy *et al.* 2001, Hardman-Mountford *et al.* 2003). Strong seasonal patterns in wind direction and intensity are characteristic of winds in the southern Benguela, whereas north of 31°S the wind displays weaker seasonal variation with a permanent along-shore wind being observed throughout the year (Chang 2003). Berg winds (strong easterly winds driven by a large high-pressure system over the interior of the subcontinent) occasionally occur during Spring and Autumn in the northern Benguela and persist on average four days (Chang 2003). During berg wind cycles, upwelling along the coast is greatly suppressed as the hot and dry wind blowing off the plateau flows over the heavier, cool coastal wind layer adjacent to the sea surface. This cool air layer prevents the berg winds from coming in contact with the water surface and exerting the sheer stress required for upwelling to occur (Berger & Wefer 2002, Chang 2003).

Current dynamics

The Benguela ecosystem is controlled by a fairly intricate current regime (Fig. 1.3 and 1.4). The main body of water making up the cold Benguela Current is found roughly 120 km offshore and is an enormous conical, trough-like body of water with a mean current velocity of 2 cm/s and water temperatures ranging from 7.4°C at the sea surface to 2.75°C at a depth of 2000 m (Shannon 1985, Wedepohl *et al.* 2000). The Benguela Current has a net north-western flow of water. However, Lass & Mohrholz (2005) and Boebel *et al.* (2003) found that there is a complex set of currents including a deep pole-ward counter current, coastal-trapped waves moving south, and eddies (Fig. 1.3).

In the southern regions of the Benguela, large gyres of warm water (roughly 120 km in diameter) arising from the Agulhas Current along the southern coast of South Africa occasionally break free from the Agulhas circulation and drift over into the Benguela circulation (Fig. 1.3 A). Once in the Benguela circulation they are forced northwestward over the Walvis Ridge while gradually losing heat to the surrounding cold water until they eventually disintegrate (Boebel *et al.* 2003). These gyres travel at a rate of 3.8 cm/s across the Benguela Current (Boebel *et al.* 2003).

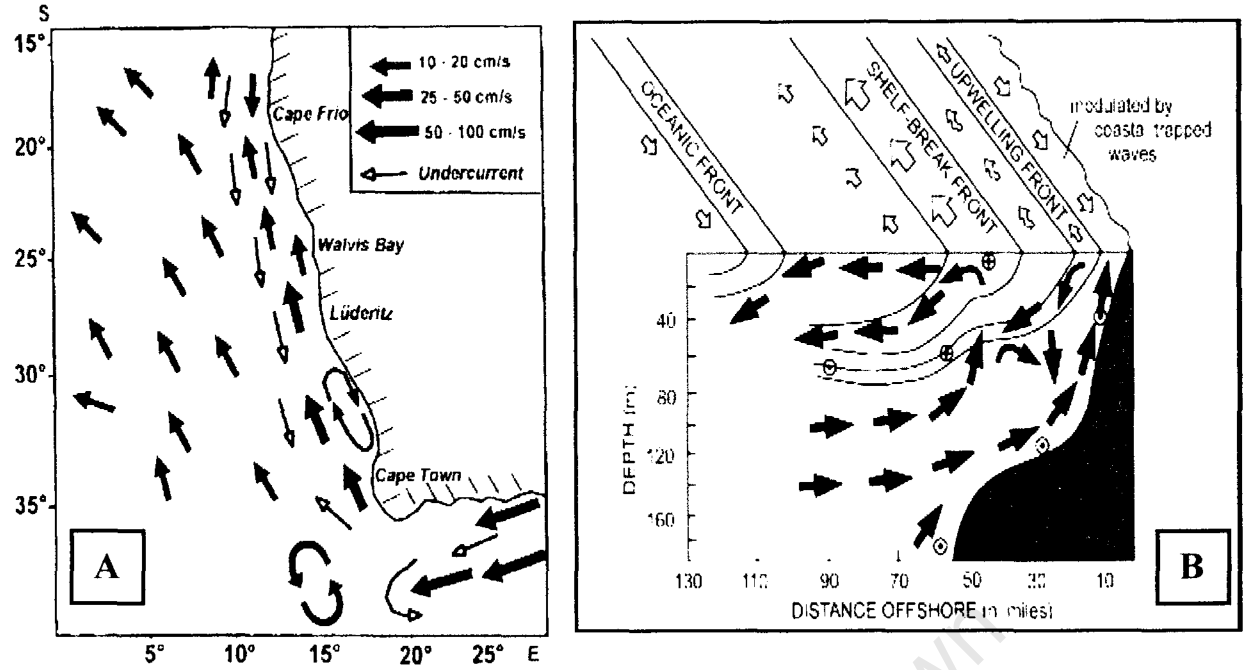


Figure 1.3: Diagrammatic representation of the current regime in the Benguela Cold Current system. (A) shows an aerial depiction of the Benguela current system (Fennel 1999) and (B) a cross sectional profile of the conceptual circulation during an upwelling event (Berger & Wefer 2002). Encircled crosses in (B) represent equator-ward flowing currents and encircled dots the pole-ward counter current (Berger & Wefer 2002).

Over the continental shelf the Benguela Current dynamics are also controlled by very complex current regimes, this being more significant for the southern regions of the Benguela (Fig. 1.4). However, at certain areas along the coast the alongshore flow becomes very dynamic with filaments of water flowing in the opposite direction in relation to the general current flow. *In situ* current measurements have shown that the surface current in the southern region of the Benguela travels at a mean speed of 17 cm/s with a variance of between 11 and 23 cm/s (Shannon 1985, Wedepohl *et al.* 2000). According to Wedepohl *et al.* (2000) the greatest surface velocities occur during Winter in the north and Summer in the south as induced by the seasonal wind fields. Current speed generally decreases with depth because of the increased frictional forces between adjacent water layers as well as between the substratum and the overlying water layer in the case of bottom currents (Fennel 1999).

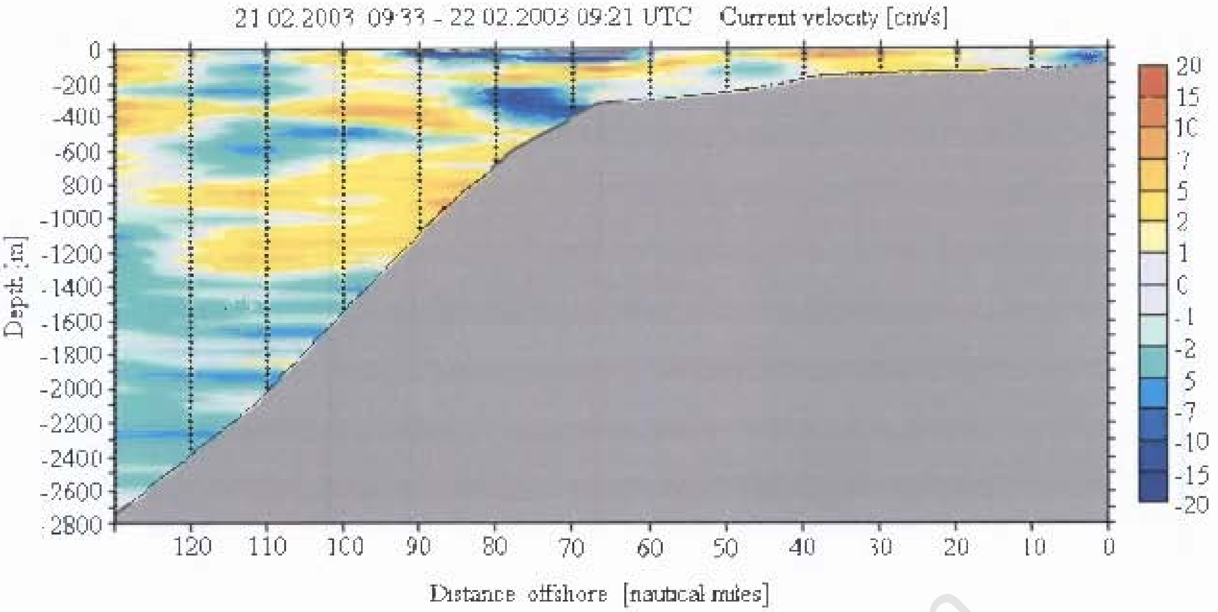


Figure 1.4: The Benguela Current profile at 23°S showing current speed (cm/s) on a colour grid (Lass & Mohrholz 2005). Orientation: North is perpendicularly into the page and West to the left hand side. Negative velocities indicate counter-current velocities (Lass & Mohrholz 2005).

Intrusions of warm water into the Benguela at the Angola-Benguela Front further add to the highly variable nature of the Benguela Current. These intrusions are known to occur annually and have been recorded as far south as Walvis Bay, 23°S (Boyd 1987, John *et al.* 2004). These filaments of water are restricted to the upper reaches of the continental shelf (above the 300 m isobath) and are characterized by elevated salinity (35.8-36.2‰), temperature (20-24°C), and chlorophyll concentration (8-21 mg/m^3) and reduced (<2 mℓ/ℓ) dissolved oxygen profiles (John *et al.* 2004, Monteiro *et al.* 2006, Van der Lingen *et al.* 2006).

Dredging and turbidity attributes

Of the available literature on marine dredging activities the main focus has been on problems associated with environmental impacts caused by the release of pollutants, particularly heavy metals, from harbour-dredged material that is discharged at sea (Morton 1977, Pearson &

Rosenberg 1978, Bonvicini Pagliai *et al.* 1985). With regards to environmental impacts related to commercial aggregate extraction, several papers have documented the initial massive destruction of benthos and the fact that recovery takes between one and seven years depending on the nature of the sediment – with the fauna of fine sediments recovering quickly and that of gravels very slowly (Newell *et al.*1998, Ellis 1996, Kenny & Rees 1999, Boyd *et al.* 2005). From the fisheries perspective most studies focus on the dredge-harvesting of benthic organisms such as scallops, clams and benthic fish species, the destruction of the benthic habitat from trough-dredging and subsequent recovery of these benthic environments (Morton 1977, Conner & Simon 1979, Bonsdorff 1980, Poiner & Kennedy 1984).

Over the past two decades there have been increasing concerns with regard to the sediment discharged from coastal diamond mining plants along the southern coast of Namibia. This has led to the initiation of numerous impact assessment studies in the area. These initiatives, amongst others, include a series of sediment-plume modeling projects contracted to the Council for Scientific and Industrial Research (CSIR) intended to shed light on the sediment-current dispersion characteristics along the coast. In their study of treatment plant discharges north of Chameis Bay, CSIR (2005) showed that currents are capable of displacing sediment, particularly fines <63 μm, over 3 km away from the point of discharge. In this study, only the near-shore areas, shallower than 30 m, were considered – areas that are generally characterized by strong wind-induced current profiles (Lass & Mohrholz 2005, CSIR 2004). Pulfrich *et al.* (2003b) also investigated the effects of the near-shore release of fines tailing from land based diamond mines on intertidal and subtidal benthic communities and found that the laws governing the impacts of sedimentation on the benthos are very complex. These authors found that, depending on the variable being assessed (i.e. specific species, community structure or diversity) the outcomes may be positive or negative, or there may be no effect at all on the benthic community. In the case of rock lobsters, they found that sedimentation had no impacts on the species' density or size-structure in the areas studied (Pulfrich *et al.* 2003b). However, studies done at intermediate water level depth of 30-80 m showed that sediment distribution by currents is more extensive (Fig. 1.5) and occurs on a slightly longer time scale than that in the near-shore areas (Penney & Smith 2004). This is a result of the combined effects of dynamic current profiles (i.e. high variance with regards to speed and directional attributes) and water stratification (Lass & Mohrholz 2005).

The main concerns regarding dredge-mining impacts are: (1) the destruction of the benthic habitat through dredging; (2) suspension of sediment caused by the dredge tool, overspill from the dredger (vessel) and discharge of 'fines' form land-based treatment plants; and (3) the eventual settlement of suspended material. With regard to the first concern very little can be done to mitigate negative impacts. However, possible mitigation measures include: (1) designation of marine protected areas in which mining is precluded; (2) assessment of benthic communities to identify areas that are particularly sensitive to dredging or sustain commercially important stocks that should be avoided, and (3) the temporary cessation of dredging if a predetermined level of damage is detected – e.g. a certain number of animals (e.g. 100 lobsters) are dredged up in a specified time period (Pulfrich 2004).

The last two possibilities are somewhat controversial because of the implied trade-off between economic benefits and ecological impacts. However, studies have shown that benthic communities do recover after dredging has ceased and that in some cases the secondary succession in dredge areas give rise to communities that are more diverse than before (Conner & Simon 1979, Bonsdorff 1980, Poiner & Kennedy 1984, Steffani *et al.* 2003). This can serve as the basis for a solution unless, of course, the habitat is home to threatened or endangered species.

Ellis (1996) quantified the recovery rate of various benthic habitats after dredging and found that recovery time was dependent on the geomorphology of the benthos (Table 1.1).

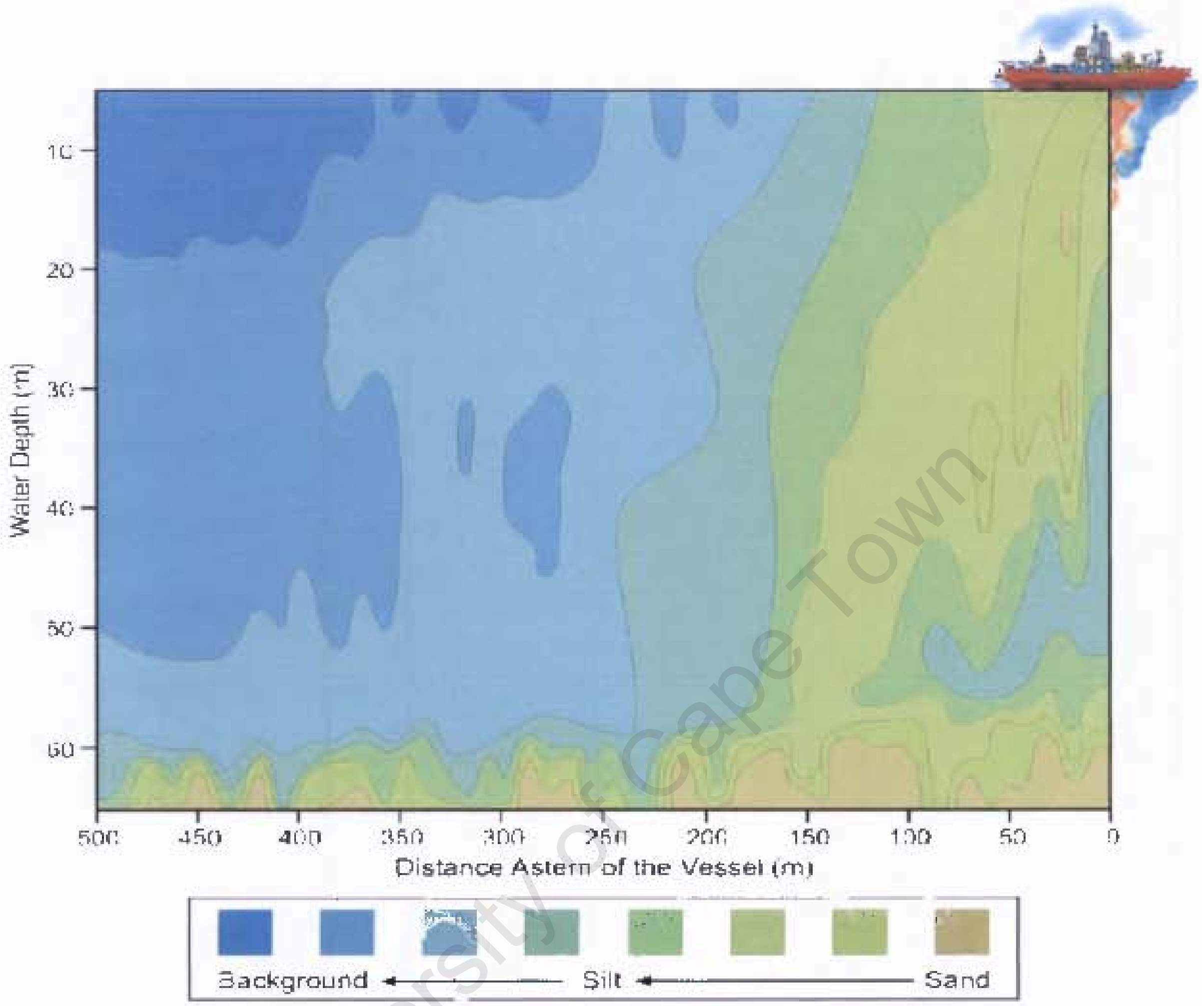


Figure 1.5: Acoustic Doppler Current Profiler (ADCP) longitudinal section along a tailings plume astern of a marine diamond mining vessel off southern Namibia, showing the depth and distance distribution of sediment particles of various size fractions behind the vessel (Penney & Smith 2004). Background readings refer to natural turbidity profiles resulting from the suspension of particulate organic and inorganic materials.

When considering the 2nd and 3rd concerns regarding the impacts of dredge-mining activities, sediment grade becomes an important factor. Fine sediments, such as silt and clay (<125 µm), are easily suspended and remain in suspension for longer periods of time than larger-grade sediments, such as sand (63 µm-2 mm) and gravel (>2 mm); as a consequence fine sediment can be displaced greater distances by currents (Fig. 1.5). This infers that silt and clay, rather than sands and gravels, are the main contributors to turbidity in the water column near dredging sites.

With regard to sediment settlement, two negative impacts are of concern: (1) scouring of sessile animals by coarse-grained sediment (a feature more prominent close to the coast and in areas with strong current regimes), and (2) siltation and smothering of benthic organisms by intermediate and fine grade sediment (Carter 1995). In addition, specific to the Benguela region, is the concern that if biogenic muds are agitated, as may happen during dredging in areas of muddy overburden, hydrogen sulphide will be released into the water column. This, in turn, can have catastrophic ecological impacts (Bailey *et al.* 1985). The significance of sediment laden with hydrogen sulphide will be discussed in the following section.

Table 1.1: Recovery time of different benthic types after dredging (Ellis 1996).

Sediment type	Recovery time
Fine-grained deposits [mud, silt and clay which contain some rocks and boulders].	1 year
Medium-grained deposits [sand which can contain some silt, clay and gravel].	1-3 years
Coarse-grained deposits [gravels which can contain some finer fractions of sediment and some rocks and boulders].	5 years
Coarse-grained deposits [gravels with many rocks and boulders].	>5 years

Dissolved oxygen and hydrogen sulphide

Dissolved Oxygen

In the coastal regions of the Benguela episodic and geographical variation in dissolved oxygen concentration is a natural feature of the system. A host of factors (ranging from remote wind forcing to localized upwelling) influence the dissolved oxygen profile over the Benguela continental shelf (Bailey *et al.* 1985, Monteiro *et al.* 2006, Van der Lingen *et al.* 2006). From a geographical perspective there is a general decrease in dissolved oxygen concentration over the shelf region from north to south (Fig. 1.6). Whereas water from the southern Benguela is naturally 80-85% saturated with oxygen (4.8-5.2 mg O_2/ℓ), water from the northern Benguela is normally 63-80% saturated (3.8-4.8 mg O_2/ℓ) (Shannon & O'Toole 1999). This is believed to be due to higher levels of primary production in the northern Benguela than in the southern Benguela (Fig. 1.6).

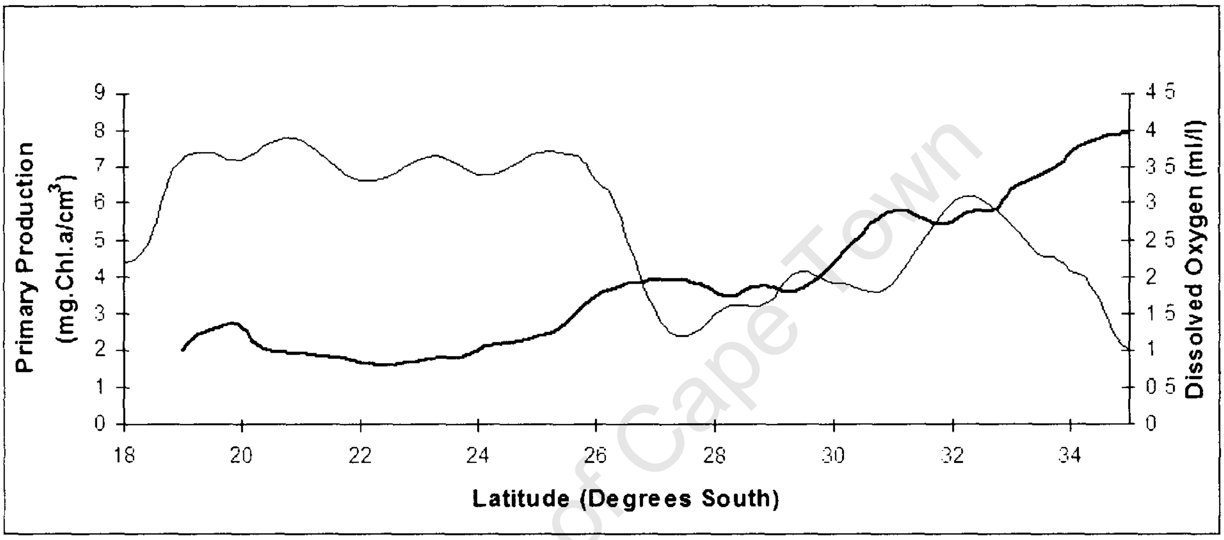


Figure 1.6: Dissolved oxygen (mℓ/ℓ) and primary productivity (mg Chlorophyll a/cm^3) along the coast over the Benguela shelf region (adapted from Dingle 1995). The thick line represents the dissolved oxygen profile and the thin line primary productivity.

Bailey *et al.* (1985) investigated the processes thought to be responsible for the state of the dissolved oxygen at Lüderitz and found that there are four modes of oxygen removal from the water. The first and most significant is respiration (3.04 x 10^3 mg/ℓ). This was followed by the removal of oxygen by the sediment (1.92-2.70 x 10^3 mg/ℓ), then the removal by decomposition, and fourthly, the loss of dissolved oxygen across the surface, i.e. at the interface of sea and atmosphere. In this study Bailey *et al.* (1985) made a clear distinction between oxygen used during normal respiratory processes and that used mainly by bacteria during decomposition of organic matter. Removal by sediment (known as the sediment oxygen demand) refers to the consumption of oxygen by bacteria and other microscopic organisms associated with sediments of the seabed, during the breakdown of dead organic matter (Bailey *et al.* 1985, Van der Plas *et al.* 2004).

Events of extreme oxygen depletion generally coincide with periods of intensified coastal upwelling and usually occur downstream from the heart of upwelling cells (Fig. 1.2). Upwelling brings nutrients as well as dormant photosynthetic algae from the aphotic zones to the well-lit coastal waters. The increased temperature and lighting conditions in these shallow regions cause the microscopic algae to come out of their dormant cyst phase and start growing (Fernández-Tejedor *et al.* 2004). During the active phase algae grow rapidly and multiply at exponential rates resulting in extremely dense concentrations of microscopic organisms in the surface waters, sometimes to the point where they become visible and are referred to as algal blooms. These algal blooms exert exceptional demands on the surrounding resources and as a result the nutrients supplied by upwelling soon become depleted. When this happens, a large component of the algal community, the component that is not advected off-shore into cooler waters, exhausts internal stores of nutrients and expire as respiration exceeds photosynthetic rates (Fernández-Tejedor *et al.* 2004).

When algal cells die they are no longer able to maintain their position in the water column, achieved through the active motion of flagella, and sink to the ocean floor. This constitutes an input of nutrients to the benthos and as a result additional increases in microscopic organisms surviving on the catabolism of dead organic matter, which leads to substantial consumption of dissolved oxygen via respiration (Van der Lingen 2004). It is this consumption that largely dictates the hypoxic status of the shallow bottom waters. In addition to this, certain dinoflagellate species produce biotoxins as they grow and multiply, and are referred to as harmful algal blooms. These toxic algal blooms result in substantial deaths of filter-feeding fauna, which in turn add to the benthic nutrient input, further escalating the overall reduction of dissolved oxygen – sometimes to the point of complete anoxia (Bailey *et al.* 1985, Shannon & O'Toole 1999, Fernández-Tejedor *et al.* 2004).

Under normal algal-bloom conditions populations of lobsters become concentrated in shallow, well-aerated regions, as deeper waters become increasingly oxygen depleted. However, during harmful algal blooms anoxic conditions are more widespread over the coastal shelf region, forcing large numbers of lobster to concentrate in the only constantly aerated area of the coast – the intertidal area. This often results in lobsters becoming stranded during low tide, and can lead to the loss of thousands of tons of an otherwise utilizable resource, equating to losses in excess of R100 million (Newman & Pollock 1971, Cockcroft *et al.* 1999, Cockcroft 2001). The main causes of death for stranded lobsters are predation and exhaustion, linked to asphyxiation, and desiccation as well as crushing, trampling and removal by people (Bailey *et al.* 1985, Matthews & Pitcher 1996, Shannon & O'Toole 1999, Cockcroft 2001).

Hydrogen Sulphide

In aquatic environments the disappearance of oxygen is often followed by the formation of hydrogen sulphide (Theede *et al.* 1969, Shooter 1999). Under anoxic conditions certain anaerobic bacteria use sulphate as a base (Eq. 1) during respiration (Theede *et al.* 1969, Shooter 1999, Cuomo *et al.* 2003). This releases sulphide anions into the water that subsequently react with free hydrogen cations (Eq. 2) to form gaseous hydrogen sulphide (Shooter 1999). In the ocean additional contributions to the dissolved sulphide in the system are made through the production of carbonyl sulphide (Elliot *et al.* 1987) and by phytoplankton (Shooter 1999). In marine systems the most significant concentration of hydrogen sulphide is found in poorly aerated and soft sediment – i.e. slime, silt and mud (Cuomo *et al.* 2003).

$$SO_4^{2-}{}_{(aqueous)} \leftrightarrow S^{2-}{}_{(aqueous)} + 2O_{2\,(gaseous)} \qquad \text{...Equation 1}$$

$$S^{2-}{}_{(aqueous)} + 2H^{+}{}_{(aqueous)} \leftrightarrow H_2S_{\,(gaseous)} \qquad \text{...Equation 2}$$

In the Benguela, hydrogen sulphide is mainly restricted to the northern regions, between the Orange River mouth and Cape Frio (Cushing 1971, Bailey *et al.* 1985, Lutjeharms & Meeuwis 1987, Garzoli & Gordon 1996, Boyer *et al.* 2000). Bailey *et al.* (1985) found water-soluble sulphide concentrations of 10-30 mg/ℓ in the sediment of certain areas of the northern Benguela. These regions of hydrogen sulphide laden sediment are limited to depths shallower than the 200-m contour and are referred to as the diatomaceous/biogenic mud-belt layer (Lutjeharms & Meeuwis 1987, Emeis *et al.* 2004).

Seasonally variable eruptions of hydrogen sulphide gas from the sediment are common along the Namibian coast and are believed to be part of the natural cycle of the Benguela (Emeis *et al.* 2004, Weeks *et al.* 2004). Many views exist about why these eruptions occur, but the pervasive view is that gas build-up in the sediment exceeds the hydrostatic pressure exerted on the sediment by the water above it, resulting in hydrogen sulphide gas bubbling out of the sediment (Garzoli & Gordon 1996, Boyer *et al.* 2000, Emeis *et al.* 2004). Although being a natural feature of the Benguela, these gas eruptions have significant socio-economic effects on the fishing industries as they affect the densities of commercial marine species (Weeks *et al.* 2004). In addition to this, geographical regions on the shelf with the highest incidence of observed eruptions coincide with nursery grounds of important pelagic and demersal fish species (Emeis *et al.* 2004).

Hydrogen sulphide is very poisonous, having negative effects on a host of eukaryotic organisms (Theede *et al.* 1969, Hahlbeck *et al.* 2000, Laudien *et al.* 2002, Searcy & Peterson 2004). Experiments done on rats found that exposure to 1100 mg/m^3 (hydrogen sulphide gas in air) resulted in death within 12 minutes of exposure (Beck *et al.* 1979) and that prolonged exposure (30 minutes a day for five days) to lower levels of hydrogen sulphide (28-280 mg/m^3) caused severe health conditions encompassing cardiovascular, neurological, metabolic, reproductive and developmental defects (Kosmider *et al.* 1967). Searcy & Peterson (2004) found that concentrations of >10 μMols (≈ 34.08 mg/m^3 in air) are generally toxic to eukaryote cells. However, certain species are able to survive in areas with high levels of hydrogen sulphide and in some cases, as along the hydrothermal vents of the mid-Atlantic Ridge, even utilize hydrogen sulphide for survival (Theede *et al.* 1969, Hahlbeck *et al.* 2000, Brüchert *et al.* 2003, Searcy & Peterson 2004). In the Benguela context discoveries during the late 1990s off the Namibian coast revealed the presence of macroscopic (sometimes 2 mm in diameter) sulphur oxidizing bacteria, *Thiomargarita namibiensis*, that are found throughout the biogenic mud-belt and are believed to play an important role in maintaining the precarious equilibrium through the detoxification of sulphide produced in the region (Brüchert *et al.* 2003).

Temperature and pH implications

In aquatic environments, temperature plays an important role because the high heat capacitance of water presents problems with regard to thermal regulation. Being a homeotherm in an aquatic environment means having to prevent succumbing to the environment as a result of the physiological strain induced by the loss of heat to the surrounding water (Schmidt-Nielsen

1997). Smaller animals, like crustaceans and fish, have a larger surface-area-to-volume ratio than larger organisms, such as marine mammals and birds, and would thus run a greater risk of heat-loss to the environment if they were to maintain a temperature even slightly higher than that of the surrounding water (Schmidt-Nielsen 1997). By conforming to their surroundings they increase their chances of survival by decreasing the physico-chemical difference between them and the environment, and ultimately the strain exerted on them by the environment. However, the negative implication of this is that the temperature of the surrounding water normally dictates the rates of biological processes such as respiration and enzymatic reactions. At the cellular level, the temperature of the environment even dictates the permeability of diffusion membranes, such as those found in gills and around cells (Schmidt-Nielsen 1997).

This last point is of particular importance as it signifies the absolute dependence of survival for poikilotherms on the temperature of their environment. Amongst other things, it dictates the rates at which organisms can escape (Schmidt-Nielsen 1997). The same applies to scenarios of poisoning. Theede *et al.* (1969) found that an indirect relationship exists between tolerance period and temperature, in that most animals survive for much longer periods of time at a given toxic level of hydrogen sulphide and oxygen deficiency if they are held at lower temperatures than at higher temperatures. Thus, temperature plays a very crucial role in aquatic environments.

On the other hand, pH plays a somewhat minor role, especially in marine environments. Like salinity, pH is a conservative physical parameter and (under natural conditions) is extremely well buffered within the marine alkaline to neutral range of eight and seven (Schmidt-Nielsen 1997, Shooter 1999). However, in the case of the Benguela ecosystem, hydrogen sulphide eruptions over the continental shelf can severely skew this natural range towards the acidic side of the scale. As hydrogen sulphide is liberated into the water column from the underlying sediment the concentration of H_2S molecules increases relative to the HS^- and H^+ ions in the water to the point where the potential difference between the two species forces the dissociation of H_2S molecules into HS^- and H^+ ions (Eq. 3) to reduce the difference on opposite sides of the equilibrium – arrow/reaction (Theede *et al.* 1969, Shooter 1999).

$$H_2S_{\text{(gaseous)}} \leftrightarrow HS^-_{\text{(aqueous)}} + H^+_{\text{(aqueous)}} \qquad \text{...Equation 3}$$

The liberation of protons (H^+ ions) into the water reduces the localized pH of the water making it more acidic. This has profound implications for the toxicity of hydrogen sulphide due to the compounding effect of the increase in acidity together with an increase in the concentration of sulphide in the water. This has been substantiated in studies done by aquaculturists on the effects of hydrogen sulphide on various fish species that found that over a decrease of four pH levels the toxicity of hydrogen sulphide increased from 1% (at a pH of 9) to 99% (at a pH of 5) since pH is correlated to hydrogen sulphide in a negative exponential relationship (http://www.aquatext.com/tables/h2s.htm#toxic 2005). This implies that as the pH decreases the toxicity of hydrogen sulphide in the water increases exponentially.

Project Objectives

As part of the 2005 (Chameis Bay) pre-feasibility study by De Beers Marine Namibia for a new diamond dredge-mining proposal in the mid-water regions off the Namibian south coast, a complete impact assessment was conducted to assess and manage any potential negative effects that the proposed dredge-mining might have on the surrounding environment. The current study was proposed to gauge the possible impacts of any secondary effects dredge-mining might have on juvenile rock lobster populations in and around the area of the proposed dredging. The experiments formulated were laboratory-based and were designed to assess the effects of (1) reduced oxygen levels, (2) elevated hydrogen sulphide, and (3) the effects of turbidity on the behavioural responses of juvenile lobsters. The main focus of this study was on the short-term impacts of changes in these variables associated with dredging.

Environmental stresses such as these will influence the physiological functioning of organisms including processes such as photosynthesis and metabolic rate, and nervous and hormonal responses. In turn, these physiological reactions will lead to alterations in behaviour and, if they are severe enough, to incapacitation and even death (Evans 1999). I elected to monitor behavioral responses to hypoxia, hydrogen sulphide and turbidity rather than physiological responses because behavioural responses observed in the laboratory are more readily applied to and monitored under field conditions than are physiological responses.

Against this backdrop, the project had the following objectives:

1. To conduct a literature-based assessment of the natural variability in turbidity, dissolved oxygen and hydrogen sulphide conditions in the Benguela, and an analysis of how diamond-dredge mining in the mid- to deep-water regions could change these variables, with particular relevance to the proposed mining areas along the southern coast of Namibia.
2. To conduct a laboratory experiment to establish the behavioural responses of juvenile rock lobsters to reduced levels of dissolved oxygen and elevated hydrogen sulphide.
3. To undertake laboratory experiments to determine the behavioural responses of juvenile lobsters to turbidity, and whether turbidity is correlated with any other physical variables under laboratory conditions.

4. Using the information obtained from work done to fulfill the three preceding objectives of the study to develop rational measures for mining-related impact mitigation.

The thesis is structured in the following manner. The present chapter introduces the thesis, reviews relevant published information and outlines the objectives. Chapter 2 reviews data from two studies done in the Chameis Bay area with regards to natural and mining-induced variability in water temperature, oxygen concentration and turbidity. Chapter 3 focuses on laboratory studies done on the effects of diminished oxygen and elevated hydrogen sulphide on the behaviour of juvenile rock lobsters. Chapter 4 covers a laboratory study on the effects of turbidity on rock lobster behaviour and the implications for diamond-mining. Chapter 5 concludes the thesis with an overview of the project findings, outlines limitations in the present work, and discusses recommendations to De Beers Marine Namibia regarding mitigation of mining impacts and the potential for future research.

CHAPTER 2: NATURAL AND MINING-RELATED VARIABILITY IN CHAMEIS BAY

Introduction

Natural fluctuations in physical variables (spanning hourly to inter-annual timeframes) constitute a major part of the characteristics of marine ecosystems, and are an essential baseline against which the magnitude of anthropogenic influences can be gauged. In line with this, a review of the physico-chemical variability in and around the intended diamond mining areas of De Beers Marine Namibia (DBMN) was proposed. Acknowledging that it is important to distinguish between natural and anthropogenic effects, the separation of the two was beyond the scope of this thesis. However, existing literature was sourced and used to provide a general backdrop for the research incorporated in this study (Chapters 3 & 4), as well as serving to provide pointers for future research (Chapter 5).

Two main sources of data were used to achieve the objectives of this study: (1) an unpublished report by CSIR on the physical effects of sediment-release from land-based paddocks and marine dredge-mining vessels, which focussed mainly on sediment disturbance caused by dredging and the dynamics of resultant plumes (CSIR 2004, 2005b & c), (2) a Fisheries Independent Monitoring Survey (FIMS) done by Pisces Environmental Services for the DBMN pre-dredging Environmental Impact Assessment of the Atlantic 1 and Chameis Bay areas (Pulfrich *et al.* 2006). The CSIR study was conducted from the region just north of South Rock (Fig. 2.1 and 2.2) down to Kerbe Huk (CSIR 2005b), while the FIMS data sourced from DBMN covered the region from Green Precipice (Fig. 2.1) to Mittag (Pulfrich *et al.* 2006) north of the Orange River mouth (Fig. 2.2). The FIMS was carried out with the main objective of evaluating rock lobster abundance, recruitment and distribution patterns within the Chameis Bay area as well as beyond the southern limit of the Chameis Bay mining area (Pulfrich *et al.* 2006).

Materials and Methods

Data Attributes

The two studies on which the majority of this review is based were done by two different institutions and thus the nature of the data differs somewhat. The FIMS were conducted on a seasonal basis (i.e. once every fourth month) over a period of 16 months, from January 2005 to April 2006 (Pulfrich *et al.* 2006). The coastline from Easter Point down to the Orange River mouth (Fig. 2.1) is divided into 16 commercial fishing zones based on the existing Namibian rock-lobster fishing industry (Grobler, MFMR, Namibia 1995, Pulfrich *et al.* 2006). For the 2005-2006 FIMS only the region between Dreimaster Punt and the Orange River mouth (Zones 14 and 15) was focussed on, with Zone 14 being divided into a further three sub-zones (14A, 14B & 14C – Fig. 2.1). Initially the whole of Zone 14 (14A, 14B & 14C) was considered for the FIMS, but after the baseline survey in 2005 it was decided to exclude the northern-most region (Zone 14A) from the survey (since no lobsters were caught there) and to focus on the higher priority zones – i.e. Zones 14B and 15. Zone 14B had six transects, 14C four transects and Zone 15 six transects. Each transect had four sampling stations (1-4) from nearshore to offshore, respectively (Fig. 2.2).

During the six quarterly surveys physical parameters as well as lobster catches were recorded at each sampling station. Physical parameters monitored included wind attributes, wave attributes, bathymetry, dissolved oxygen and water temperature (Pulfrich *et al.* 2006). However, the data sourced from DBMN for the purpose of this study included rock lobster catches, bycatches, wind attributes, swell height data, temperature and dissolved oxygen from zones 14B, 14C and 15 (Fig. 2.1). For this review, only region 14B was focussed on, as it was the site of the mid-water dredging trial and coincided with the CSIR plume profiling study. Gaps in the data sourced for zones 14C and 15 covering the September 2005 and January 2006 FIMS made it difficult to incorporate these two zones in the review.

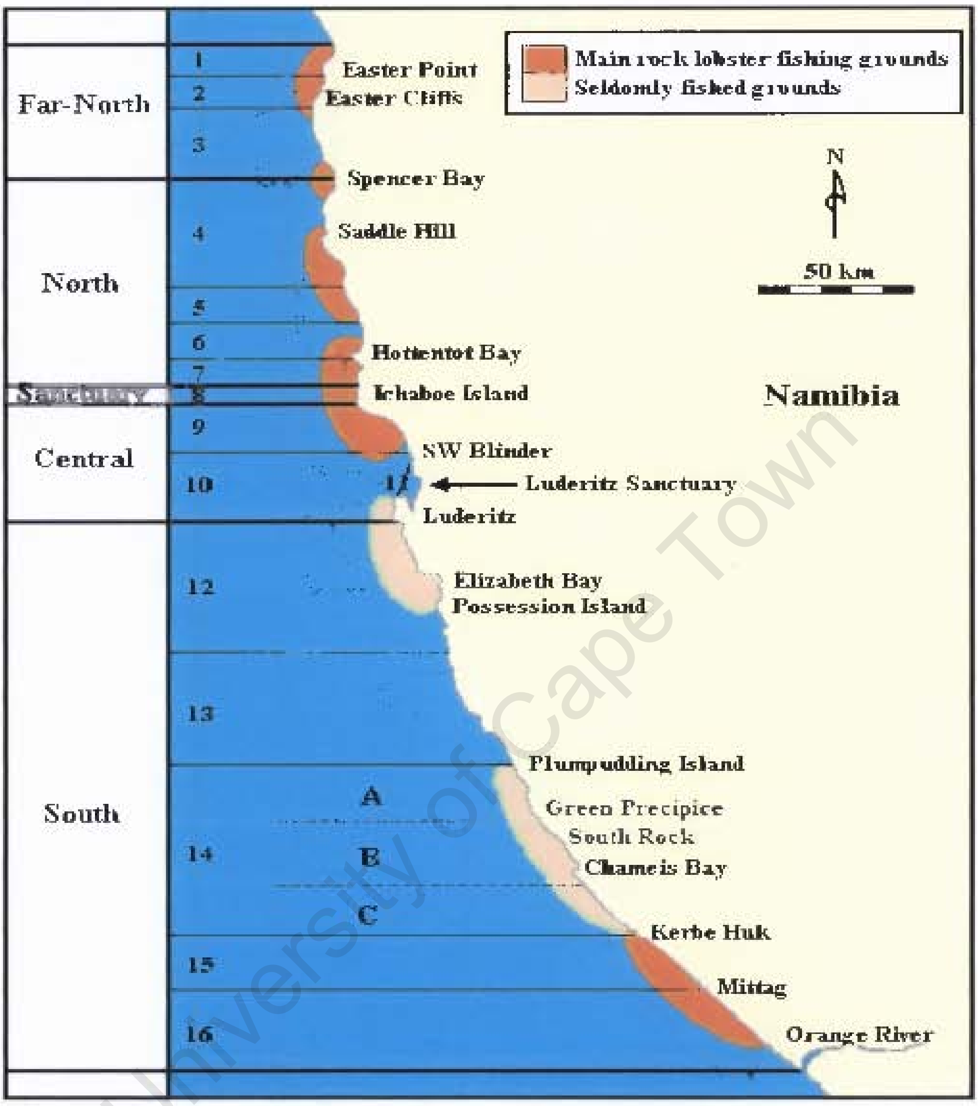


Figure 2.1: Map showing the Namibian commercial rock lobster fisheries zoning system (adapted from Pulfrich *et al.* 2006). Data sourced cover only zones 14 and 15, and the data used in the review covered only section 14B.

The CSIR study was conducted in two phases: an initial short phase (conducted over a period of eight days), and a much longer study spanning a period of six weeks (CSIR 2005b). The short-phase study was conducted during the DBMN, Chameis Bay Dredge Trial, on dredging panels one and three (Figs 2.2 and 2.3); whereas the longer-phase study was conducted afterwards with the aim of validating mathematical simulations of turbidity plumes created by a hydrodynamic model (CSIR 2005b). For this review, only data from the short-term dredging-trial study were examined, as the longer-term data were limited to wind, wave, current attributes and simulated turbidity profiles and were thus not suitable for the review.

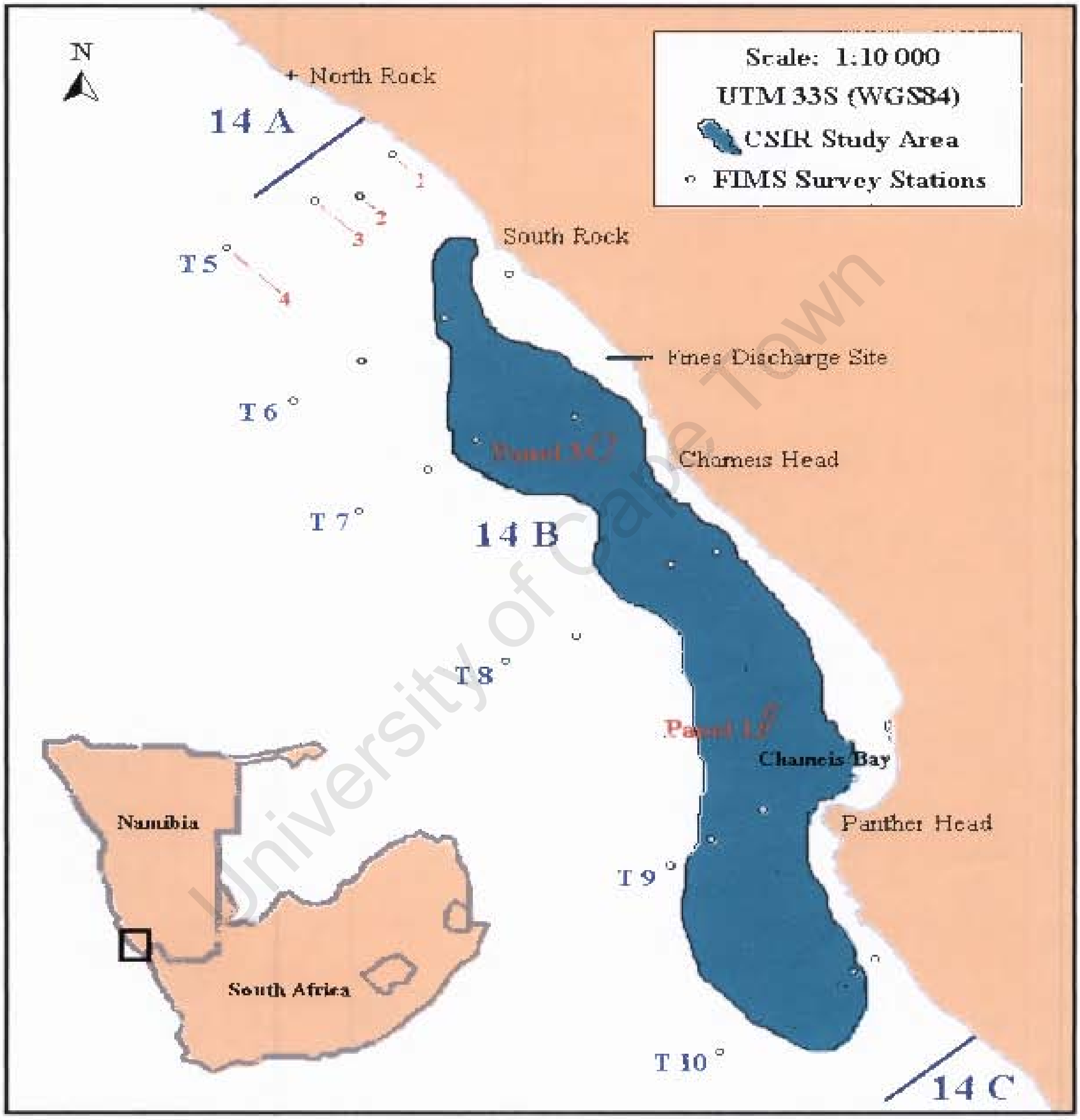


Figure 2.2: Location map of the study areas and dredge panels 1 and 3 of DBMN (Sources: CSIR 2005b & Pulfrich *et al.* 2006). Maps were superimposed to show the overlap of the two studies. Open circles represent the FIMS sampling transects (T5-T10) and the shaded region the CSIR study area. The blue lines and text (14A, 14B & 14C) indicate the FIMS grouping system – with the red font at Transect 5 (T5) indicating the station numbering system.

The short-term CSIR data comprised depth-correlated data from randomly selected control and mined areas at which *in-situ* turbidity, dissolved oxygen and temperature profiles were taken in and around the two dredging panels (Fig. 2.3). Data were recorded at each of the sampling or profiling points with a multi-parameter sensor (Seabird SBE 19) by lowering the device into the water at a rate of 0.5 m/s to the seafloor and then raising it again at the same rate (CSIR 2005c).

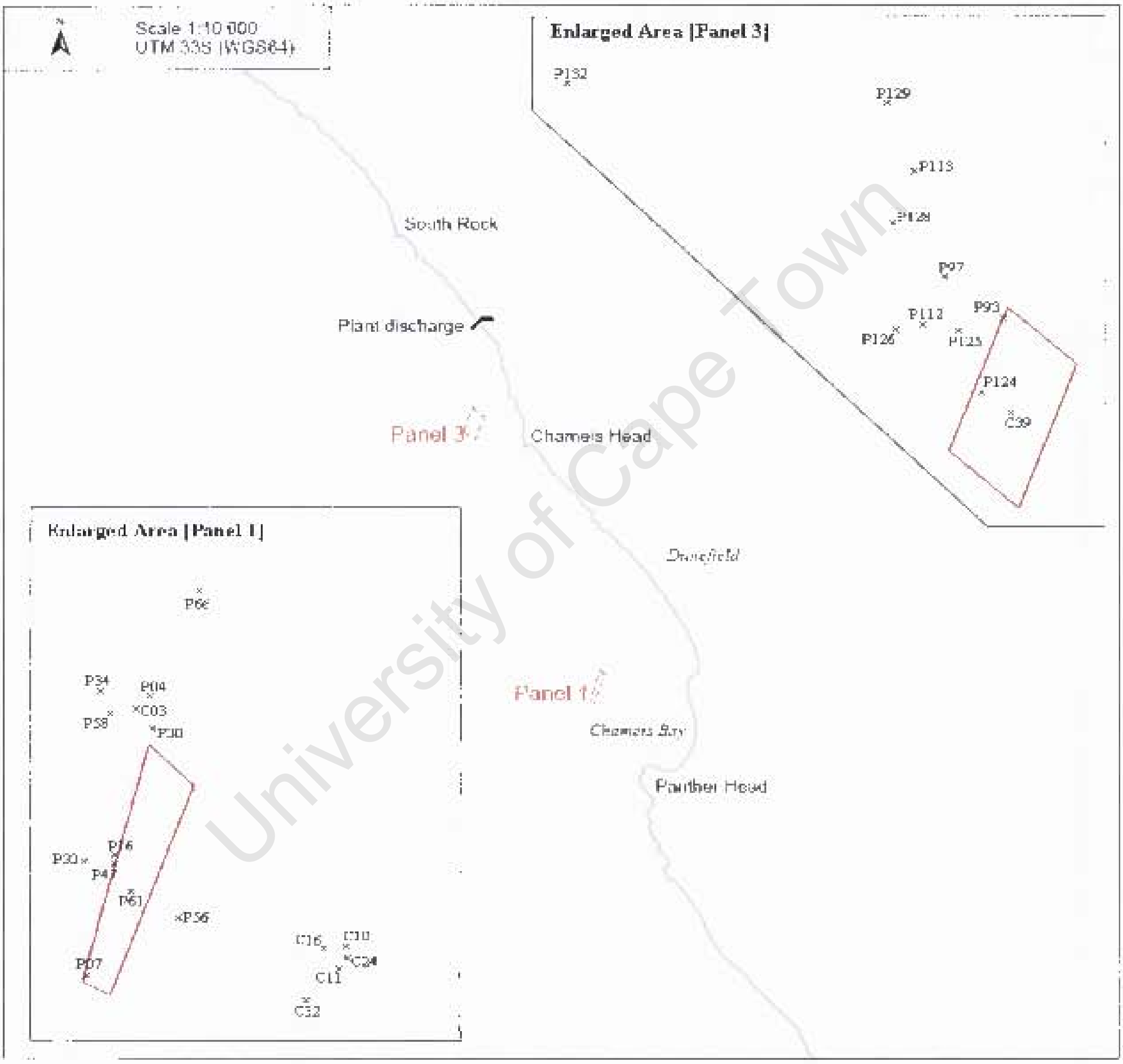


Figure 2.3: Visualization of profiling positions in and around dredging panels 1 and 3 (modified from CSIR 2005b). The positions of all the plume (denoted by "P") and control (denoted by "C") profiles used in the review are represented on the map.

The Seabird SBE 19 recorded depth, conductivity, temperature, dissolved oxygen as well as turbidity, all of which were time-referenced and later presented graphically (as exemplified in Fig. 2.4). Turbidity was recorded with an optical backscatter sensor (OBS) calibrated in mg/ℓ and was only sensitive to turbidity readings greater than 17.38 mg/ℓ (CSIR 2005c). Thus straight lines falling on a value of 17.38 in the profiling graphs actually reflect turbidity readings of between 0 and 17.38 mg/ℓ. The OBS had a detection radius of about 20 cm around its sensor (CSIR 2005c).

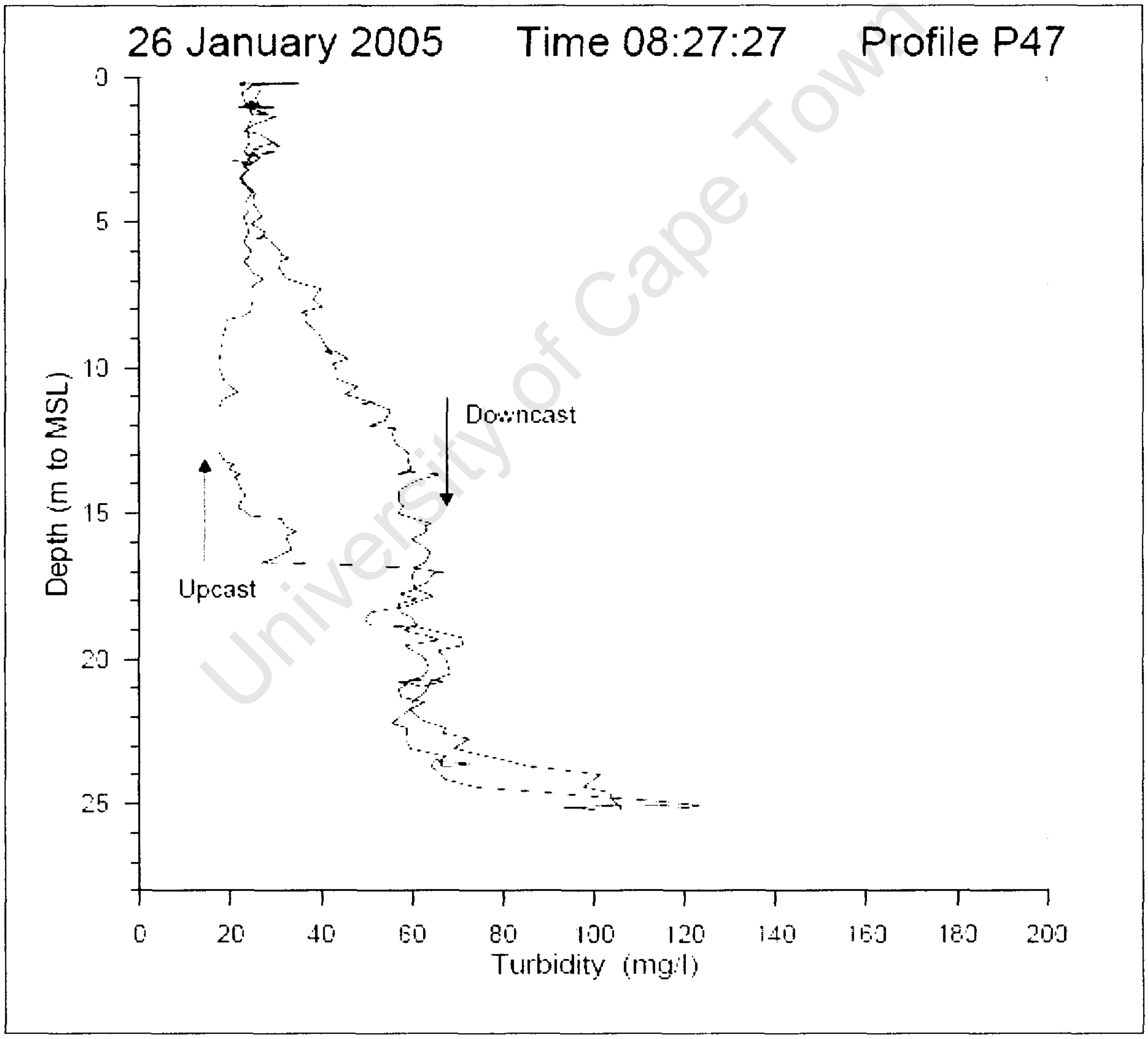


Figure 2.4: Example of the data out-put from the Seabird SBE 19 (CSIR 2005b).

For the review, three profiles per day were selected from the CSIR (2005b & c) study to allow a balanced comparison between control and mined areas. The aim of these comparisons was to assess any differences in turbidity, temperature and dissolved oxygen before, during and after mining within the areas of significance. The positions of all the profiles (i.e. three from each of the eight days of the study) used in the review are illustrated in Fig. 2.3. Profiles were done on different days, hence the apparent clustering and/or overlapping of profile positions in Fig. 2.3 should be viewed in conjunction with the dates on which sampling took place.

To illustrate the mining effects, the profiles obtained from the CSIR (2005b & c) study were classified into three categories: (1) Control profiles, which constituted those taken before dredging or at sites beyond the visible extent of the turbidity plumes created by dredging (CSIR 2005). Profiles sampled in dredging plumes were categorized as (2) Mined profiles while those sampled at the edges of visible dredging plumes or on the lee-current side of mining were categorized as (3) Impacted profiles. Dredging occurred intermittently during the 8-day study (CSIR 2005b), which made it possible to assess the three scenarios for each day of the study.

Data Extraction, Manipulation and Presentation

Fisheries Independent Monitoring Survey (FIMS)

For the assessment of natural variability the majority of the data used in the review were obtained from the FIMS (Pulfrich *et al.* 2006). For the seasonal variability unprocessed FIMS data sourced from DBMN for Zone 14B were used to construct seasonal bottom-temperature and dissolved oxygen graphs. Data from all six transects in Zone 14B were used for the seasonal comparisons with transects being pooled for each depth. Latitudinal or alongshore differences (i.e. among transects) were assessed from vertical profile charts sourced from the Pulfrich *et al.* (2006) report.

CSIR Study

The CSIR study was used mainly to illustrate the mining impacts on turbidity, temperature and dissolved oxygen within Chameis Bay. No raw data were available from the CSIR documents sourced from DBMN and thus methods had to be devised to extract information from already processed data in the documents. The profile graphs (e.g. Fig. 2.4) created from the data recorded by the Seabird SBE 19 for turbidity, temperature and dissolved oxygen were the most important data set for the purposes of this review.

Data were extracted from these profiles and used to construct synthesised depth profiles for the three treatments (i.e. control, mined and impacted) for the three variables assessed (i.e. turbidity temperature and dissolved oxygen). For the synthesised depth profile graphs the upcast and downcast data per profile (Fig. 2.4) were averaged to obtain mean values for each depth level. For the comparative analysis three depth intervals were defined: Top (upper 40%), Middle (40-80%) and Bottom (lower 20%), to standardise for depth differences between samples.

All statistical analyses and graphical representations were carried out in Statistica 7. Nonparametric Friedman ANOVAs were employed to assess inter-seasonal differences for both dissolved oxygen and temperature. For the turbidity, dissolved oxygen and temperature profiles, the three water-column depth strata (top, middle and bottom) and the treatment types (control, mined and impacted) were compared using 2-way ANOVAs, followed by Tukey HSD post-hoc tests.

Results

Natural Variability

Variability in bottom dissolved oxygen over an increase in depth was greater during Autumn, Summer and Winter than in Spring (Fig. 2.5). During the spring season of 2005 the dissolved oxygen concentration remained below 2 mg/ℓ over the entire depth range.

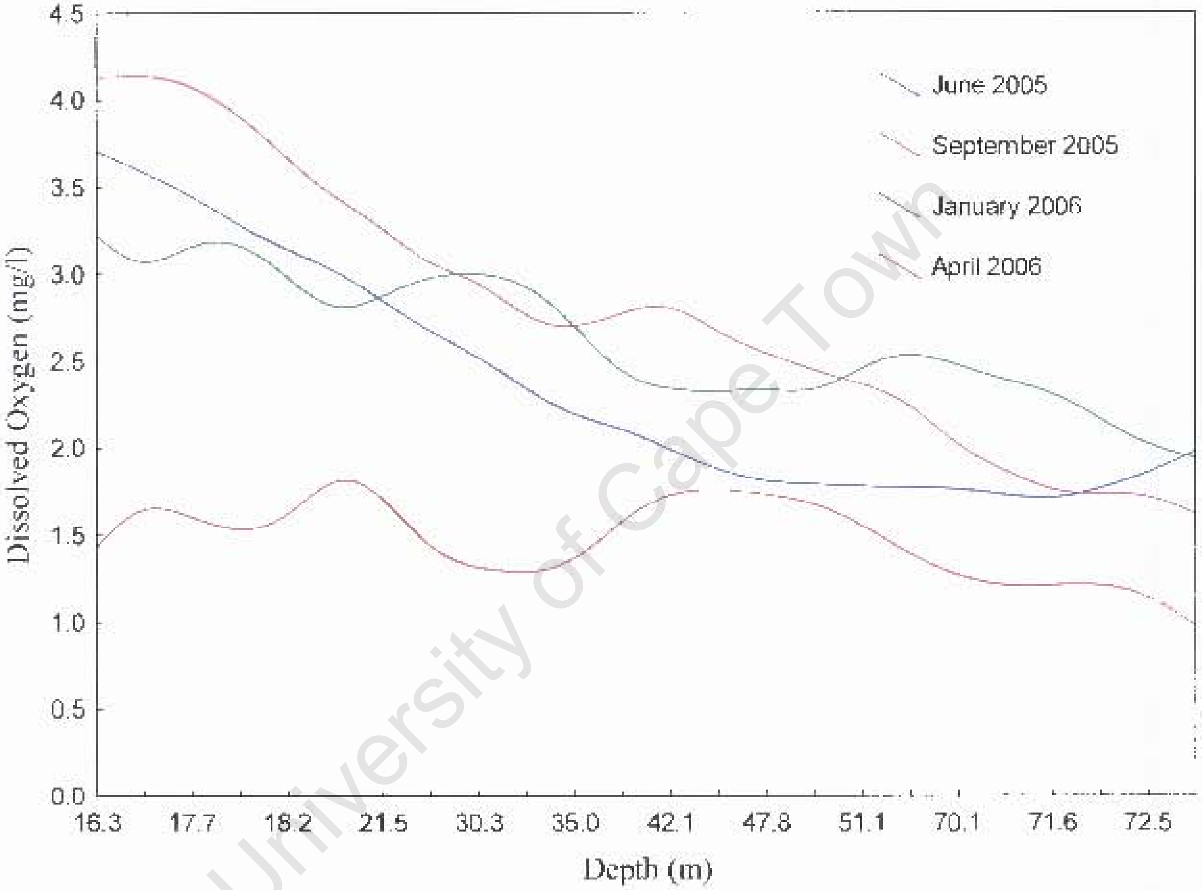


Figure 2.5: Bottom dissolved oxygen variability over depth in Chameis Bay – constructed from raw data sourced from DBMN. All six transects in Zone 14B were pooled across depths. The depth scale is non-linear because it was related to distance offshore, which spanned 1-10 km.

The inter-seasonal difference in bottom temperature between June 2005 and April 2006 across bottom depths of 16-72 m followed the natural cycle of a peak in April (Autumn) and dropped to a minimum in September (Spring) but differed little (<0.5°C) across these depths (Fig 2.6). As expected, bottom dissolved oxygen and bottom temperature decreased with an increase in depth, with this pattern being more subtle in the case of bottom temperature (Fig. 2.6) than for bottom dissolved oxygen (Fig. 2.5).

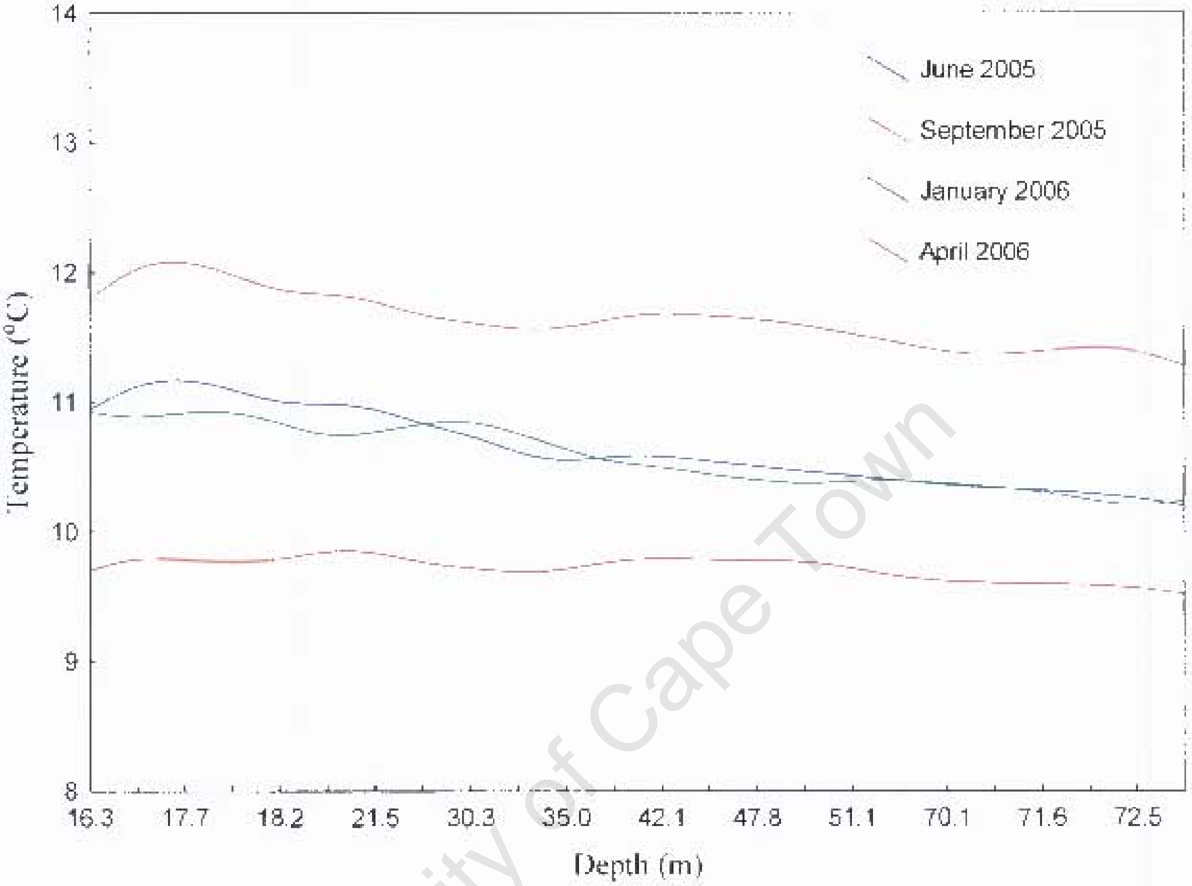


Figure 2.6: Bottom temperature variability over depth in Chameis Bay – constructed from raw data sourced from DBMN. All six transects in Zone 14B were pooled across depths. The depth scale is non-linear because it was related to distance offshore, which spanned 1-10 km.

Similar to bottom temperature the lowest values for bottom dissolved oxygen were recorded in Spring and the highest values during Autumn. The mean differences between the two seasonal extremes were 1.31 mg/ℓ for the dissolved oxygen and 1.92°C for the temperature (Table 2.1; Fig. 2.7).

Non-parametric Friedman ANOVAs showed that both oxygen and temperature differed significantly among seasons (Table 2.1). Post-hoc Tukey-type tests indicated that mean bottom temperatures in the Chameis Bay area were similar during Winter (June) and Summer (January) of the two consecutive years and both significantly different (p <0.0001) from the Spring (September) and Autumn (April) extremes – which were also significantly different from each other (p <0.001). The mean bottom dissolved oxygen readings were closely linked as the Winter, Summer and Autumn oxygen concentrations were statistically indistinguishable (p >0.05) with the only detectable difference being between Spring and Summer (p <0.0001).

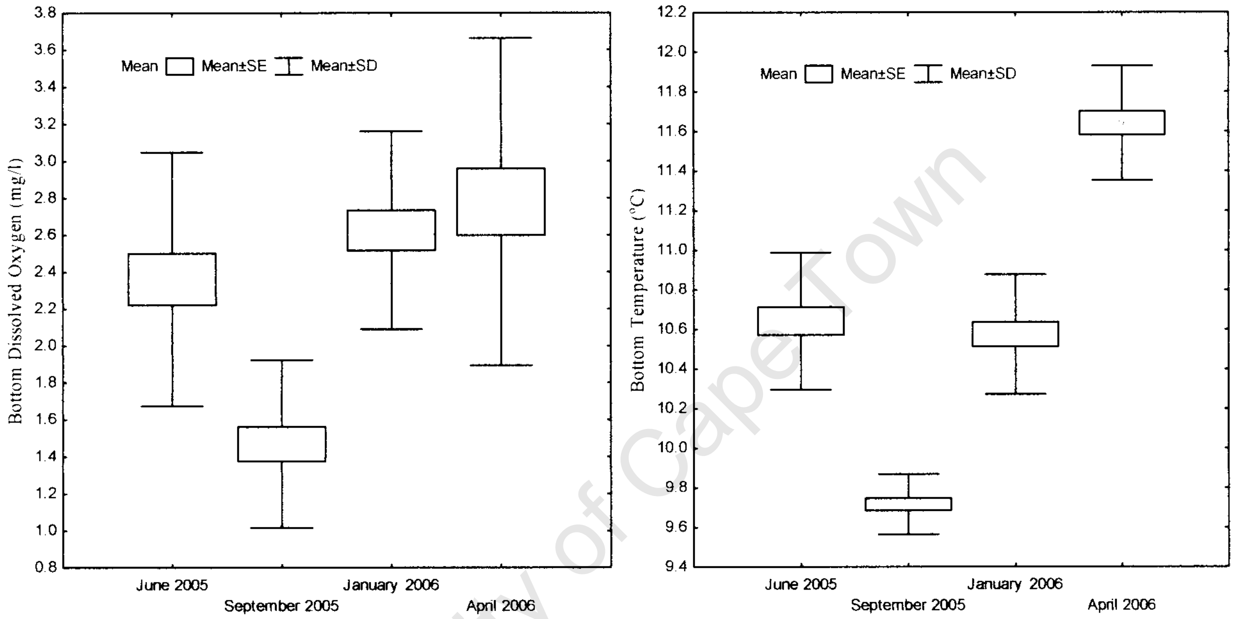


Figure 2.7: Seasonal means for both bottom dissolved oxygen and temperature variables (data constitute unprocessed data sourced from DBMN).

Table 2.1: Nonparametric Friedman ANOVA statistics for the seasonal bottom temperature and bottom dissolved oxygen variables within Chameis Bay (Pulfrich *et al.* 2006). [Chi Square stats for both variables: N = 24; DF = 3; p <0.0001].

	Mean	Rank	Standard Deviation	Chi Square
Bottom Oxygen [June 2005]	2.36 mg/ℓ	2.520	0.69	45.72
Bottom Oxygen [Sept. 2005]	1.47 mg/ℓ	1.041	0.45	
Bottom Oxygen [Jan. 2006]	2.63 mg/ℓ	3.187	0.54	
Bottom Oxygen [April 2006]	2.78 mg/ℓ	3.250	0.88	
Bottom Temperature [June 2005]	10.64 °C	2.500	0.35	64.80
Bottom Temperature [Sept. 2005]	9.72 °C	1.000	0.15	
Bottom Temperature [Jan. 2006]	10.58 °C	2.500	0.30	
Bottom Temperature [April 2006]	11.64 °C	4.000	0.29	

FIMS data for January 2005 (Fig. 2.8) for zone 14B were sourced from the Pulfrich *et al.* (2006) report for comparison with the CSIR study, also done in January 2005. The most striking pattern was an increase in the surface (0-6 m) dissolved oxygen concentration the further one moved from the coastline. Nearshore stations had dissolved oxygen values of between 3.2 an 4.4 mℓ/ℓ while offshore stations had values of between 4.6 and 7.4 mℓ/ℓ (Fig. 2.8).

In general, under natural conditions, the dissolved oxygen profiles near Chameis Bay showed a decrease in dissolved oxygen over an increase in depth. Dissolved oxygen values in the surface layers (0-2 m) varied between 7.4 and 3.4 mℓ/ℓ, declining to between 4.2 and 1.6 mℓ/ℓ near the seafloor. None of the profiles in zone 14B had bottom concentrations of below 1 mg/ℓ, although a review of the extended data set did reveal one reading <1 mℓ/ℓ, in zone 15, in line with the anoxic event during the 4th FIMS (Pulfrich *et al.* 2006). On average, the difference in dissolved oxygen between the surface and bottom (70 m) water regions ranged between 2 and 5 mℓ/ℓ (Fig. 2.8) at the stations furthest offshore.

Temperature, similarly, decreased with an increase in depth (Fig. 2.8). However, there was little variation in temperature below 30-40 m (see station 4 for each transect). Temperature profiles reached an asymptote of about 10.4°C at a depth of 76 m, and ranged between 13.8 and 11.4°C near the surface. Another pattern evident from the profiles was an increase in surface water (0-2 m) temperature southward along the coastline, in line with the general southerly increase in temperature within the Benguela system (Shillington 1998). The surface water temperatures followed the same pattern as the surface dissolved oxygen, increasing with distance offshore (Fig. 2.8). Nearshore stations had temperatures between 11.6 and 13.0°C while offshore stations spanned 12.8 to 14.2°C (Fig. 2.8).

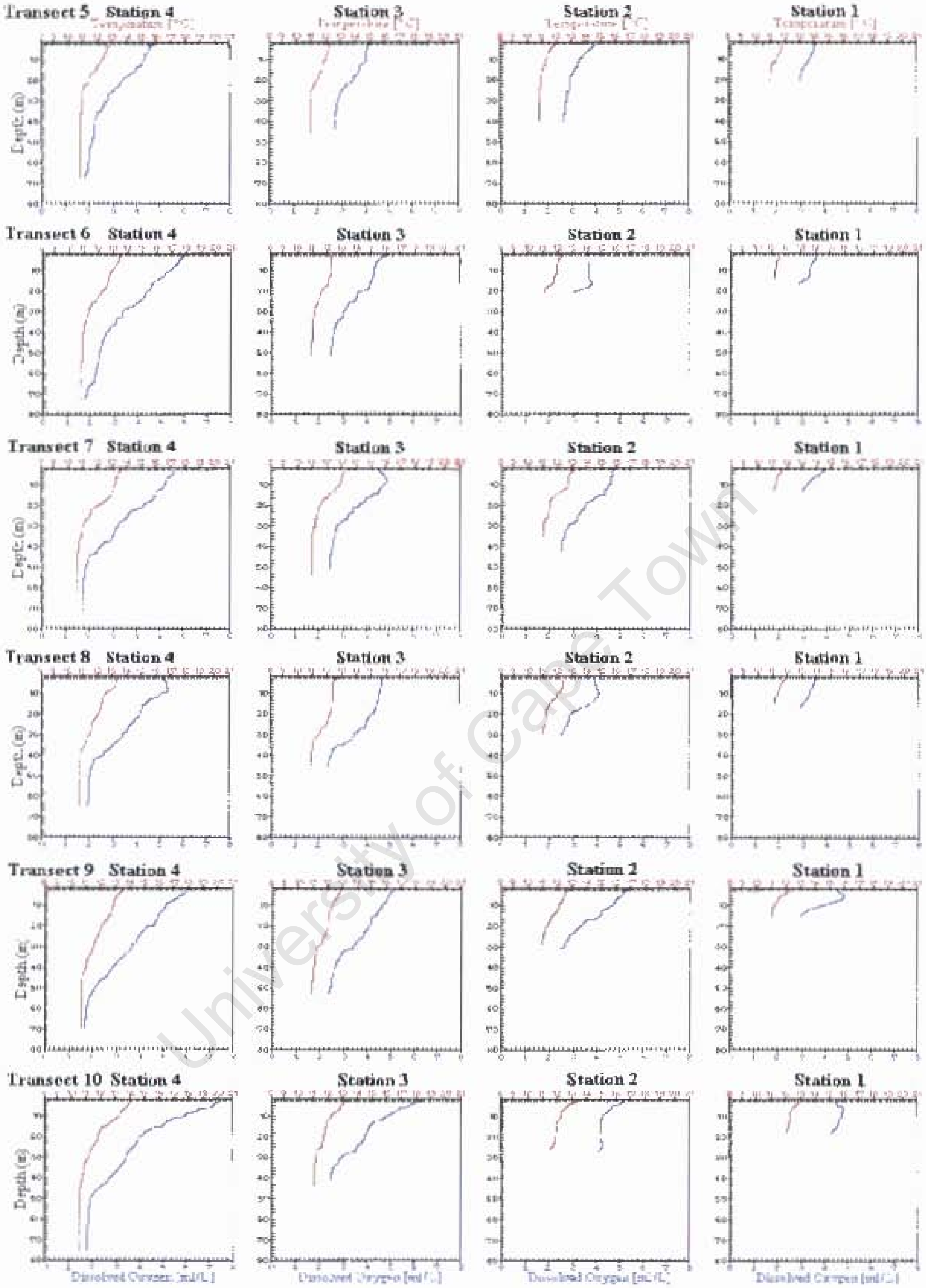


Figure 2.8: Dissolved oxygen and temperature profiles of January 2005 for the six transects in zone 14B (source: Pulfrich *et al.* 2006). Blue and red profiles show the respective dissolved oxygen and temperature variability over depth.

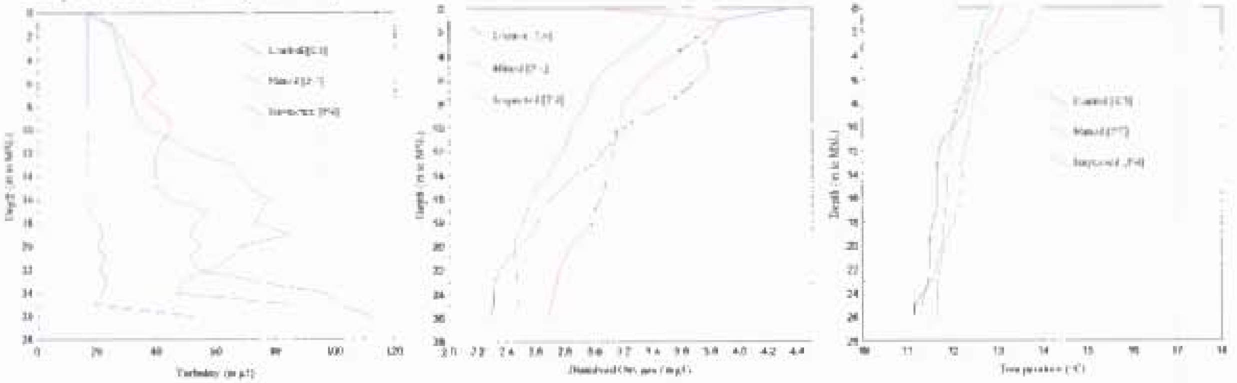
Mining Related Variability

Data on turbidity, temperature and dissolved oxygen were extracted from profiles in the CSIR (2005b) report and used to construct integrated depth-related profiles for each of the parameters assessed (Fig. 2.9).

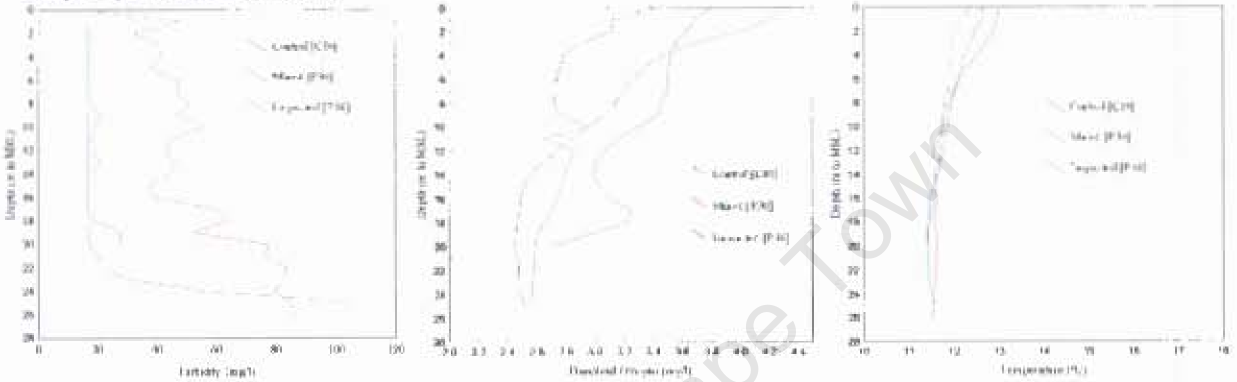
With the exception of the control profile measured on 28 January (Fig. 2.9), control sites had low values of turbidity relative to either impacted or mined profiles. Control profiles exhibited low levels of turbidity that were often below 17.38 mg/ℓ, from the surface down to the bottom of the profile, where a spike was usually recorded, due to disturbance of the sediment by either the instrumentation rig as it hit the ocean floor, or by water movements close to the seafloor (CSIR 2005c). Mined profiles, on the other hand, exhibited significantly higher readings either throughout the profile or, in most cases, from the mid-water depths down to the bottom, and had highly variable amounts of suspended material throughout the profile (e.g. mined profiles on 23, 24, 28, 29 and 30 January).

Dissolved oxygen and temperature profiles followed no easily distinguishable pattern(s). With the exception of the last three days (6, 7 & 8) of the CSIR study, temperature profiles for the control, mined and impacted sites were generally closely correlated throughout the depth ranges compared (Fig. 2.9). Day 7 exhibited the largest range in temperature difference between the surface and bottom depths where the control profile (C39) exhibited a decrease in temperature of roughly 6.5°C from the surface to a depth of 25 m. Dissolved oxygen results exhibited highly erratic profiles over depth with no obvious correlation between any of the three treatment sites (i.e. control, mined and impacted sites) for the duration of the CSIR study (Fig. 2.9).

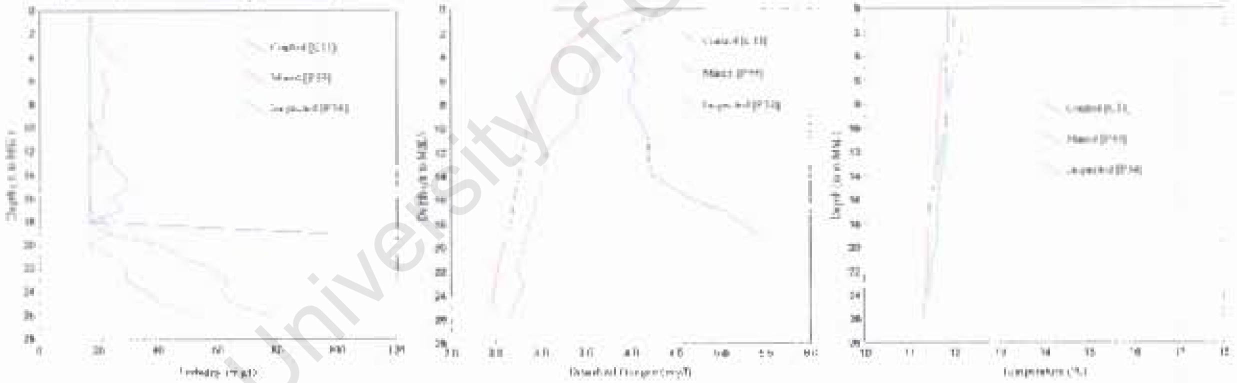
Day 1 (23 January 2007)



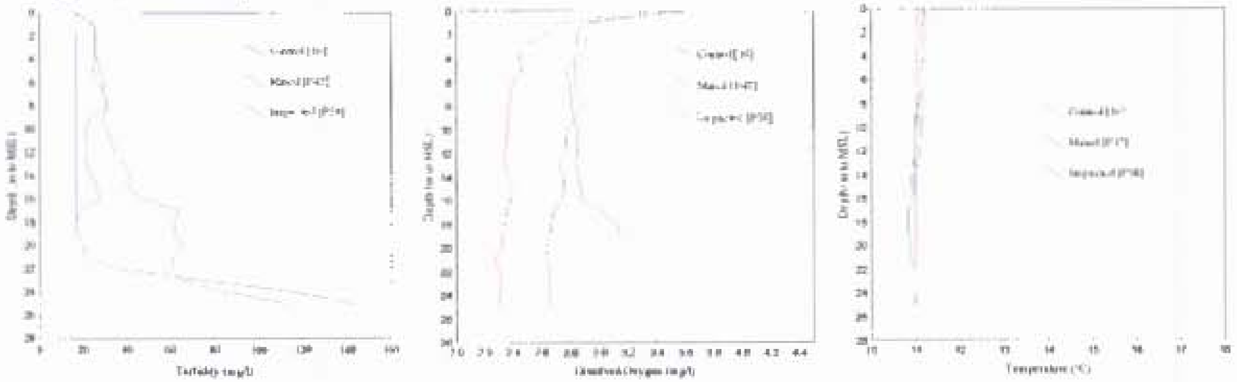
Day 2 (24 January 2007)



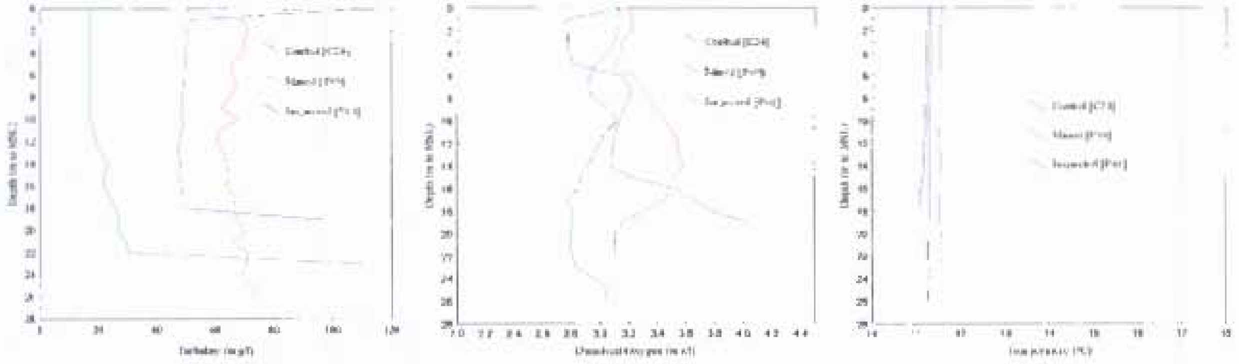
Day 3 (25 January 2007)



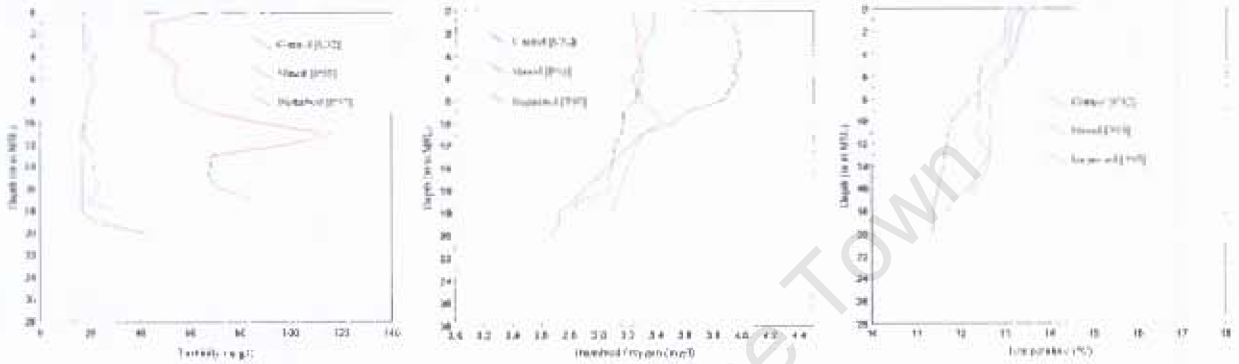
Day 4 (26 January 2007)



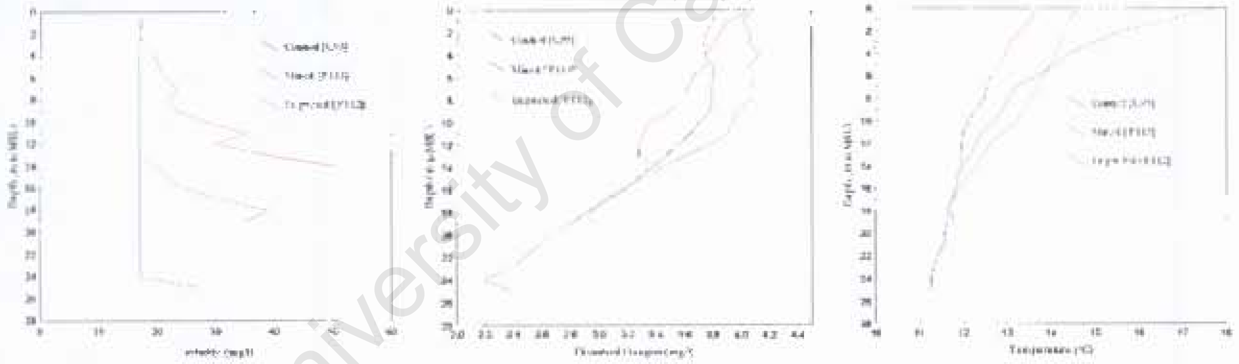
Day 5 (27 January 2007)



Day 6 (29 January 2007)



Day 7 (30 January 2007)



Day 8 (31 January 2007)

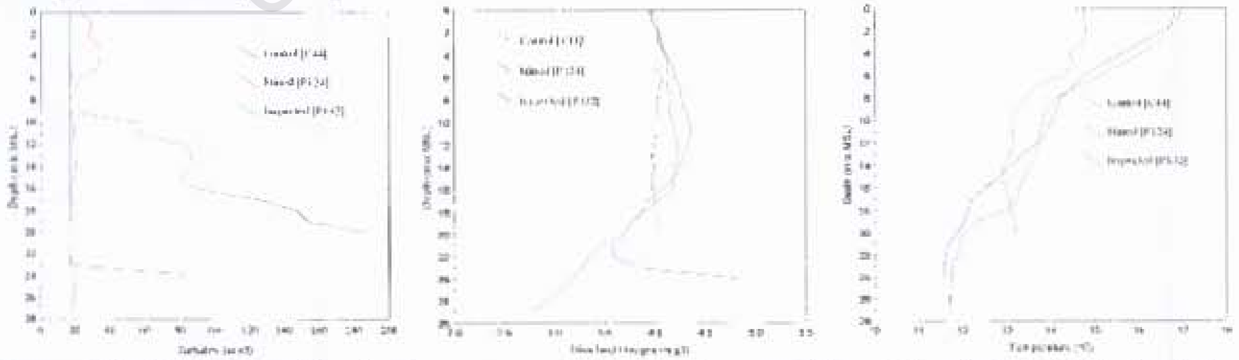


Figure 2.9: Turbidity, dissolved oxygen and temperature profiles of the 8-day CSIR dredging study in Chameis Bay (data extracted from the CSIR 2005c report).

Figures 2.10, 2.11 and 2.12 show the data derived from the CSIR (2005c) study in Chameis Bay, categorised by depth strata, for the turbidity, dissolved oxygen and temperature profiles shown in Figure 2.9.

In general, turbidity increased from the top to the bottom strata (Fig. 2.9). ANOVAs showed that both depth ($p < 0.0001$ on all days) and treatment ($p < 0.05$ on all days) and their interaction ($p < 0.009$ on all days except day 4) had significant effects. Tukey post-hoc tests summarized in Fig. 2.10 revealed that control profiles had values that were significantly less than those in the mined profiles ($p < 0.05$), particularly in the middle and bottom strata (Fig. 2.10). Impacted profiles were nearly always either statistically indistinguishable from the controls ($p > 0.05$) or intermediate in value between the control and mined profiles: only in three out of the 24 cases tested was this not true.

Dissolved oxygen declined with depth on four of the eight days (Fig. 2.11) which coincided with days on which there were strong temperature gradients (Fig. 2.12). On the first four days control profiles generally had significantly higher oxygen concentrations than mined profiles, with impacted profiles being intermediate, but this pattern broke down on the last four days (see summary statistics in Fig. 2.11). Thus, although there was a strong suggestion that mining diminished oxygen levels, the evidence for this was confined to the first four days. There was a negative power relationship between oxygen and turbidity ($y=4.844x^{-0.1274}$; $r^2=0.198$; $p<0.001$).

Temperature declined with depth on all days. The magnitude of the decline ranged from 0.2°C on day five to 7°C on day seven (Figs 2.9 and 2.12). There were no compelling differences between the three treatments at any of the depths, with Tukey post-hoc tests showing either insignificant differences among treatments, or an absence of any pattern among treatments that were different (see statistics in Fig. 2.12).

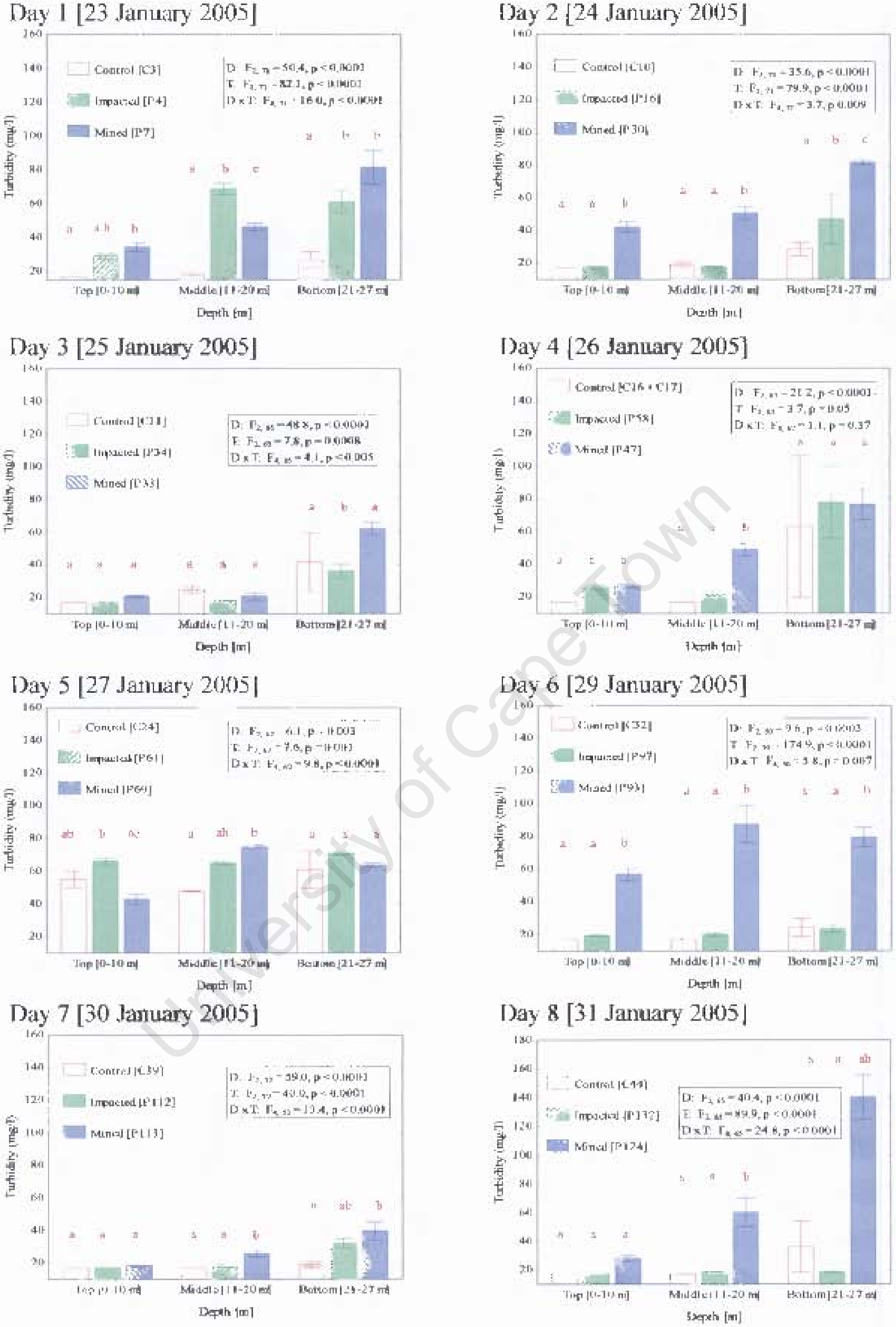


Figure 2.10: The depth-related turbidity comparisons for the 8-day CSIR study. Data are means ± 0.95 SE. Boxes summarize ANOVA analyses, and shared letters above bars show insignificant differences among treatments (Tukey HSD post-hoc; $p = 0.05$). D = Depth and T = Treatment.

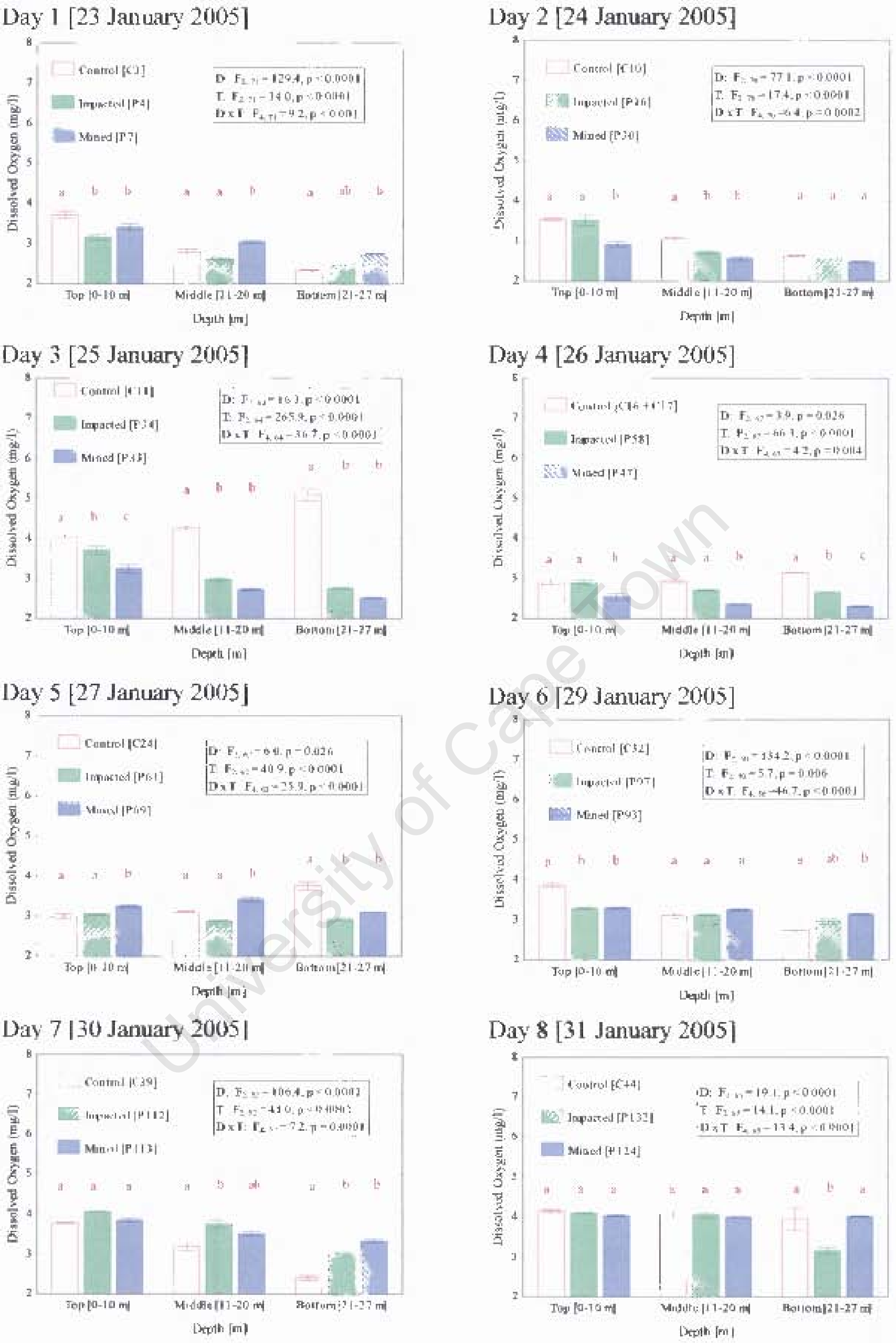


Figure 2.11: The depth-related dissolved oxygen comparisons for the 8-day CSIR study. Boxes summarize ANOVA analyses, and shared letters above bars show insignificant differences among treatments (Tukey HSD post-hoc; p – 0.05). D – Depth and T – Treatment.

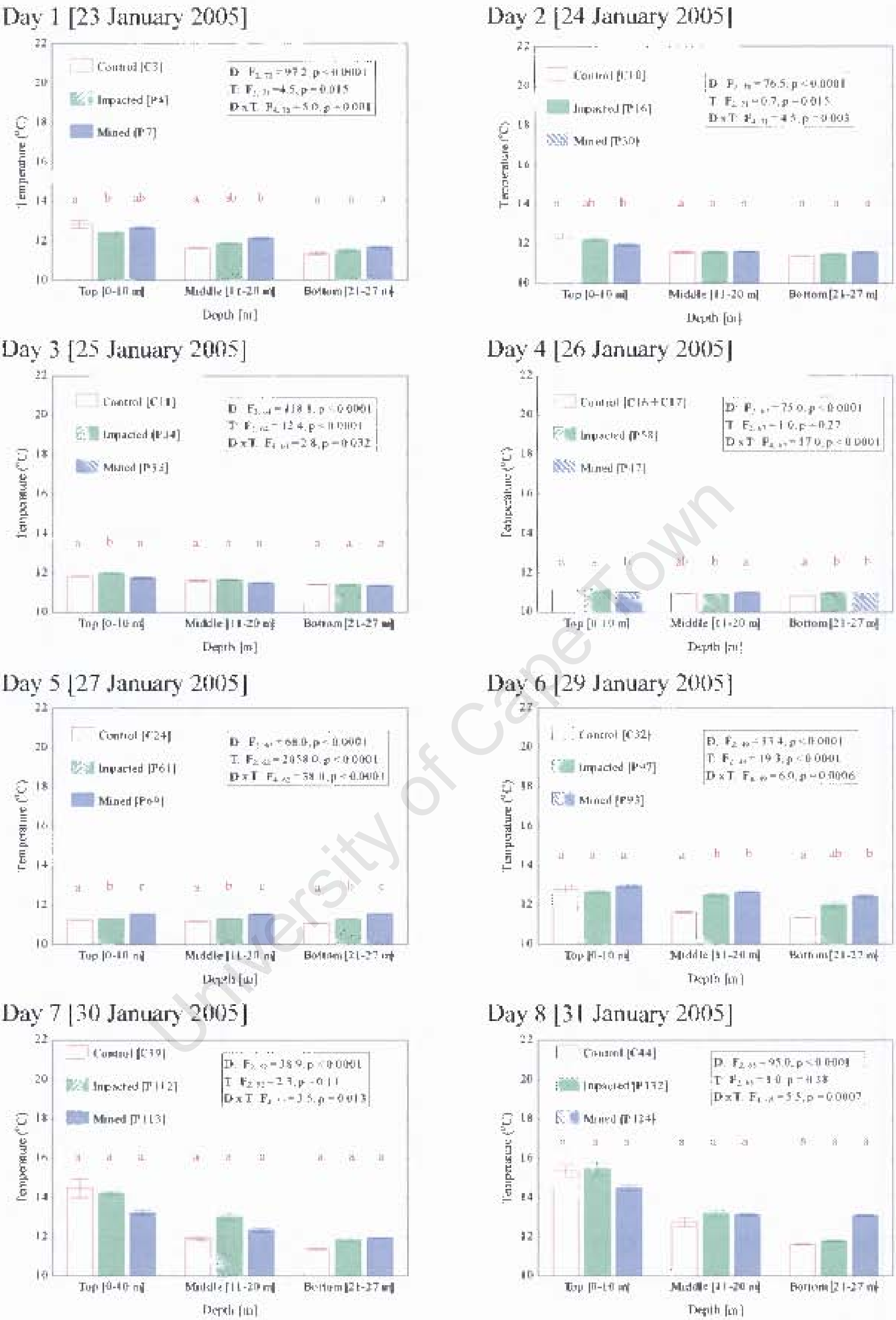


Figure 2.12: The depth-related temperature comparisons for the 8-day CSIR study. Boxes summarize ANOVA analyses, and shared letters above bars show insignificant differences among treatments (Tukey HSD post-hoc; p = 0.05). D = Depth and T = Treatment.

Discussion

Natural variability of temperature and dissolved oxygen (FIMS Data)

The low bottom temperatures prevalent during September 2005 (Figs 2.6 and 2.7) suggested that a major upwelling event was responsible for the anoxic conditions observed during Spring 2005. Dissolved oxygen generally decreased with an increase in depth (Figs 2.5 and 2.8), so upwelling of water from depths well beyond the photic zone (150 m) could have brought anoxic or hypoxic water to the offshore (70-100 m), inshore (40-70 m) and even nearshore (15-40 m) regions of the coast (Schmidt-Nielsen 1997, Chang 2003, Pulfrich *et al.* 2006).

Low-oxygen events are a common feature of the northern Benguela (Bailey *et al.* 1985, Shannon & O'Toole 1999, Fernández-Tejedor *et al.* 2004) and can be spatially and temporally variable depending on the nature of the event. In the FIMS report (Pulfrich *et al.* 2006) one such hypoxic event was recorded. During this event oxygen-poor conditions (of >0.2 mg O_2/ℓ) developed at an area just south of Kerbe Huk (Zone 15 in Fig. 2.1) at a depth of roughly 50 m during March 2005, and then propagated northward over a period of seven months to the southern extent of Chameis Bay (September 2005). The spatial contexts of the event covered an area from the shallow coastal zone outwards to seabed depths of >70 m. Interestingly, unlike the seven-month onset of the hypoxic event, the disappearance of hypoxic conditions took place over a period of three months, so that the system had completely returned to normal by the time the January 2006 survey was conducted (Pulfrich *et al.* 2006).

The observed trend that dissolved oxygen in surface water increased as one moved further away from the coastline (Fig. 2.8) is expected. This is because photosynthetic rates are higher in calmer water (away from the coastline) than they are in the turbid regions of surface water nearer to the coast – i.e. photosynthesis is limited as light is attenuated by the turbid conditions nearshore (Schmidt-Nielsen 1997). Thus, the high dissolved oxygen readings in the offshore surface waters may have been the result of greater primary productivity there.

Dredge-mining Induced Variability

The highly variable nature of the dissolved oxygen profiles from the mined areas, in comparison with those from the FIMS (Fig. 2.8), suggests processes that may be linked to the discharge of sediment (overspill) back into the sea, as well as sediment suspension resulting from the dredging process. Assessed in conjunction with the dissolved oxygen data presented in Fig. 2.9 this notion becomes more compelling. A plausible explanation for the trend observed in Fig. 2.9 (i.e. that the control profiles tend to have higher oxygen values than the mined profiles) lies in the nature of the mining operations.

During the dredging process sediment is sucked up – along with water – from the target mining area and, while the heavier sediment settles in the hopper, the excess water mixed with fine sediment flows back into the ocean as overspill via the overflow sluices of the hopper. The water being discharged at the sea surface as overspill has been extracted from the bottom and is likely to have chemically different and thermally cooler properties than the surface water, and thus influence the surface water. In the case of the mining panels at Chameis Bay, the bottom depths were not substantial, so this effect would have been less than if mining had taken place at deeper regions of the ocean.

There was no evidence that mining decreased the temperature of the surface layers by importing cool bottom water. Oxygen was depleted in the surface waters in 68% of the mined profiles, and oxygen levels were negatively correlated with turbidity, which was increased in all depth strata by mining. This negative correlation may have had three possible causes. Firstly, there are natural declines in oxygen and increases in turbidity with depth. The correlation may in part simply be due to their joint correlations with depth. Secondly, the introduction of sediment in the surfaces waters could have constituted a nutrient (or an additional microbial) input, which in turn might have increased respiration rates, thus depleting the dissolved oxygen in the surface waters. Thirdly, turbidity could have reduced light penetration, possibly limiting photosynthesis while respiration rates remain constant, ultimately leading to an imbalance in the favour of respiration. However, the probability of reduced oxygen levels having been brought on as a result of the second and third causes is very low, since readings for mined profile sites were taken during the dredging process (CSIR 2005b), which is too brief for such drastic biologically-induced changes to take place.

Moving on to the turbidity profiles, it is clear that these profiles yielded the most dramatic evidence of mining impacts. Considering the control, mined and impacted profiles, it is evident that natural turbidity profiles within the areas assessed varied from <17 to 100 mg/ℓ throughout the depth profiles, and that in the uppermost stratum turbidity only exceeded 40 mg/ℓ 13% of the time. During dredge-mining periods this frequency increased to 38% - which is an almost 3-fold increase. For the mid-water depth there was a 6.8-fold increase in the frequency of values exceeding 40 mg/ℓ during mining. This cut-off point (40 mg/ℓ), although having no significance to critical or lethal tolerance levels, is a good indicator since it is twice the natural turbidity level recorded for the Chameis Bay area (Pulfrich *et al.* 2006).

In the bottom-most strata of the water column, the impacts of mining on turbidity were indistinguishable because even in control profiles there were abrupt increases in turbidity caused either by the natural suspension of sediment or the disturbance of the seabed sediment by the equipment rig.

Dredge-mining clearly increases turbidity – elevating it to as high as 200 mg/ℓ. However, this needs to be put into context. Sediment discharged by the Orange River, storms and sand blown by Berg winds naturally increase turbidity – with values of up to 7400 mg/ℓ having been recorded (Zoutendyk 1995). Moreover, benthic species that live in the Benguela are likely to be tolerant of turbidity because it is a regular feature of the system. Nevertheless, particular stages of lifecycles may be vulnerable to turbidity, as Herrkind *et al.* (1988) demonstrated for the settlement of pueruli of the spiny lobster *Panulirus argus*.

Conclusions

The CSIR and FIMS data reviewed clearly indicate that dredge-mining activities induced substantial turbidity in the water column, especially in the mid-water to surface layers. It is arguable that inter-related processes between turbidity and other physical factors such as dissolved oxygen do exist, although the nature of these relationships still need to be defined. It is against this background of information that the potential impacts of dredge-mining on rock lobsters need to be assessed.

In the following chapters these issues are evaluated with the aid of laboratory experiments on the tolerance and reactions of juvenile rock lobsters to variations in hydrogen sulphide, dissolved oxygen and turbidity. It remains unknown whether dredge-mining influences hydrogen sulphide since no *in-situ* measurements of this variable were obtained during dredging trials. However, the ranges of dissolved oxygen and turbidity likely to be associated with mining and natural 'control' conditions do provide a yardstick for evaluating their effects on rock lobsters.

CHAPTER 3: EFFECTS OF DISSOLVED GASES ON JUVENILE ROCK LOBSTERS

Introduction

The first and, probably most, significant concern related to the proposed diamond dredge-mining activity along the southern coast of Namibia relates to the indirect effect of dredging on dissolved gases in the dredged and nearby areas. Two gases are of particular importance: dissolved oxygen and hydrogen sulphide. Scenarios as to how dredging may affect gas profiles are discussed in the following two sections.

Dissolved Oxygen Scenarios

Diamond dredge-mining in coastal waters off the west coast of southern Africa may indirectly indirectly perturb dissolved oxygen levels through the suspension of sediment during the dredge mining thinning processes. One reason for this is that elevated sediment suspension from the dredging process will attenuate light and in turn limit photosynthesis (Carter 1995). In addition, the suspension of organically rich sediment could constitute a nutrient input into the water column and boost microbial respiration rates and, in so doing, reduce dissolved oxygen concentrations in the water column (Van der Lingen *et al.* 2006).

If these conditions (i.e. low light and high nutrient levels) persist for even relatively short periods of time (e.g. 24 to 48 hours) respiration rates will quickly surpass photosynthetic rates (Schmidt-Nielsen 1997). This may ultimately result in the demise of large populations of primary producers, since the available oxygen is consumed at a rate faster than it is being generated within the ecosystem. The death of these photosynthetic organisms will add to the nutrient supply to the benthos, which will further boost respiration rates as decomposition of the dead organisms ensues (Bailey *et al.* 1985). The combined effect of elevated respiration, compounded by slow current regimes, will lead to the rapid reduction of dissolved oxygen near the seabed, resulting in severely hypoxic to completely anoxic conditions in relatively short periods of time (Bailey *et al.* 1985, Schmidt-Nielsen 1997).

Hydrogen Sulphide Scenarios

In the northern Benguela hydrogen sulphide is associated with the mud-beds found on the continental shelf along the coast (Bailey *et al.* 1985, Lutjeharms & Meeuwis 1987, Emeis *et al.* 2004). According to Emeis *et al.* (2004) hydrogen sulphide is a permanent feature of the northern Benguela, being evident throughout the year, and thus constitutes an important part of the natural state of the ecosystem. The sediment in the region is at times so saturated with hydrogen sulphide that the release of gas from the sediment ensues naturally (Emeis *et al.* 2004, Weeks *et al.* 2004). This implies that any form of sediment agitation, as in the case of dredging, may liberate hydrogen sulphide gas into the surrounding water – with potential negative impacts on the faunal communities in the region, as well as implications for the natural balance of gas release in the system (Garzoli & Gordon 1996, Boyer *et al.* 2000).

Motivation for and relevance of the study

Severe seasonal and sporadic reductions in dissolved oxygen concentrations (and production of hydrogen sulphide) over the continental shelf have marked influences on the distribution, and sometimes overall biomass, of marine species ranging from pelagic to benthic habitats (Bailey *et al.* 1985, Cockcroft 2001, Van der Lingen *et al.* 2006). This, in turn, has direct negative implications for fisheries that rely heavily on certain species such as the rock lobster species, *Jasus lalandii*, which is an important marine resource for both Namibia and South Africa (Grobler & Noli-Peard 1997, Cockcroft 2001, Van der Lingen *et al.* 2006). With the overlap of marine diamond-mining operations and the natural distribution of *J. lalandii* along the Namibian south coast this study was proposed to examine possible impacts of gaseous perturbations on the behaviour of juvenile rock lobsters.

Although physiological responses (such as heart rate, metabolic rate and hormonal adjustments) could have been assessed, I elected to monitor behavioural responses because they can easily be observed in the field whereas physiological responses are difficult to monitor under field conditions. In line with this motivation, behavioural responses of juvenile rock lobsters to reduced dissolved oxygen (hypoxia) and elevated hydrogen sulphide levels were assessed under laboratory conditions. Behavioural responses that were monitored included general movement, escape responses and incapacitation. The study was designed to test the null hypothesis of no difference in behaviour of juvenile rock lobsters in responses to induced hypoxia and increased hydrogen sulphide between sexes or between sites. The study was also aimed at documenting the behavioural progression of juvenile lobsters to incremental levels of hydrogen sulphide.

Materials and Methods

Comparative studies between localities and sexes

The experimental component of the study was conducted at the Marine and Coastal Management Research Aquarium located in Cape Town, South Africa. The experimental phase encompassed a period of roughly 16 months and was entirely laboratory based. Lobsters were collected from both Lüderitz and Cape Town to establish if responses were consistent between regions, and both males and females were used to test for any sexual differences.

Lobster acquisition from Lüderitz, Namibia

One hundred and fifty juvenile lobsters (75 males and 75 females) of carapace lengths between 50 and 60 mm were transported from Lüderitz to Cape Town via road. The lobsters were caught by way of overnight baited traps deployed from the *M.F.V. Snowgoose*, a 20-m rock-lobster fishing vessel. Lobsters were kept alive onboard in 100-ℓ bins filled with seawater that was frequently replenished during the two-day catching period. Once on land, the lobsters were immediately transferred to a walk-in chiller at the Ministry of Fisheries and Marine Resources (MFMR) in Lüderitz where they were kept at -5°C for 20 minutes while being packaged for transportation.

The lobsters were packaged in groups of 15 placed on top of slightly wet wood-wool inside 20-ℓ polystyrene containers with two frozen gel packs (Fig. 3.1 A). The wood-wool kept the air inside the container moist and maintained the position of the lobsters within the container during transportation. The gel packs maintained a cool air temperature inside the container. After the lobsters had been arranged in the container another layer of wood-wool was placed over them and the containers closed with polystyrene lids (Fig. 3.1 B & C). The containers were then transported by road to Cape Town. During the 16-hour journey, additional seawater was intermittently sprinkled over the top layer of wood-wool to maintain the moist conditions inside the containers.

At the Marine and Coastal Management Research Aquarium in Sea Point, South Africa, the lobsters were immediately transferred to holding tanks. All 150 lobsters survived the transportation and appeared to exhibit normal behaviour within 15 to 20 minutes of being transferred to the holding tanks. Similar numbers of rock lobster were collected at Cape Town and stored in identical holding tanks for a comparable period of time. Experiments were carried out nine days after the Lüderitz lobsters were acquired, thus allowing them to acclimate to the laboratory conditions.

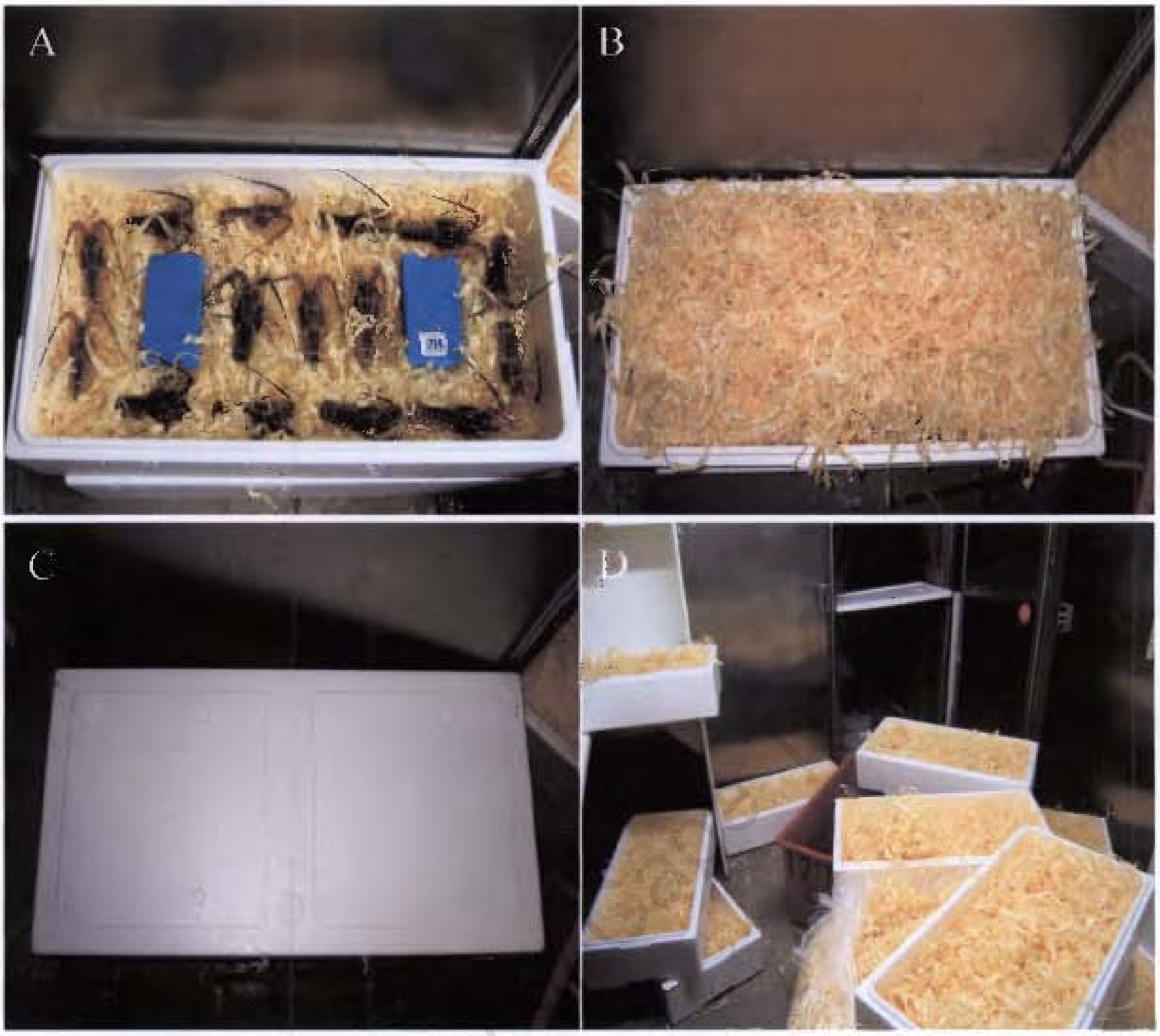


Figure 3.1: The packaging sequence for transport of Lüderitz lobsters. (A) Lobsters packed in wood-wool; Blue plastic bottles are the gel packs used to keep air temperature inside the polystyrene containers low. (B) Lobsters covered with wood-wool. (C) Cooler boxes sealed. (D) Polystyrene containers in the walk-in chiller at MFMR, Lüderitz, together with one of the 100-ℓ equipment bins (red) used to keep the lobsters alive onboard the fishing vessel.

Experimental Setup

To manipulate gas levels, the Cochran & Babcock (1974) setup, as modified by Beyers & Wilke (1990), was employed to regulate dissolved gases and temperature conditions in small aquaria with a closed water circulation (see Beyers & Wilke 1990). For all the gas experiments, aquaria made from 5-mm thick glass equipped with glass lids were used. Six aquaria were employed, each with a volume of 27 ℓ (30 x 30 x 30 cm), and were separated into two sections by 6-mm

PVC sheets with 15-mm diameter holes drilled in them (Fig. 3.2). The separators served to prevent interactions between adjacent lobsters during experiments while not hampering the flow of water through each aquarium. All aquaria were equipped with plastic valves to regulate the flow of water (Fig. 3.3 A).

Figure 3.2: Gas experimental setup during the initial stages of construction. The six short-term aquaria are visible on the right hand side while the aquaria used in the turbidity experiments described in Chapter 4 are in the centre. The four Perspex gas-regulation cylinders are mounted on the wall. All the short-term tanks already have the regulation valves attached and the first two short-term tanks are connected to the nitrogen-stocking cylinder on the right. The short-term tanks already have single grey PVC separators installed.

Four Perspex gas-stocking cylinders were mounted to the wall of the laboratory with aluminium brackets and self-tapping screws (Fig. 3.2). Two of the four stocking cylinders were used to regulate nitrogen supply, and one each for the supply of hydrogen sulphide and oxygen. All four stocking cylinders were fitted with 10-cm thick non-toxic rubber bungs on both ends – with each bung having three holes: two 15-mm holes on the outer sides for water flow and one central 5-mm hole for gas input (in the case of the bottom rubber bungs) or pressure release (in the case of the top rubber bungs). The four stocking cylinders were then connected to the relevant gas cylinders with the aid of 5-mm diameter rubber tubing. Each stocking cylinder was equipped with a regulator valve at the top of the cylinder to control the inflow of water from the reservoirs and the water level inside the cylinder. Regulation of water level had significant implications for the efficiency of gas stocking.

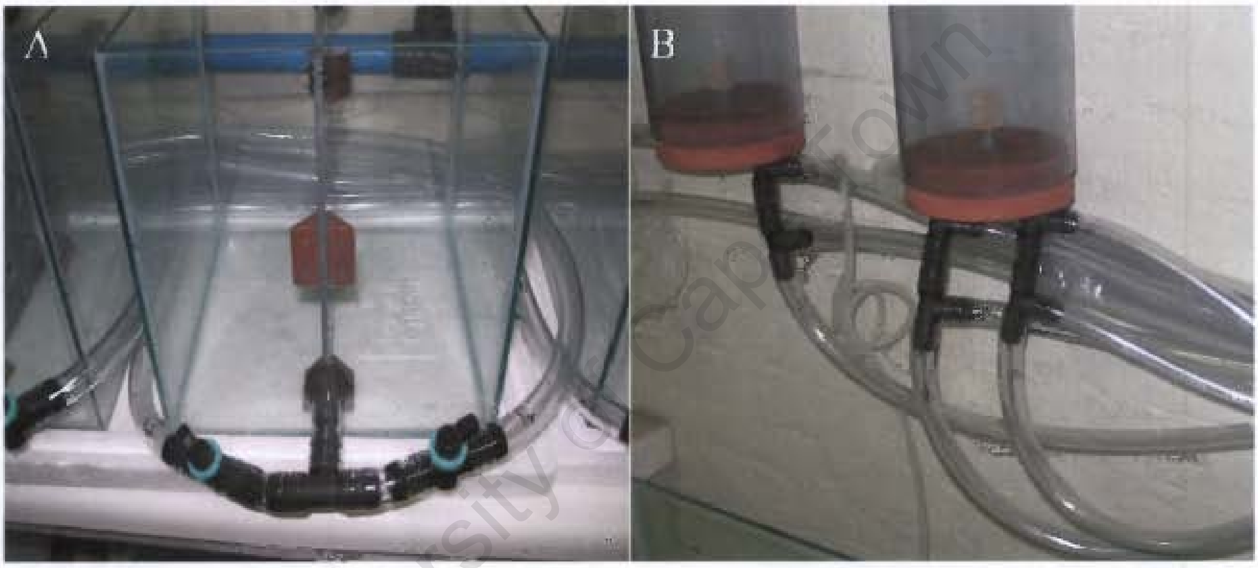


Figure 3.3: Two components of the gas experimental setup. (A) Short-term tanks with one inlet and two plastic regulation valves, and the single grey PVC separator held in place by two brown plastic spacers on opposite ends of the separator. (B) Two of the four stocking cylinders, the nitrogen cylinder on the right and the oxygen cylinder on the left, together with the wooden air stones and outlets, which had three connections per outlet.

Each of the stocking cylinders had a wooden airstone at the bottom of the cylinder which was crucial for the effective stocking of the relevant gases. Wooden airstones (Fig. 3.3 B) were found to be most effective in creating the smallest bubbles thus increasing the diffusion of gas in water – a factor related to the high surface-area-to-volume ratio of the bubbles created. This was very important in that it minimized the amount of gas used during gas-stocking phases.

Each stocking cylinder had three connections to each outlet at the bottom so as to supply one sets of three aquaria per outlet (Fig. 3.3 B), and a single inlet at the top of the cylinder to complete the circuit. The circulation of oxygen and hydrogen sulphide was kept separate to limit oxidation of the hydrogen sulphide. Two 100-ℓ fiberglass reservoirs, with removable polystyrene lids, were used to keep the two gas-stocked water-types separate (see Fig. 3.4). Two submersible, centrifugal water pumps were used to circulate the water through the setup and a Hailea flow-through water chiller held the water temperature at about 17°C. The gas experimentation tanks each had a single inlet and outlet (Fig. 3.3 A). However, each inlet had two source inlets from the relevant gas-stocking cylinders – depending on the type of experiment (i.e. sulphide or oxygen experiments) the tank was intended to be used for (Fig. 3.3 A, Fig. 3.4).

Depending on the treatment, the two source inlets were regulated to create the required conditions inside the aquarium (Fig. 3.3 A). In the case of the hydrogen sulphide experiments each aquarium had a source inlet from the hydrogen sulphide stocking cylinder as well as a source inlet from the nitrogen-stocking cylinder. In the case of the oxygen experiments, each aquarium had a source inlet from the oxygen stocking cylinder and one from the nitrogen-stocking cylinder. Nitrogen was used to displace oxygen, inducing varying levels of hypoxia in the treated water. By regulating the rates of flow from the two sources, the concentration of hydrogen sulphide or oxygen could be controlled inside the aquaria.

Water was treated or stocked by initially filling the appropriate stocking cylinders with normal seawater to a level roughly 10 cm below the top rubber bungs and then bubbling the relevant gases through the water from pressurized gas cylinders. The water level inside the stocking cylinders was monitored and regulated, since this had direct implications on the pressure at which the gases could be bubbled through the water column and the efficacy of gas-stocking.

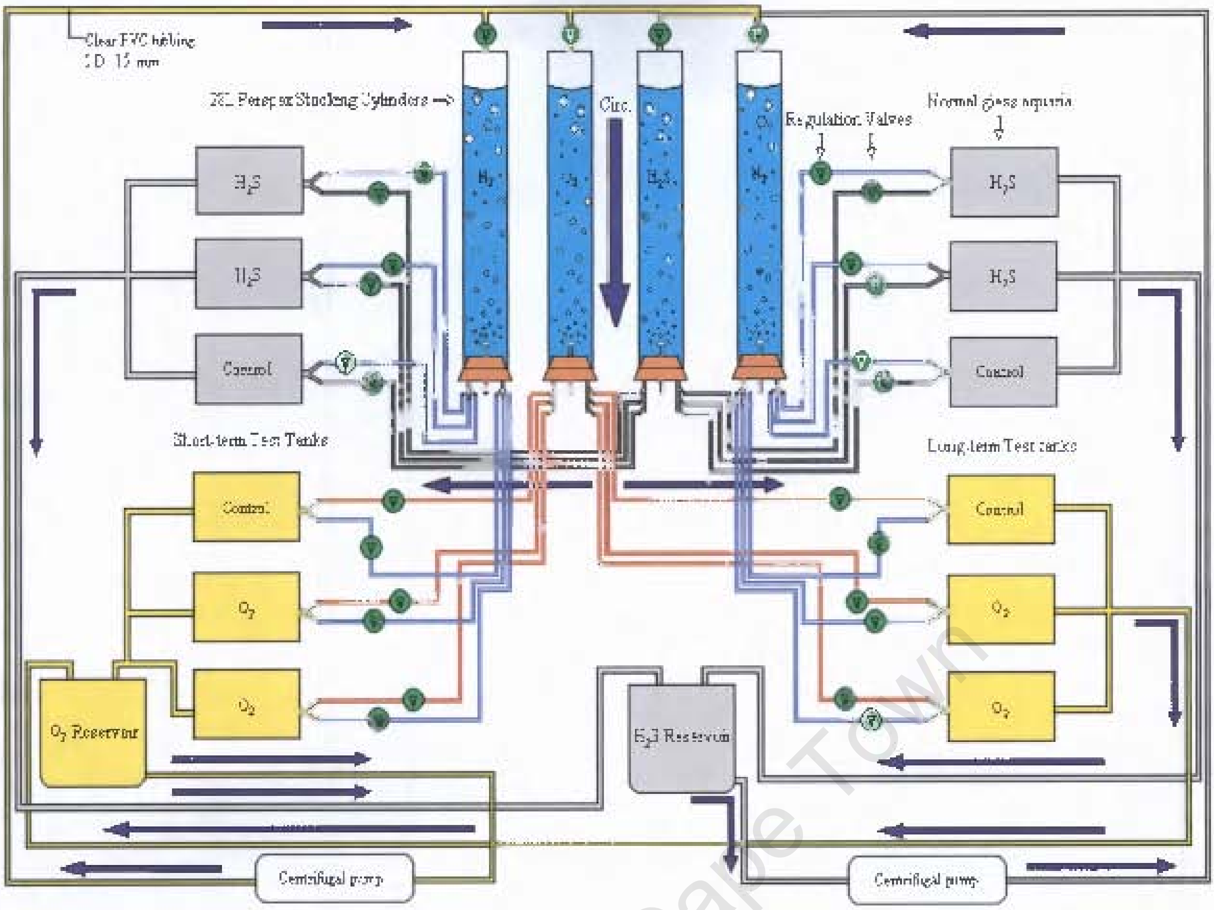


Figure 3.4: Diagrammatic representation of the concept and design of the setup employed for the laboratory gas experiments. The water chiller and rubber bungs at the top of each stocking cylinder have been omitted from the diagram.

For the hydrogen sulphide experiments a 98% grade, liquefied, hydrogen sulphide gas was used while technical oxygen was used for all the oxygen experiments. All experiments were initiated in the following manner: (1) filling the reservoirs with normal seawater; (2) filling the stocking cylinders with seawater from the reservoirs; (3) running the circulation for a few minutes to establish baseline conditions in the experimentation aquaria; (4) stocking the water in the stocking cylinders with the relevant gases; and (5) selecting and allocating lobsters to the experimentation aquaria. The next step (6) constituted the actual experimentation phase as described in the following section.

Experimental Phases

Short-term experiments comparing moribundity

For the 'short-term' experiments comparing moribundity of lobsters from Lüderitz and Cape Town, both hydrogen sulphide and oxygen deficiency were examined over a timescale of 60 minutes. Hydrogen sulphide and oxygen experiments were conducted separately on consecutive days, as it was impractical to run these tests simultaneously.

For the hydrogen sulphide moribundity experiments, male and female lobsters (all in the inter-moult phase) from both Lüderitz and Cape Town were selected and allocated in series of pair-wise combinations to each of the three hydrogen sulphide experimentation tanks. Lobsters were then left to adjust to the experimentally controlled temperature of 17°C for 15 minutes before the experimental phase commenced. After the 15-minute acclimation period the hydrogen sulphide valve was opened for roughly seven seconds to allow hydrogen sulphide-saturated water to flow into the test tank and induce the first experimental stress level. Incremental stress levels (i.e. concentrations of hydrogen sulphide) were then induced at 10-minute time-intervals until the experiments were concluded. Pilchard heads were added to the test tanks after the first stress level was induced and any feeding behaviour was recorded throughout the experiment.

At each stress level, a fright stimulus was induced with the aid of a plastic probe and the responses of the lobsters recorded. The fright stimulus entailed lightly brushing the fleshy underside of the lobster's telson against the side of the aquarium wall with the probe. This always induced a tail-flip flight response from the lobsters, which was considered as a positive escape reaction to the stimulus. Lobsters were declared moribund when no flight response was elicited. 'Moribundity' was defined as the condition in which lobsters were no longer capable of responding to the fright stimuli. Although still alive (and capable of recovering if the stress was removed in time), lobsters that are moribund are incapable of moving away from stress and thus prone to death through continued exposure to the stressful conditions. All lobsters were used only once during the hydrogen sulphide experiments. Control animals constituted those held in tanks that were not exposed to hydrogen sulphide.

The hypoxia experiments were conducted in a similar manner as the hydrogen sulphide experiments except for minor differences in methodology. Water from the oxygen-stocked cylinder was mixed with that from the nitrogen-stocking cylinder to achieve reduced concentrations (levels) of dissolved oxygen, with the 10-minute irregular reduction in dissolve oxygen being monitored with an oxygen probe. Whenever the oxygen experiments were conducted on a day following the hydrogen sulphide experiments, an initial purging procedure was required to clean the plastic tubes of all hydrogen sulphide that might have been left over from the previous day's experiments. Purging was achieved by filling the oxygen reservoir with fresh seawater and maintaining the circulation (while adding fresh water to the reservoir) until no traces of hydrogen sulphide could be detected – assessed on the basis of pH readings, colour and smell of the circulating water. The other difference was that the fright stimulus used in the hydrogen sulphide experiments proved ineffective as a means of monitoring responses to oxygen depletion because even when dissolved oxygen levels fell below minimum values that the sensor could detect, the lobsters still gave positive flight responses. Hence, a more subtle method had to be devised to monitor responses.

Lobsters were observed to adopt a 'bracing' reaction (in preparation to flight) when a hand was waved above the test tank. This bracing action included moving the antenna toward the waving hand and holding them firm in the direction of the hand, raising the torso off the bottom of the tank, and temporary cessation of scaphognathite (bailer) movements pumping water over the gills (McMahon & Wilkens 1975). Although subtle, these responses were observed every time the hand-waving stimulus was employed, and since the intensity of these responses diminished with increasing hypoxia, they were considered an effective response cue. Lobsters were considered 'non-responsive' when none of the responses described above was induced by the hand-waving stimulus. It should be noted that the 'non-responsive' state referred to here is not the same as moribundity, as the 'non-responsive' state is reached before the severely incapacitating state induced by the exposure to high levels of hydrogen sulphide.

Short-term behavioural experiments

Short-term behavioural experiments were conducted along the same lines as the short-term moribundity experiments with the exception that (1) males and females were pooled because no significant difference were evident between the sexes during the short-term moribundity experiments, (2) only hydrogen sulphide was used in the experiment, and (3) behaviour was monitored in addition to moribundity. No behavioural experiments were conducted for the short-term hypoxia experiments because the subtle responses from lobsters during the hypoxia experiments made it difficult to distinguish different levels of behaviour as hypoxia intensified.

For the short-term hydrogen sulphide behavioral experiments various types of behavioural responses were defined to monitor the reactions of the lobsters to increased hydrogen sulphide. These response were, in sequential order, related to an increase in hydrogen sulphide concentration: (1) No movement [defined as the cessation of locomotory activity following an active period]; (2) Prolonged movement [defined as excessive and/or locomotory activity for lengthy time periods following a non-active period]; (3) Tail flip [defined as the active backward swimming of lobsters by rapid and rhythmic contraction of the abdomen under the thorax of the lobster]; and (4) Moribundity [defined as the absence of flight responses normally exhibited by the lobster after fright stimulation as describe previously]. After each stress level was induced, the behaviour of the lobsters was observed and recorded. After the experiments were concluded all the lobsters were removed from the test tanks, placed in small perforated containers equipped with lids and returned to well-aerated holding tanks to observe recovery time.

Experimental measurements

A WTW Multi 350i multi-parameter sensor was used to assay seawater pH, temperature, salinity and dissolved oxygen. No affordable and effective real-time measuring device for hydrogen sulphide was available. The only method by which hydrogen sulphide could be determined was by means of titrations. However, this method was impractical given the time limitations of the experimental design and thus an alternative method of sulphide measurement had to be found.

However, in a study done by Vismann (1996), pH was correlated to total sulphide using a copper electrode for measuring sulphide concentrations. On the basis of this concept, pH was used as a proxy for hydrogen sulphide measurements in the current study. The WTW multi-parameter sensor was used to measure pH during the hydrogen sulphide experiments and the concentration of the total dissolved sulphides later calculated from the pH.

The correlation sourced from Vismann (1996) was verified and an independent correlation constructed to ensure the integrity of the measurements made in the laboratory. For this a sodium thiosulphate titration method for sulphide determination was sourced from Dr. D. Schiedek in Warnemuende, Germany, and used to develop the pH-total sulphide correlation (Fig. 3.5). The thiosulphate titration method works on the principle of reacting excess iodate and potassium iodide with hydrogen sulphide to form iodine and ions of the hydrogen sulphide compound. Diluted sulphuric acid is added to the solution to ensure that the necessary reactions have taken place. A standard starch solution is added to serve as an indicator. The solution is subsequently titrated with sodium thiosulphate until a brownish-yellow to milky-white colour change is observed. The volume of sodium thiosulphate used in the titration is then used to calculate the actual quantity of sulphide that was present in the water.

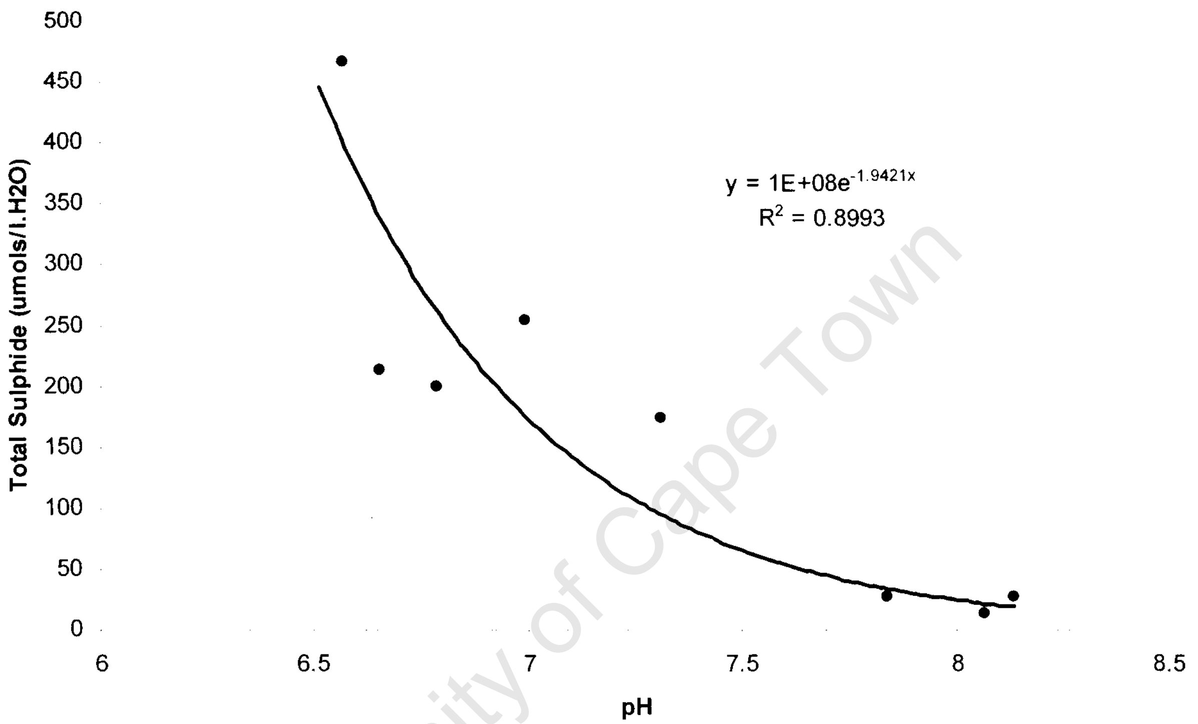


Figure 3.5: The correlation between pH and total sulphide (with data points included) derived from the sodium thiosulphate method. Titrations were carried out at a water temperature of 17°C. Total sulphide was measured in μmols sulphide/ℓ seawater.

During the independent correlation construction phase, levels of hydrogen sulphide were stepped up at 10-minute intervals by opening the regulation valves to the test tanks and the pH was monitored using the WTW multi-parameter sensor. Fixed volumes (30 mℓ) of water were then tapped off from the test tanks into Erlenmeyer flasks with glass stoppers, at each of the stress levels, and placed in a dark cupboard until the titrations were done. Three titrations were done on each sample and the means employed in the regression against pH. A pipetting factor of one was used in the hydrogen sulphide calculations since a 0.02 Normal solution of sodium thiosulphate was used in the titration.

Data analyses

Microsoft Excel (2002) and Statistica 7 (2004) software packages were used to construct graphs and carry out statistical analyses on the data. The assumptions for the statistical procedures employed were assessed with Levene's test for homogeneity, normal probability plots, together with Shapiro-Wilk tests and histograms. All the data (hydrogen sulphide, dissolved oxygen and behavioural response data) were found to be heteroscedastic and therefore required transformation to comply with the parametric ANOVA assumptions of normal distribution and variance equality.

The short-term hydrogen sulphide data were Log (Y+1) transformed and then analyzed in Statistica 7. Post-hoc (Tukey's honestly significant difference, HSD) assessments were carried out to further explore the results obtained from the parametric ANOVA tests. Non-parametric analyses were carried out on the short-term hypoxia survival data and the short-term behavioural responses data, since transformation did not remove heteroscedasticity from the two data sets. Independent (Kruskal & Wallis) ANOVA was done on the dissolved oxygen data, while dependent (Friedman) ANOVA was done on the behavioural data.

Results

Short-term moribundity experiments

Groups were compared with other groups on the basis of likeness, i.e. same sex comparisons between sites and different sex comparisons within sites. Of all the comparisons made for the hydrogen sulphide experiments, only one yielded a statistically significant difference (Figure 3.6 and Table 3.1). This was the hydrogen sulphide comparison between the Cape Town and Lüderitz females, with the latter having a significantly lower tolerance to hydrogen sulphide than the Cape Town females.

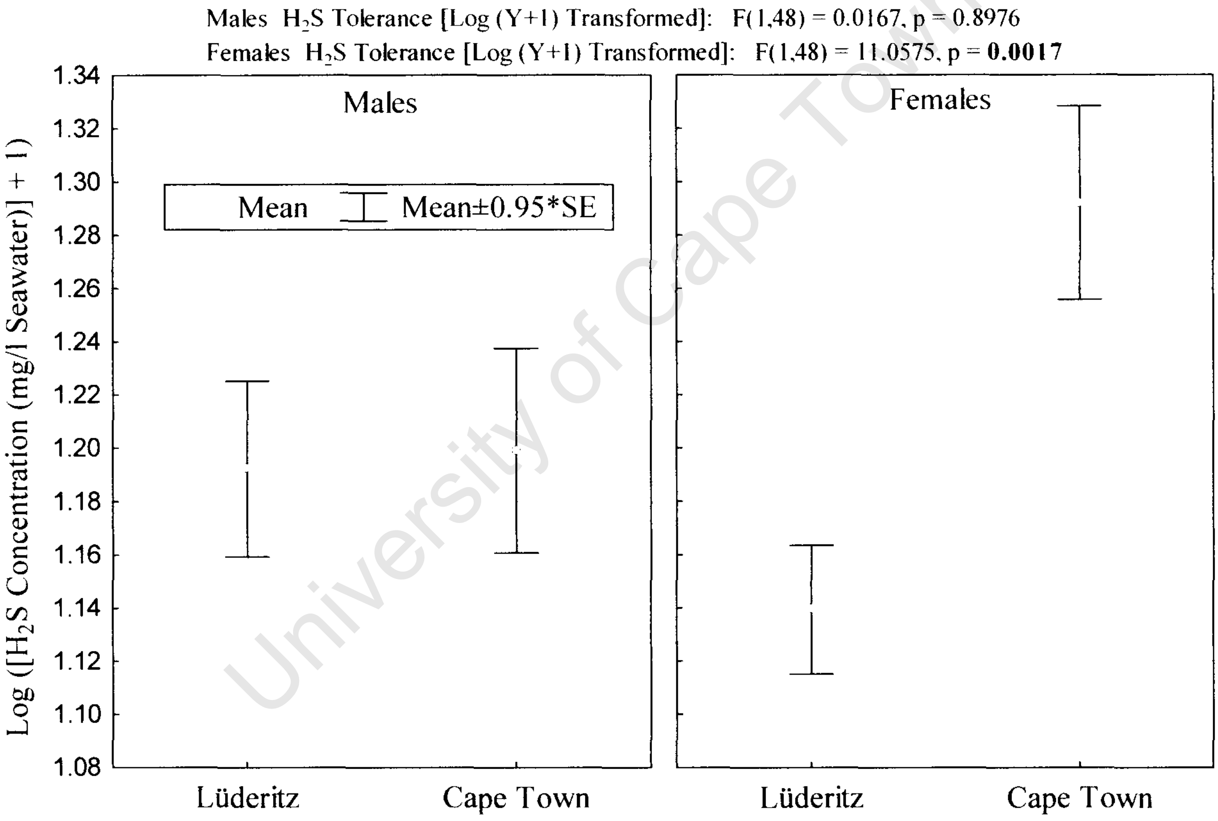


Figure 3.6: Hydrogen sulphide levels at which lobsters became moribund. Lobsters of each sex from Lüderitz, Namibia, and Cape Town, South Africa, are compared with each other.

With regards to the oxygen data, there were no significant differences in the hypoxia levels that induced 'non-responsiveness' between sites within sexes, or between sexes within sites (Figure 3.7, Table 3.2). The largest difference in the data was 0.011 mg O_2/ℓ in the comparison between the Lüderitz females versus Lüderitz males, and the smallest difference 0.004 mg O_2/ℓ in the comparison of Cape Town females versus Lüderitz females comparison. The narrow distribution of the data means between 0.55 and 0.57 mg O_2/ℓ seawater is evident in Figure 3.7.

Table 3.1: Post-hoc (Tukey's HSD) test results of the site and gender-specific comparisons for H_2S tolerance. Values in brackets are mean hydrogen sulphide concentrations at which moribundity occurred. Site abbreviations: LDZ = Lüderitz and CPT = Cape Town. [Between Group Mean Squares = 0.0309, Df = 96, N = 25, Alpha = 0.05; Y = Log (Y + 1) Transformed]. Significant p-values are presented in bold font.

Site [Sex] (Mean)	LDZ [M] (1.1922)	LDZ [F] (1.1393)	CPT [M] (1.1991)	CPT [F] (1.2920)
LDZ [M] (1.1922)	---	0.7214	0.9991	0.1927
LDZ [F] (1.1393)	0.7214	---	0.6270	**0.0146**
CPT [M] (1.1991)	0.9991	0.6270	---	0.2485
CPT [F] (1.2920)	0.1927	**0.0146**	0.2486	---

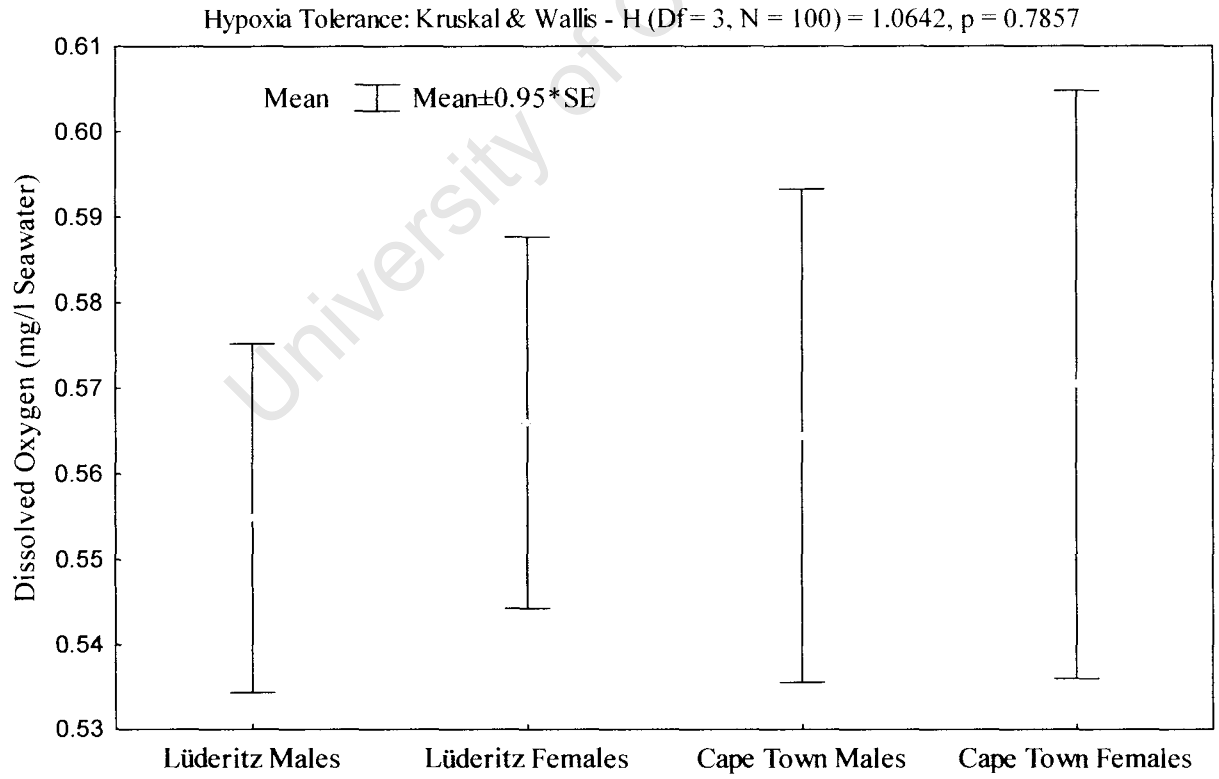


Figure 3.7: Comparisons of the level of oxygen at which 'non-responsiveness' became apparent in the male and female lobsters from Lüderitz and Cape Town.

Table 3.2: Non-parametric (multiple comparisons) test results of the site and gender-specific comparisons for hypoxia tolerance. Abbreviations: LDZ = Lüderitz and CPT = Cape Town. [Df = 3, N = 100, Alpha = 0.05].

Site [Sex] (Rank)	LDZ [M] (52.78)	LDZ [F] (53.64)	CPT [M] (49.52)	CPT [F] (46.06)
LDZ [M] (52.78)	---	0.1048	0.3973	0.8189
LDZ [F] (53.64)	0.1048	---	0.5021	0.9237
CPT [M] (49.52)	0.3973	0.5021	---	0.4216
CPT [F] (46.06)	0.8189	0.9237	0.4216	---

Short-term behavioural experiments

Figure 3.8 shows the typical rate of increase in hydrogen sulphide concentration during the short-term behavioural experiment, which ran for one hour (Fig. 3.8). As time on the *x*-axis was linked to the different levels of hydrogen sulphide, there were six 'stress levels' at which the responses of the lobsters could be recorded. The behavioural responses of lobsters were analyzed in terms of the lowest hydrogen sulphide concentrations at which the respective behavioural responses were first observed, and comparisons made between Lüderitz and Cape Town lobsters.

As hydrogen sulphide levels increased, the lobsters first ceased movement ('no movement'). This was then followed by an agitated and continuous movement ('prolonged movement'), an energetic escape response ('tail-flip') and finally incapacitation ('moribundity') (Fig. 3.9). Statistical analysis of the between-site comparisons revealed that there were significant differences between lobsters from the two different locations for the 'no movement', 'tail-flip' and 'moribund' responses (Table 3.3), with the Lüderitz lobsters generally showing negative impacts at lower concentrations of hydrogen sulphide than the Cape Town lobsters (Fig. 3.9).

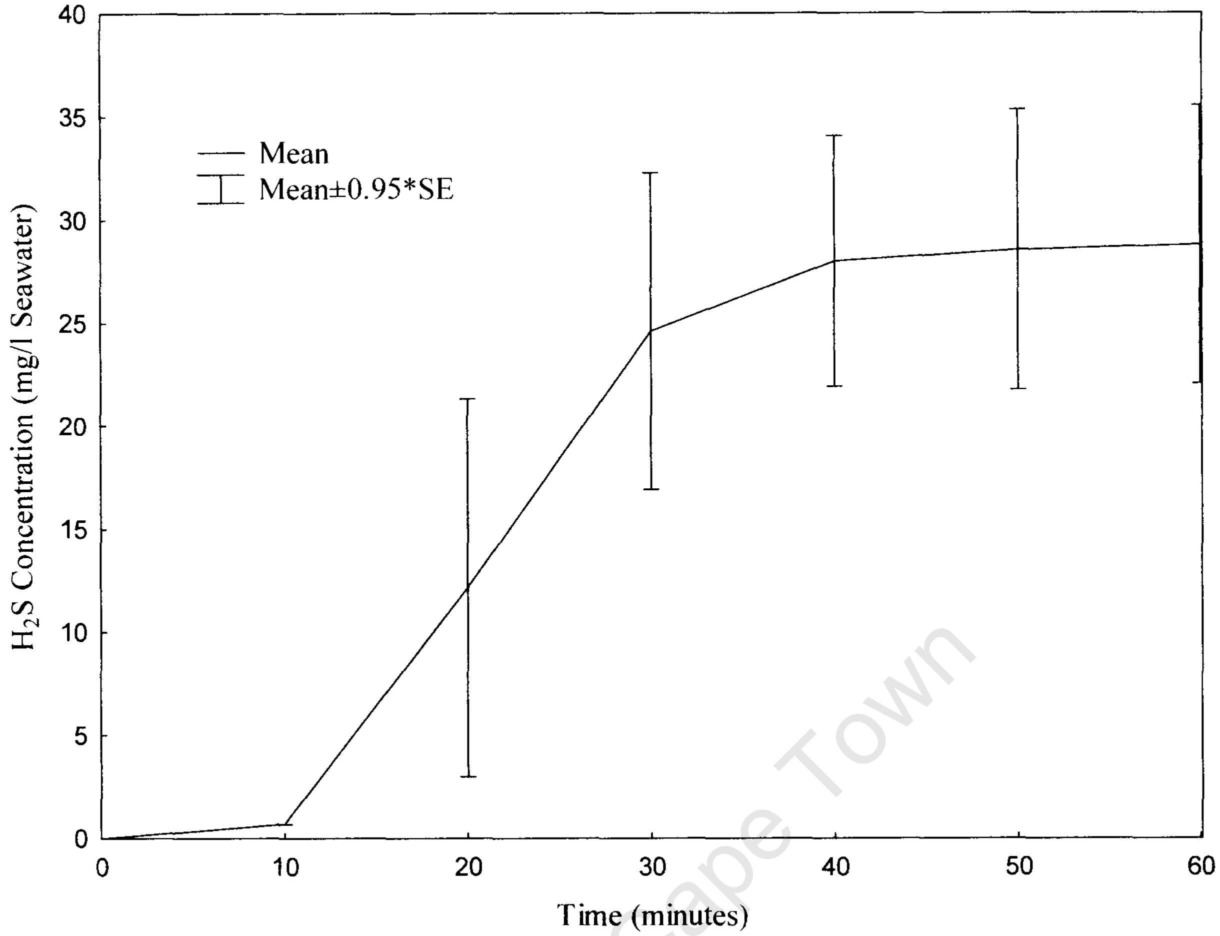


Figure 3.8: Graphic representation of the hydrogen sulphide concentration increases during the short-term behavioural experiments.

Lobsters generally exhibited continuous walking responses during the second and third stress levels, and showed rapid incapacitation after the third behavioural (tail-flip) response, which was usually observed during the fourth stress level. The short periods of tail-flip responses were always followed by periods of total inactivity that lasted until the experiments were ended. However, fright stimuli during this incapacitation period did yield flight responses up until the level where the lobsters were declared moribund. In some experimental runs, moribundity was observed only 30 minutes into the experiment, but it was generally observed 50 to 60 minutes into the experiment.

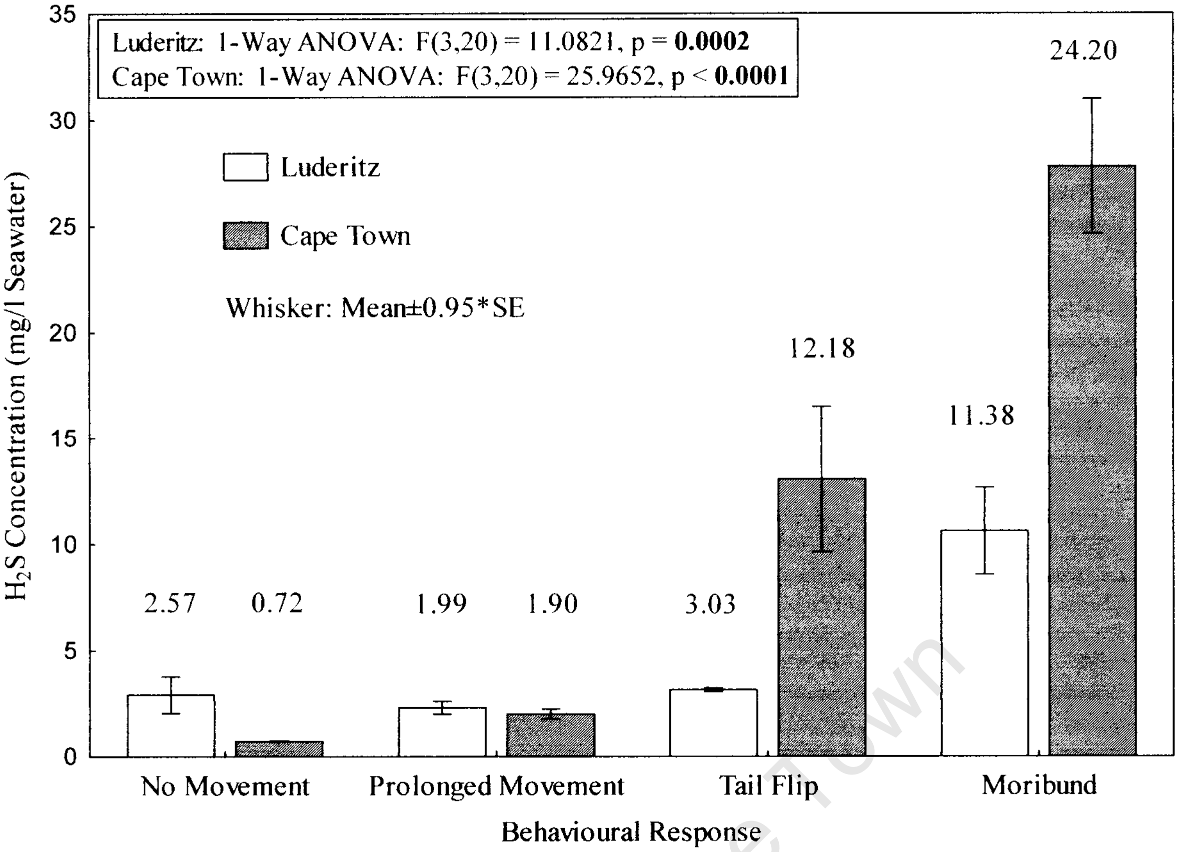


Figure 3.9: Comparative results for the short-term behavioural experiments. Clear bars represent Cape Town lobsters and shaded bars represent the Lüderitz lobsters. Means (mg/ℓ) for the site-specific behavioural responses are shown above the each bars.

Table 3.3: Non-parametric, between-site (Wilcoxon's Matched Pairs tests) comparisons of the levels of hydrogen sulphide at which different behavioural responses were initiated. Significant statistics are shown in bold font. Abbreviations: NM = No movement, PM = Prolonged movement, TF = Tail-flip and M = Moribundity. [Alpha = 0.05]

Site [Behav. Resp.] vs. Site [Behav. Resp.]	N	T	Z	p
Lüderitz [NM] vs. Cape Town [NM]	6	1.00	1.992	**0.0464**
Lüderitz [PM] vs. Cape Town [PM]	6	9.00	0.314	0.7532
Lüderitz [TF] vs. Cape Town [TF]	6	0.00	2.201	**0.0277**
Lüderitz [M] vs. Cape Town [M]	6	0.00	2.201	**0.0277**

After moribundity was declared lobsters were exposed to the existing hydrogen sulphide stress for 10 minutes and then individually placed in well oxygenated water. A 100% recovery was observed for these lobsters within a 14-hour period.

Discussion

Limitations

The Cochran & Babcock (1974) setup used for the experiment presented numerous complications. Although it provided a means of regulating gas experimentally in a closed system, the setup was cumbersome. The most important difficulty pertained to the pressurized gas system used to stock water with the required gases. This system was manually controlled and required constant adjustments of the rates of flow of gases to achieve a desired concentration. For example, with the hypoxia experiments, nitrogen gas was used to strip the oxygen from the water in the stocking cylinders. This meant that a particular pressure from the source gas cylinder needed to be maintained to bubble the gas through the water in the stocking cylinder at a constant rate. As a result the outflow from the gas cylinder needed to be continually monitored and regulated. This has obvious limitations for long-term experiments since the pressure from the gas cylinder can drop in less than fifteen minutes depending on the rate of gas flow. Even during the short-term experiments, the pressure regulation proved demanding. Improvement or automation of this process will be necessary for long-term use of the setup.

Short-term gaseous experiments

The object of comparing lobster populations from different areas was to determine whether lobsters from Lüderitz and Cape Town react similarly to increases in hydrogen sulphide or depletion of dissolved oxygen. The study was designed to be of relevance to the northern Benguela where marine diamond mining takes place, and it was thus of interest to know whether lobsters from the southern Benguela would respond similarly to those in the northern Benguela.

From the hydrogen sulphide and dissolved oxygen experiments, no significant differences emerged between the sexes at either location. Comparisons of the two different sites yielded only one significant difference: the Cape Town female lobsters became moribund at a significantly higher hydrogen sulphide concentration than the Lüderitz females (Fig. 3.6 & Table 3.1). This difference was unexpected, given the physico-chemical histories of the two geographical regions. Being exposed to hydrogen sulphide conditions more frequently in the northern Benguela the Lüderitz lobsters were expected to be more tolerant to hydrogen sulphide than southern Benguela (Cape Town) lobsters, based on the assumption that natural selection would have favoured varieties with greater tolerance in the northern Benguela (see Schmidt-Nielsen 1997). Further discussion of this subject is addressed in the 'Short-term behavioural experiment' section that follows.

Of interest, however, was the range of tolerance exhibited by the different groups. Moribundity among Lüderitz lobsters spanned values from 5.7 to 43.4 mg H_2S/ℓ seawater. For the Cape Town stock the value ranged from 7.4 to 62.5 mg H_2S/ℓ seawater. Thus, the significant difference in tolerance between females from the two regions needs to be viewed in the context of the wide variance in the data. The facts that (1) no differences in hydrogen sulphide tolerance between sexes within regions were evident; (2) males from the two regions had statistically indistinguishable responses; and (3) a wide variance existed in the data, all argue that the difference recorded between the females from the two regions should be interpreted with caution.

With regard to the oxygen deficiency, no significant differences were observed between the two sexes or between lobsters from Lüderitz and Cape Town. The mean levels of oxygen at which the lobsters became 'non-responsive' were all closely distributed around 0.57 mg O_2/ℓ seawater. Thus, the null hypotheses of no difference for inter-sex and site-related hypoxia tolerance were not rejected.

Short-term behavioural experiments

The main objective of the short-term gas experiments was to gauge the effects of increasingly stressful hydrogen sulphide conditions on the behaviour of lobsters. This approach helped to determine the levels at which lobsters first detected the stress, and the overall progression of behavioural responses to increasing levels of stress.

A clear sequence of behavioural responses emerged. The first noticeable response was the 'no movement' response, which was generally observed at the first stress level and signified the detection of the hydrogen sulphide by the lobsters. This response was often exhibited in two modes. In the one mode the lobsters temporarily ceased all motion upon sensing the hydrogen sulphide and then continued with locomotory activity after a short (~ 40-second) period of 'freezing'. During the 'freezing' periods even bailer movements were halted for brief periods of time (3 to 5 seconds). In the alternative mode the lobsters exhibited no locomotory activity after the initial freezing period but remained in their 'frozen' states until the next response was initiated. This can be likened to 'thermal stunning' usually observed when rapidly transferring exothermic animals between two thermally different bodies of water (Schmidt-Nielsen 1997). This response was generally observed from 1.3 to 1.6 mg H_2S/ℓ for Lüderitz lobsters and from 0.7 to 1.2 mg H_2S/ℓ for Cape Town lobsters.

The next behavioural response was 'prolonged movement', which was usually induced during the second stress level of the experiment. This response was characterized by rapid walking against the sides of the aquaria for prolonged periods of time (usually six to ten minutes uninterruptedly). Lobsters tended to walk away from the source of hydrogen sulphide and were thus found clambering against the glass in the corners at the far end the aquaria. They occasionally used the water circulation holes in the plastic separators to climb to the water surface, necessitating the use of glass lids to prevent them from climbing out of the aquaria. This response was observed over a range of 1.7 to 2.2 mg H_2S/ℓ for the Lüderitz lobsters and 1.3 to 2.5 mg H_2S/ℓ for the Cape Town lobsters.

The third behavioural response was the 'tail-flip' response, mostly associated with the third stress level. The 'tail-flip' response was considered to be a critical stress-level indicator as it indicated the need for a rapid and forceful escape. Tail flips demand substantial amounts of energy with the muscles in the tail tiring out quickly; thus the lobsters are only able to sustain brief bouts of this activity, as a last resort of escape (Heydorn 1969, Cockcroft 2001). Tail flips took place at concentrations of 3.9 to 5.1 mg H_2S/ℓ for the Lüderitz lobsters and 4.7 to 5.7 mg H_2S/ℓ for the Cape Town lobsters.

The 'tail-flip' response was always followed by a period of inactivity (i.e. partial incapacitation). However, the fright stimulus still yielded flight responses during this inactive period.

'Moribundity' was indicative of the fourth and final stress level of the experiments, when lobsters were completely incapacitated so that no flight response was possible. 'Moribundity' occurred at concentrations of 14.8 to 33.4 mg H_2S/ℓ for Lüderitz lobsters and 19.9 to 30.7 mg H_2S/ℓ for Cape Town lobsters. 'Moribundity' signified severe incapacitation, but not death.

The recovery of the lobsters after being exposed to debilitating concentrations of hydrogen sulphide constitutes a significant observation. Observations made during diving surveys immediately following sulphur eruptions along the Namibian coast have revealed that although most of the lobsters walk out onto the beach during these events, many remain on the reef (Kolette Grobler, Ministry of Fisheries and Marine Resources Namibia, pers. comm.). This follows that, depending on the rate of onset and intensity of the hydrogen sulphide eruptions, not all lobsters will be able to evacuate the impacted regions of the seabed in time to escape the debilitating effects of the sulphide. However, the results presented here suggest that, depending on the duration of the sulphide event – a function of current regime – this may not always prove fatal for the lobsters that are caught off-guard by the eruptions and are trapped (i.e. become 'moribund') while still on the reef.

In the 1999 international symposium on lobster health management, Evans (1999) discussed important issues pertaining to the physiological (and behavioural) responses of lobsters to various degrees and types of environmental and anthropogenic stress. He noted that lobster 'health' (i.e. the physiological state of the animal) covered a host of biological responses such as growth, maintenance, disease defense and reproduction. Evans (1999) identified that decapod crustaceans employ a wide range of defense reactions (stress responses) aimed at preventing damage to body tissue or abnormal physiological function, and outlined four different stages of stress responses for lobsters *viz.*: Stage 1 – the initial alarm phase; Stage 2 – the subsequent resistance phase; Stage 3 – the late resistance phase or early exhaustive phase; and Stage 4 – the advanced exhaustive phase.

Obviously, the severity and type of stress being dealt with determines the onset of these stages as well as how long the stages last (Hall & Van Ham 1998, Evans 1999). The responses observed during the short-term behavioural experiments can easily be linked to the different stages described by Evans (1999). The 'no movement' response is analogous to Stage 1, which according to Evans (1999) is the start of the nervous reaction and encompasses physiological responses such as the secretion of stress hormones. The 'prolonged movement' response falls under Stage 2 where physiological responses include the increased mobilization of energy (lactate) and is associated with pathological reactions such as water and ion disturbances as well as early cell injury (Evans 1998).

During Stage 3 extensive cell injury usually ensues which necessitate the use of forceful flight measures to evade the stressful conditions (akin to the 'tail-flip' response) – which is considered to be a chronic stress response level by Evans (1998). Stage 4, the late exhaustive phase, is associated with such extensive cell damage that it results in organ failure and as a consequence leads to impaired motor and other physiological responses (Evans 1998). The 'moribund' phase of the short-term behavioural experiments is thus part of Stage 4. In line with the findings of this study, Evans (1998) also noted that if the stress was not removed during the fourth stage, death would eventually ensue. The reason for this is that at this point, even if the stress does not induce complete incapacitation, sub-lethal physiological effects (such as weakened immune responses, reduced respiratory rates and metabolic efficiency) set in and render the animal vulnerable to bacterial infections which catalyse and further enhance abnormal physiological operations eventually leading to death (Evans 1998, Cockcroft 2001, Cockcroft *et al.* 2002).

In terms of the inter-site differences in behavioural responses, lobsters from Lüderitz initiated the 'tail-flip' response and became 'moribund' at concentrations of hydrogen sulphide that were significantly lower than the levels at which the Cape Town lobsters exhibited the same responses (Fig. 3.9 & Table 3.4). It is hard to escape the conclusion that reaching moribundity at low levels of stress is a negative adaptive trait in the Lüderitz populations. However, it can be argued that early initiation of flight responses (tail flips) is a positive, rather than a negative, trait of the Lüderitz lobsters, since if they sense the stress earlier and can evade its impacts in good time. There may be both negative and positive outcomes of this. Early escape may allow avoidance of stress; but tail flips are energetically expensive. If stress and avoidance are sustained they may deplete energy supplies, leading to 'moribundity' being reached early.

Environmental conditions

Dissolved oxygen

The experimental dissolved gas concentrations were well within range of naturally observed concentrations. Dissolved oxygen levels in bottom to mid-water (30-80 m) regions naturally range between 1 and 3 mg/ℓ (Pulfrich *et al.* 2006). However, low-oxygen events (<1 mg/ℓ) are a common feature of the Benguela upwelling system and are controlled by seasonal cycles as well as stochastic processes within the system (Dingle 1995, Shannon & O'Toole 1999, Pulfrich *et al.* 2006).

In terms of the scope of this study juvenile lobsters seem to be resilient to short-term exposure to extremely low (<0.5 mg/ℓ) dissolved oxygen conditions. It is thus plausible that mass strandings associated with low oxygen events will take place only when low-oxygen conditions are prolonged (Cockcroft *et al.* 1999, Wilson *et al.* 2003, Pulfrich *et al.* 2006) or when they are compounded by other factors such as the generation of hydrogen sulphide (Cuomo *et al.* 2003). Fatigue, resulting in the eventual asphyxiation of confined lobsters during extended periods of hypoxia, may be the overriding physiological factor compelling the avoidance of oxygen-poor water, with consequent walk-outs. In addition, prolonged exposure to reduced dissolved oxygen conditions could weaken the lobsters' immune systems making them more susceptible to disease which could lead to death (Evans 1998, Robohm & Draxler 2003). Furthermore, sub-lethal effects resulting from prolonged exposure to hypoxia could severely impact growth, fecundity and recruitment (Beyers *et al.* 1994, Cockcroft 2001).

The findings by Bailey *et al.* (1985) that lobsters generally avoid water with dissolved oxygen concentrations of <2 mg/ℓ suggest that the critical physiological strain may be at this level of dissolved oxygen. However, it would be interesting to know what fatigue-level lobsters tolerate before they are compelled to move away from the stress (i.e. is 2 mg/ℓ indeed the critical tolerance level of hypoxia for lobsters and the reason for walk-outs?). If this does prove to be the case then it would imply that pre-walkout tolerance might be at a level of dissolved oxygen higher than 2 mg/ℓ.

Hydrogen sulphide

Bailey *et al.* (1985) recorded concentrations of 10 to 30 mg H_2S/ℓ in the sediment along the Namibian coast. This infers that during natural hydrogen sulphide eruptions, the sulphide concentrations in the water immediately above the sediment will be within this range (Bailey *et al.* 1985, Matthews & Pitcher 1996). Interestingly Matthews & Pitcher (1996) found that, during the mass marine mortality event of 1994, hydrogen sulphide concentrations were generally higher in the surface, rather than the bottom, layers of the water column. This was attributed to the depletion of dissolved oxygen in the surface waters by massive algal blooms that lead to the surface waters becoming anoxic. The subsequent reduction of sulphates, during anaerobic respiration by bacteria in the water column, resulted in the concentration of hydrogen sulphide gas near the sea surface (Matthews & Pitcher 1996).

This study showed that the critical level of hydrogen sulphide tolerance for the Lüderitz lobsters was between 3.0 and 5.7 mg/ℓ and that the level at which 'moribundity' occurred was between 14.8 and 33.4 mg/ℓ. This implies that natural sulphide eruptions could result in the rapid incapacitation of lobsters, rendering them defenseless. The outcomes of such catastrophic events are fundamentally dependent on the intensity and duration of exposure to the stress.

Diamond dredge-mining impacts

It has been suggested that diamond dredge-mining activities may alter dissolved oxygen and hydrogen sulphide conditions in mined areas, potentially increasing the frequency of hypoxic events by increasing the suspension of nutrient-rich sediments. The review of data (Chapter 2) yielded no conclusive evidence that dredging had an influence on dissolved oxygen profiles within the water column or near the seabed. As regards the hydrogen sulphide, I unreservedly conclude that negative impacts will become an issue if dredge-mining takes place in sulphide-prone areas. However, the extents of these perturbations require further investigation.

In conclusion: hypoxia and elevated hydrogen sulphide events are integral features of the Benguela ecosystem – with hydrogen sulphide being a seemingly more significant factor with regards to its catastrophic impacts. It is, thus, reassuring to know that lobsters are able to survive short-term exposure to debilitating concentrations of hydrogen sulphide. However, it remains to be seen what relationship(s) exists between sediment suspension (an inherent factor of dredging) and these two important gases, and what could be done to mitigate and/or reduce potentially catastrophic outcomes with regard to lobster populations in mining areas.

CHAPTER 4: IMPACTS OF TURBIDITY ON JUVENILE ROCK LOBSTERS

Introduction

Dredging Attributes

Dredging impacts encompass the mechanical destruction of benthic communities and sediment suspension, which increases turbidity within the water column. In all dredging activities one of the most significant attributes is the sediment suspension resulting from the removal of sediment with the dredging tool. Sediment suspension is highly variable between technologies since different dredgers suspend sediments to different degrees (Carter 1995).

De Beers Marine Namibia (DBMN) undertook an Environmental Impact Assessment (EIA) to assess the impacts of a Trailing Suction Hopper Dredger, TSHD (Pulfrich *et al.* 2004; see Chapter 2). The dredger works on the basis of dragging a drag-head (i.e. the dredging tool), attached to the ship via a diagonally supported 1.4-m thick suction pipe across the seabed. Sediment is sucked up by the trailing drag-head, in line with the ship's course, and transported to the ship via the suction pipe. The dredger is capable of operating in fixed depths ranging from 45 to 131 m. Typical dredging-induced turbidity levels range between 20 and 1100 mg/ℓ depending on the nature of sediment (Carter 1995).

Turbidity Attributes

Sediment suspension

Carter (1995) has outlined turbidity features associated with dredging. The highest concentration of suspended sediment is observed at the drag-head, and turbidity rapidly declines with distance away from the drag-head. Concentration gradients are steep, with suspended sediment concentrations falling to approximately 25% of the initial concentrations just 50 m away from the drag-head.

Suspension time is a function of sediment particle size, aggregation and turbulence induced by currents (Appleby & Scarratt 1989, Carter 1995). The first and third factors are of greatest significance and are inherently linked. Appleby & Scarratt (1989) calculated critical suspension velocities for different types of sediment and found that clay particles, ≤4 μm, were kept in suspension by current velocities of 8 cm/s while sand particles, 500 μm, were kept in suspension by currents flowing at speeds of about 28 cm/s. Thus, suspension time is governed by a direct positive correlation between particle size relative to mass and current speed. Sediment aggregation functions as an indirect factor, linked to particle size, in the sense that it increases the sediment size, thus causing the particles to become heavier and altering the critical suspension velocities (Carter 1995).

Sediment suspension impacts

For extension of the Saldanha Bay quay on the west coast of South Africa, a literature review was undertaken of the impacts of suspended sediment, considered as turbidity (Carter 1995). The most common impacts involved reduced light penetration, gill clogging in fish and filter feeders and altered feeding efficiencies in filter feeders. The lowest turbidity at which significant impacts were observed was 5 mg/ℓ. Howver, studies elsewhere showed that at this level of sediment suspension, the rate of filter feeding by the mussel *Mytilus edulis* is significantly increased (Kiørboe *et al.* 1981, Carter 1995). The highest sediment suspension at which negative impacts have been observed was 117 000 mg/ℓ, at which 80% of the population of English sole (*Perophyrus vetulus*) died within 240 hours of exposure (Peddicord *et al.* 1975, Carter 1995). Interestingly, one study showed that turbidity levels ranging between 6 and 25 mg/ℓ increased food absorption efficiencies in bivalves (Robinson *et al.* 1984, Carter 1995), which indicates that turbidity can be beneficial as well as harmful to marine species.

Natural turbidity levels in southern African waters are known to range between 0.5 and 20.9 mg/ℓ (Carter 1995; and see Chapter 2). However, storms are capable of magnifying these concentrations by two orders of magnitude, and species living in the intertidal and shallow subtidal regions of the ocean are well adapted to survive such sediment loads (Carter 1995).

Objectives

This section of the study was structured around three objectives: (1) to assess the short-term effects of turbidity on the behavioural and feeding responses of juvenile rock lobsters, (2) to assess the medium-term effects of turbidity on behavioural and feeding responses of juvenile rock lobsters, and (3) to determine if any correlations exist between turbidity and other physical factors (specifically pH, temperature and dissolved oxygen) as determined in the laboratory.

In line with these objectives the turbidity experiments were designed to test the following four null hypotheses: (1) short-term sediment suspension will not induce avoidance behaviour or influence the rate of feeding of juvenile rock lobsters; (2) medium-term sediment suspension will not affect the feeding rate of juvenile rock lobsters; and (3) no correlation will exist between suspended sediment and (a) pH, (b) dissolved oxygen, (c) water temperature, as determined from laboratory experiments.

Materials and Methods

Sediment Texture and Settlement Rate Analyses

Sediment texture analyses

Three sediment samples, roughly 20 ℓ each, were obtained from three areas just south of Lüderitz, in the region earmarked for future diamond dredge-mining (E. Effenberger pers. comm.). The samples were collected on board the *Snowgoose* lobster fishing vessel during a 2-day lobster fishing trip. A Van Veen grab sampler (Fig. 4.1) was used to collect the sediment at the same stations where the lobsters were captured for the project. The samples were placed in three different bins and transported back to the Marine and Coastal Management Research Aquarium in Sea Point, South Africa for processing and experimentation.

Figure 4.1: The Van Veen grab sampler (grey) and crane (blue) used for the sediment sampling.

The sediment samples were analyzed at the Sedimentology laboratory of the Geology Department at the University of Cape Town. For the sediment composition analyses, subsamples (40 g) from each of the different sediment samples were freed of salt by osmosis across a cellophane membrane. The subsamples were each suspended overnight within cellophane tubing in a large bucket of tap water that was continually replaced with slow-flowing fresh water.

The clay and silt (mud) fractions were then separated from the sand and gravel fractions by wet-sieving the subsamples through a 63-µm sieve. The mud masses of the three sub-samples were then determined by means of the Andreasen-pipette method in which an aliquot was extracted, with a calibrated 25-mℓ pipette, from a 1:1 (stirred) homogenous suspension in a graduated cylinder and transferred to a pre-weighed glass beaker. After drying at 105°C the mass difference was multiplied by a predefined pipetting factor to obtain the mass of mud.

The course fraction (>64 µm) from each of the three subsamples was dried, dry-sieved through a 2-mm sieve to separate the sand and gravel fractions, and weighed to obtain the mass of sand and gravel. The proportions of sand, gravel and mud were calculated and textures assigned to categories using the method of Folk (1954), e.g. sandy mud or muddy sand. A binocular microscope was used to identify the components of the gravel and sand fractions as well as to control the quality of the laboratory procedures (Rogers & Li 2002).

The organic and inorganic carbon contents of the samples were also determined. Single subsamples (5 g) from each of the three sediment types were acid-leached with hydrogen chloride to remove calcium carbonate, and another set of subsamples treated with hydrogen peroxide to remove fecal pellets from the samples. The samples were dried overnight at 105°C and analyzed the following day. The percentage of organic and inorganic carbon for, respectively, the peroxide-treated and acid-leached samples were calculated by dividing the post-treatment weights of the samples by their pre-treatment weights and multiplying by 100.

Sediment settlement rate analyses

For the settlement analyses the sand fractions (64 μm to 2 mm) of each of the subsamples were split in a sample splitter to obtain statistically random splits of 2-3 g. The three subsamples were processed through a 2-m long settling column. The accumulating mass of sand was weighed at 3-second intervals on a Perspex disc, inside the settling column, suspended from an electronic balance above the settling column. Settling rates and Phi-fractions were determined in real time from an online computer connected to the settling column and electronic balance. Phi (Ø) fraction data or Wentworth grades were analyzed according to categories outlined by Wentworth (1922), defined as follows: very fine sand (63-125 μm); fine sand (125-250 μm); medium sand (250-500 μm); coarse sand (500-1000 μm) and very coarse sand (1000–2000 μm). The settlement data obtained from the settling column were used in conjunction with the textural data in the final descriptions of the different sediment types.

Short-term Turbidity Experiments

For the short-term turbidity experiments two 75-ℓ (50 × 50 × 30 cm) glass aquaria were used. Only male lobsters caught near the Cape Town harbour (in the inter-moult stage) were used in the turbidity experiments. As the analyses of the sediment samples revealed that two of the three samples (samples A and C) were virtually identical in terms of their sedimentological properties, the experiments on the effects of turbidity of the rock lobsters were run with sediment samples A and B only.

A 10-cm thick sediment layer, from either samples A and B, was placed in two separate aquaria and allowed to settle for roughly two days. An assessment was carried out to determine if there were any correlations between turbidity and dissolved oxygen and pH. For these correlation-assessment experiments the sediment in both aquaria was vigorously agitated for 30 seconds to simulate suspension by dredging, and physical variables measured at 60-min intervals for as long as the sediment remained in suspension. These experiments also served to establish turbidity attributes. Turbidity was measured by taking 100-mℓ water samples every 60 minutes and placing them in a heating oven at 34°C. Once all the water had evaporated the sediment left inside the beakers was weighed and extrapolated to estimate sediment load in milligrams per litre of seawater. A constant salt mass of 35 g/ℓ seawater was used to differentiate between the salt and sediment portions left over in the beakers. These experiments were carried out once per day over three consecutive days.

The experiments on the short-term turbidity impacts on rock lobsters were carried out in two 27-ℓ aquaria with a 10-cm layer of sediment in each – i.e. each aquarium had a different sediment type. Single temporary dividers were used to create two compartments in each of the two aquaria, and individual juvenile male lobsters were placed in one of the compartments of each the aquarium. The empty compartment was used to induce turbidity, through the vigorous agitation of the sediment with a plastic probe for roughly 15 seconds. The dividers were then immediately removed so that the suspended sediment dispersed throughout the water in the aquaria. Initial turbidities were about 35 mg/ℓ for Sediment A and 12.5 mg/ℓ for Sediment B. Druing the sediment dispersion phase the behaviour of the lobsters was observed and recorded. Food was introduced into the aquaria approximately one minute after the dividers were removed and the responses of the lobsters to the food observed and recorded. Starvation level was standardized by conducting the experiments only on feeding days (i.e. every second day). The experiment was repeated six times (for each sediment type) with different lobsters being used in each run. Control animals were held in aquaria without any sediment or stirring, but otherwise treated identically to experimental animals.

Medium-term Turbidity Experiments

For the medium-term turbidity experiments, three 75-ℓ glass aquaria were used – each mounted on four magnetic stirrers (Fig. 4.2) for keeping the sediment in suspension. In each of the three aquaria four compartments were created by placing two grey PVC sheets perpendicular to each other inside the aquaria (Fig. 4.3 C). In each of the four compartments per aquarium there was a platform (made from rigid plastic sieve elevated on 5-cm stilts) which kept the lobsters away from the magnetic stirrer bars placed on the bottom of the aquaria (Fig. 4.3 B).

Figure 4.2: Two of the three medium-term experimental aquaria. The magnetic stirrers and the polystyrene support between the aquaria and the stirrers are also shown.

Magnetic stirrers (equipped with 80-mm long, white, cylindrical magnetic stirrer bars) were used to keep the sediment in suspension throughout the experiment (Fig. 4.2 and Fig. 4.3 C & D). Each compartment of the two aquaria had a magnetic stirrer machines below it, including the control aquarium that lacked sediment.

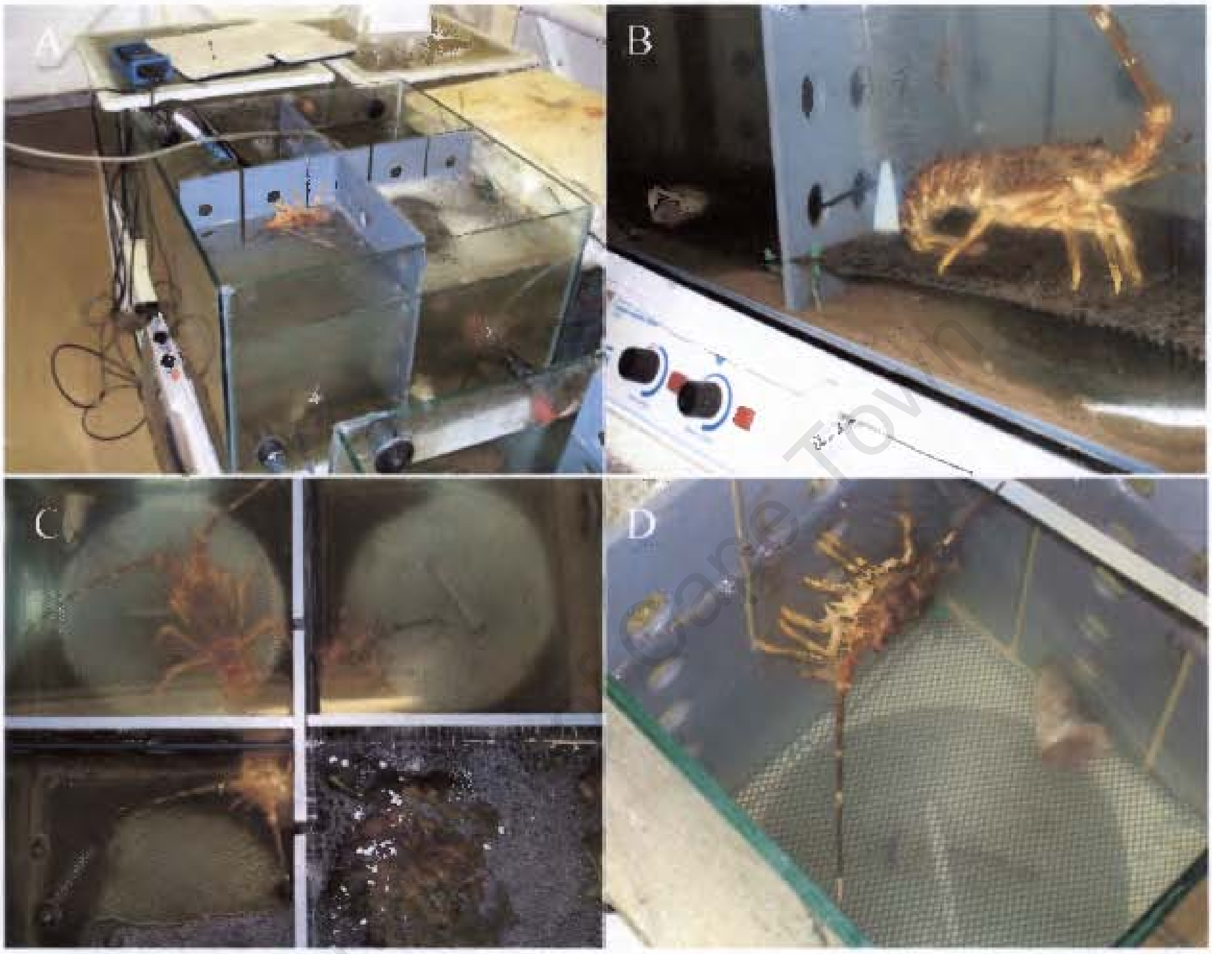


Figure 4.3: Various components of the medium-term turbidity experiments. A and C show the four compartments created with the two perpendicular PVC sheets per aquarium. B and D show the platforms that kept the lobsters away from the stirrer bars as well as the food (pilchard heads), with which the lobsters were fed. Picture D was taken from the control aquarium while pictures A, B and C were taken from aquaria used for sediments. Also evident in C and D are the precise fits of the polystyrene supports around the stirrer pads.

All the magnetic stirrer machines had square polystyrene supports with a central hole, cut to the exact size of the stirrer machine pad, supporting the weight of the aquaria on the frames of the stirrer machines (Fig. 4.2 and Fig. 4.3 B, C & D). The polystyrene supports served two main purposes: (1) to keep the bottom of the aquaria off the stirrer machine pads, and (2) to distribute the weight of the aquaria (plus water) over a broader surface (i.e. the polystyrene surface) and onto the sturdy frames of the stirrer machines rather than the stirrer pads alone. The polystyrene pads also minimized heat transfer from the stirrer machine to the water in the aquaria.

During the experimental phase, roughly 2 kg of sediment from the two sediment types was placed in the aquaria – different types in separate aquaria – and allowed to settle for two days. After the sediment had settled the magnetic stirrer bars were placed inside each aquarium compartment. The platforms were also placed in the aquarium compartments. Juvenile lobsters, all males from Cape Town, were then selected and placed on top of the platforms inside the aquaria, one per compartment. The lobsters were left to acclimate for roughly 30 minutes after which the experiments were commenced by turning on the stirrer machines. Fresh air was supplied to each of the aquaria with the aid of three 5-mm rubber tubes (each with an airstone at the end) connected to the air supply of the research aquarium. Air was supplied because the non-circulation design of the experiment resulted in rapid depletion of dissolved oxygen inside the aquaria: oxygen was depleted in less than nine hours without an external supply.

The experiments were run for five and a half days, during which time the sediment was continuously kept in suspension by the stirrers. Lobsters were fed three times during this period: on the first, third and fifth days. Lobster behaviour and feeding were monitored. Turbidities induced were in the range of 11 to 13 mg/ℓ for sediment A and 5 to 7 mg/ℓ for sediment B. In hindsight these levels were too low to meaningfully test the effects of mining-induced turbidity, which reaches much higher levels of ~ 150 mg/ℓ. This deficiency reflects limitations in the equipment used, as the magnetic stirrers employed were incapable of generating higher levels of turbidity.

Turbidity was measured in the same way as that for the short-term turbidity experiments. Three controls were put in place: (1) a male lobster with food as well as stirring activity induced by the stirrer; (2) a male lobster with food only (no stirring), and (3) stirring activity and food only. No sediment was used in any of the three controls. The third control was set up to determine if the stirring activity resulted in any food loss in the absence of feeding. However, results showed that very small amounts of food were lost in this control (1.65%, compared to 38.4% in replicates with lobsters), and this amount was deducted from all food losses recorded from the other treatments. All experiments were conducted at ambient temperature, which was around 20°C. All lobsters used in the experiments were in the inter-moult phase.

The experiment was terminated on the 6th day. Each time the lobsters were fed, the food was left in the aquaria for 24 hours, after which the left-over food was retrieved and weighed. Lobsters were fed on pre-weighed pilchard heads. A final round of measurements of physical parameters was taken on the last day of the experiments before the lobsters were removed. The physical factors measured were temperature, dissolved oxygen and pH, all measured using the WTW multi-parameter sensor described in the 'Experimental phases' section of Chapter 3.

Data analyses

The data obtained from the turbidity experiments were analyzed in Statistica 7. For the medium-term data an analysis of variance (ANOVA) was carried out on the data as well as post-hoc comparative analyses. Levene's test for homogeneity, normal probability plots and histograms revealed that the data did adhere to the ANOVA assumptions of normality and equality of variance, so the data required no transformation.

Results

Sediment Texture and Settlement Rate Analyses

The sediment composition analyses revealed that two of the samples, A and C, were almost identical, having a median particle size of 1.95 Ø (259 μm), whereas sample B was coarser, with a median particle size of 2.5 Ø (177 μm) (Figs 4.4 & 4.5). These median particle sizes correspond, respectively, to categories of fine-to-medium sand and fine sand (Wentworth 1922).

Further insight came from the overall composition of the sediments. Samples A and C had substantial silt (53%) and clay (8.4%) proportions, whereas sample B had small silt (1.75%) and clay (3.18%) components (Fig. 4.6). It was on the basis of this that the two samples A and C were considered as representing the same sediment type (sandy mud), and sample B another type (fine sand). For this reason only sediment types A and B were used in the short-term and medium-term turbidty experiments.

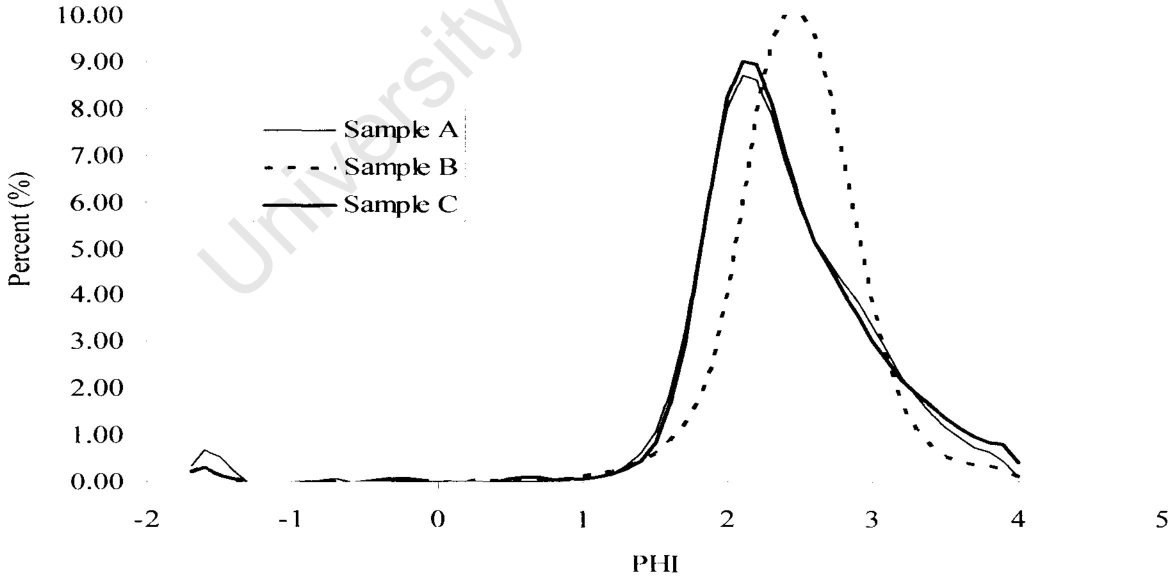


Figure 4.4: Particle-size analysis for sediment samples A, B and C treated with hydrogen peroxide to remove fecal pellets.

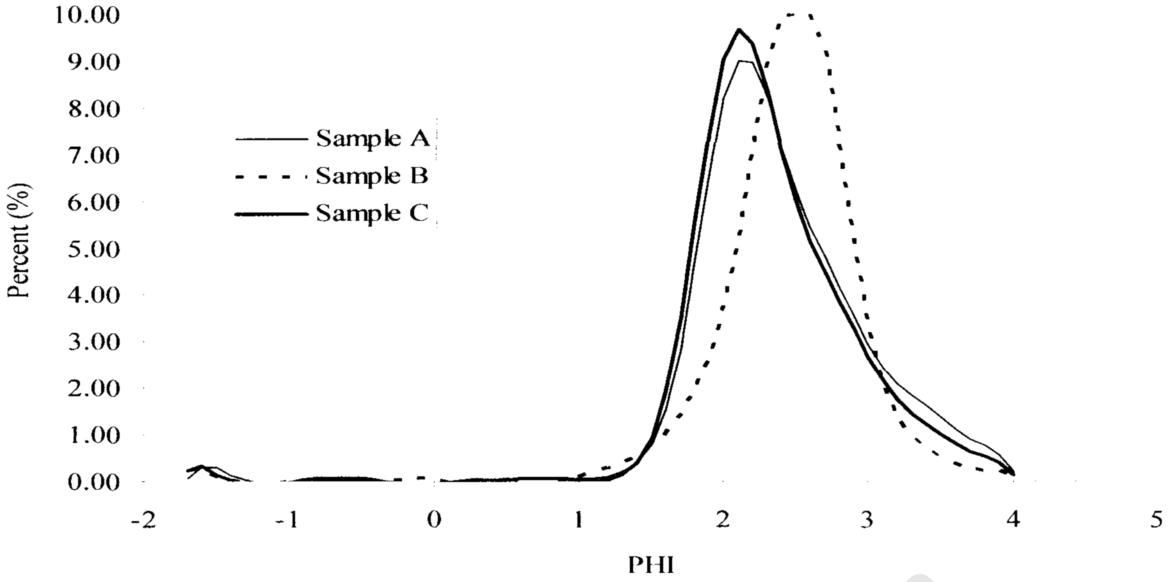


Figure 4.5: Particle-size analysis for sediment samples A, B and C after leeching with hydrogen chloride to remove calcium carbonate.

Going by the Wentworth (1922) descriptions, sample A was classified as slightly gravelly sandy mud. This is because the mud-to-sand ratio was between 9:1 and 1:1 (Wentworth 1922). Sample B was classified as slightly muddy sand. Clay is an important constituent in the textural analysis as it plays a crucial role in the classification of mud (i.e. its absence implies only very fine sand classifications and no mud classifications). Clay also has an important influence on turbidity profiles as it is suspended the longest of all the sediment grades.

Sediment settlement rates were extracted from the settling column data which covered the whole range of the sediment grades for the two sediment types. From the cumulative fraction data it was found that 4 Ø (63 μm) sediment particles settled at a rate of 4 mm/s. The 1 Ø (500 μm) particles settled at a rate of 80 mm/s.

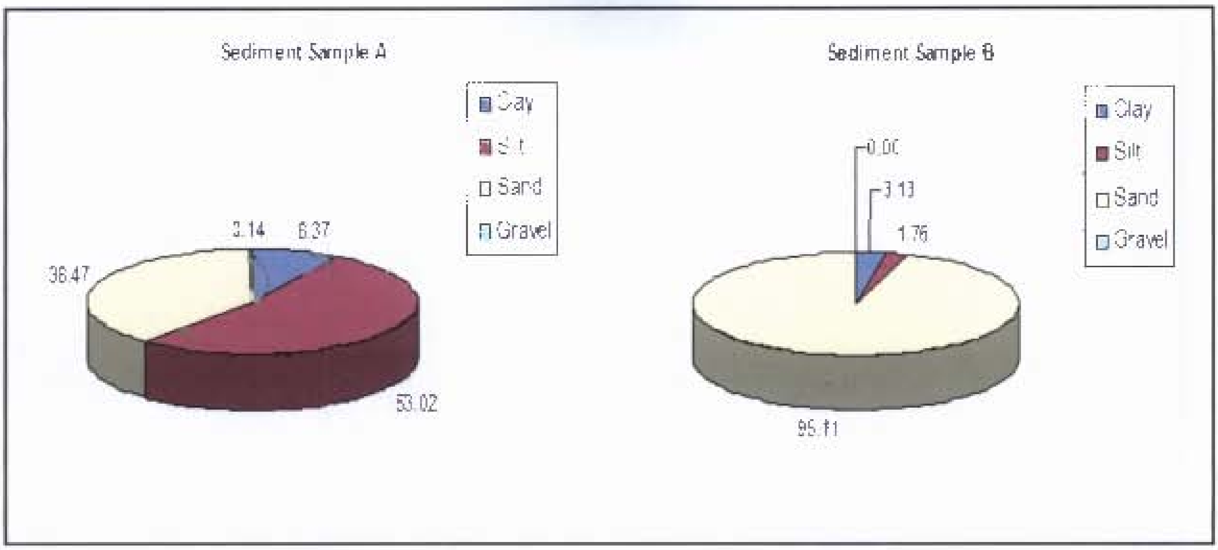


Figure 4.6: Sediment texture analyses results represented as percent composition.

Short-term Turbidity Results

Other than the changes in turbidity, no obvious differences were observed between any of the physical factors that were monitored during the short-term turbidity experiments (Table 4.1). Values of pH remained at or close to the control values of 8.1, and temperature remained constant throughout the experimental periods and within range of 19.9-20.2°C. Oxygen concentrations remained virtually unchanged in all runs but were consistently lower in the treatments with sediment A (3.85 mg/ℓ) than with sediment B (4.04 mg/ℓ) and higher in the controls (4.37 mg/ℓ). The difference in oxygen concentrations were, however, small – in the region of 0.5 mg/ℓ. The suspension times for the two sediment types were about 4 h for Sediment A and less than 1 h for Sediment B (Table 4.1).

Table 4.1: Data for the physical factors monitored during the short-term turbidity experiments. Dissolved oxygen and turbidity levels were measured in mg/ℓ seawater and temperature in degrees Celsius (°C).

Description		Experimental Run 1								
Time [min.]		0	60	120	180	240	300	360	420	480
Control	pH	8.131	8.183	8.181	8.170	8.124	8.100	8.131	8.131	8.131
	O_2	4.40	4.39	4.38	4.35	4.38	4.38	4.37	4.30	4.36
	°C	19.9	19.9	19.9	19.9	19.9	19.8	19.7	19.8	19.8
	Turb.	2.0	1.3	1.9	0.7	1.7	0.4	1.8	1.8	1.5
Sediment A	pH	7.965	7.964	7.959	7.958	7.957	7.957	7.955	7.955	---
	O_2	3.91	3.90	3.90	3.90	3.89	3.90	3.90	3.90	---
	°C	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	---
	Turb.	37.2	22.7	10.9	5.3	2.4	1.1	0.5	0.3	0.1
Sediment B	pH	8.189	8.113	8.117	8.117	8.102	8.110	---	---	---
	O_2	4.12	4.12	4.12	4.12	4.10	4.11	---	---	---
	°C	20.0	20.0	20.0	20.1	20.1	20.1	---	---	---
	Turb.	14.3	0.7	0.1	0	0	0	0	0	0
Description		Experimental Run 2								
Time [min.]		0	60	120	180	240	300	360	420	480
Sediment A	pH	8.102	8.101	8.100	8.100	8.099	8.098	8.098	8.097	8.094
	O_2	3.87	3.86	3.85	3.85	3.85	3.85	3.84	3.85	3.85
	°C	20.1	20.1	20.1	20.1	20.1	20.1	20.1	20.2	20.2
	Turb.	31.9	19.4	9.0	4.4	2.1	0.9	0.4	0.2	---
Sediment B	pH	8.145	8.144	8.144	8.144	8.144	---	---	---	---
	O_2	4.06	4.06	4.05	4.05	4.05	---	---	---	---
	°C	20.1	20.1	20.1	20.1	20.1	---	---	---	---
	Turb.	10.7	0.5	0	0	0	---	---	---	---
Description		Experimental Run 3								
Time [min.]		0	60	120	180	240	300	360	420	480
Sediment A	pH	8.090	8.090	8.088	8.085	8.085	8.085	8.083	8.083	---
	O_2	3.80	3.80	3.80	3.80	3.80	3.80	3.79	3.79	---
	°C	20.2	20.2	20.2	20.2	20.2	20.2	20.2	20.2	---
	Turb.	35.1	21.4	10.1	4.7	2.2	1.0	0.5	0.3	---
Sediment B	pH	8.140	8.141	8.141	8.141	---	---	---	---	---
	O_2	3.97	3.97	3.96	3.96	---	---	---	---	---
	°C	20.1	20.1	20.1	20.1	---	---	---	---	---
	Turb.	12.8	0.6	0	0	0	---	---	---	---

No clear patterns emerged with regards to the behavioural response of lobsters to sediment suspension during the short-term turbidity experiments. Behaviour was random during turbidity phases. Some lobsters tended to stay inactive, remaining in one place for the entire duration of the experiment while others moved around intermittently. Nothing distinguished the movements of control animals from those in the turbidity treatments. However, no tendency to feed was evident in lobsters that were in the turbidity treatments, while all control animals fed normally.

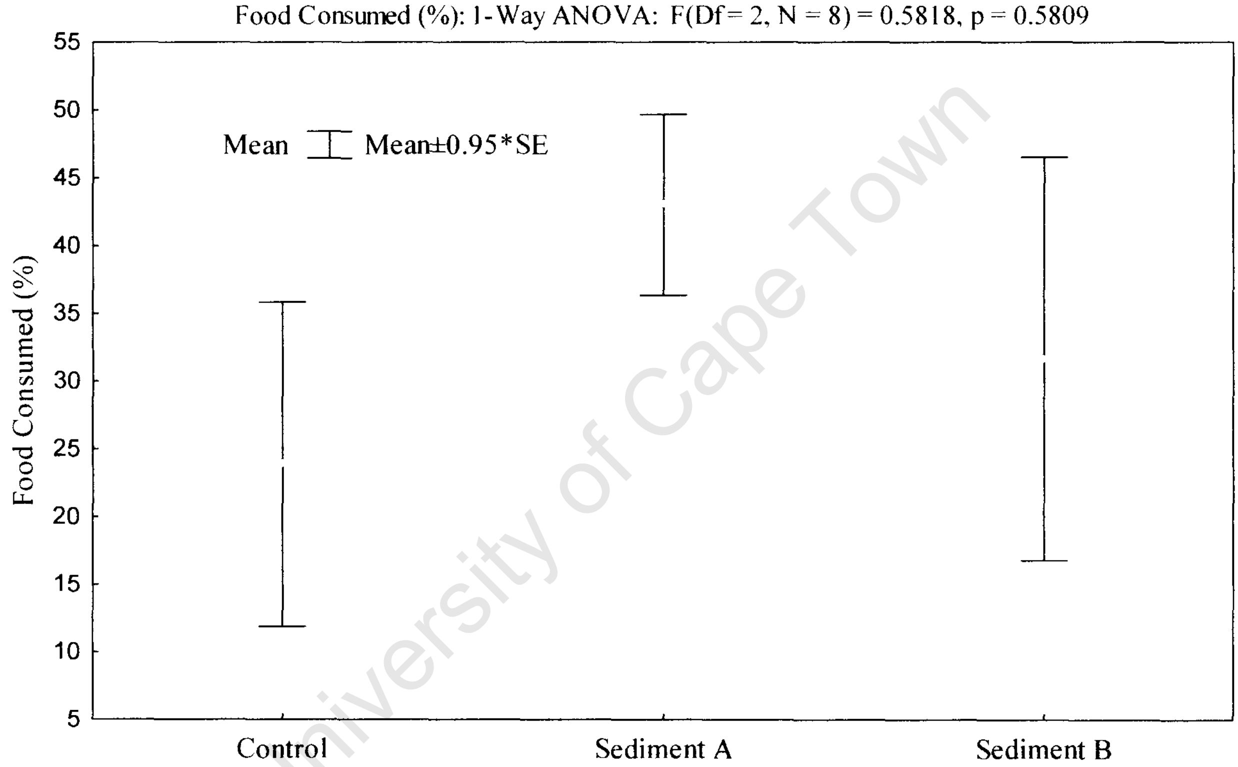


Figure 4.7: Comparisons between the feeding rates of lobsters in controls and in treatments where lobsters were exposed to either sediment A or sediment B. Data were adjusted to allow for the fact that there was a 1.65% loss of food in the stirrer-and-food-only control.

Medium-term Turbidity Experiments

The most useful feeding data were obtained on day three of the medium-term turbidity experiment, when all the lobsters fed. No lobsters - including those in the controls - fed on the first day, and only two (one control animal and one experimental animal) fed on the fifth day.

Comparative statistical analyses between the feeding rates on day three of the lobsters in controls 1 and 2 and those exposed to the two different types of sediments were carried out but no statistically significant difference was found between the three groups (Fig. 4.7; 1-way ANOVA: $F_{2,8} = 0.58$, $p = 0.58$).

The most common behaviour observed during the medium-term turbidity experiments was that lobsters climbed up against the grey PVC dividers that were used to keep adjacent lobsters apart. This behaviour was most pronounced in the Sediment A aquarium with roughly nine (compared to two for Sediment B) out of 20 observations being made throughout the 5-day duration of the experiment. The control lobster exhibited no such behaviour during the experiment. The general tendency of lobsters was to group together in the centre of the aquarium, generally closest to the source of oxygen input, as is evident in Figure 4.3 C. No mortalities were recorded during the medium-term turbidity experiments.

Discussion

The seabed on the continental shelf of the southern African west coast is characterized by three main sediment grades – mud, sand and gravel. All three types constitute unconsolidated sediment derived from land-based and marine sources of weathered consolidated and semi-consolidated rock types and are distributed randomly throughout the Benguela region, generally situated in depressions, cracks and gullies within the underlying (igneous) bedrock (Rogers & Li 2002).

The general distribution pattern is that mud is found in sheltered marine areas such as bays and harbours (Rogers & Li 2002). The reason for this is that mud settles easily in sheltered areas because of the low or slow current regime that is normally prevalent in these habitats. In the

Northern Benguela, mud is a prominent feature in the shallow to mid-water (<200 m) depths of the system (Rogers 1977, Emeis *et al.* 2004), confined to the region between Möwe Point and Sylvia Hill along the Namibian coast (Fig. 1.1 in Chapter 1).

There is an extensive mud belt extending northwards from the Orange River delta. In general, however, mud constitutes a small proportion of the total sediment make-up of the Benguela seabed. The modal sediment type is sand, with gravel being the second major constituent (Rogers 1977, Rogers & Li 2002). In essence gravel is the source of all non-terrestrial sand, since sand is basically weathered gravel particles. The greatest concentration of gravel is normally on seabed areas closer to the coastline, while sand is distributed intermittently throughout the depth profile with peaks in the mid-water regions (Rogers & Li 2002). The theory behind this is that strong currents and wave action close to the coast result in the weathering of consolidated and semi-consolidated rock. Gravel and pebble constituents are then kept within the near-coast regions, because of the fast settling characteristics of these grades, while sand is advected offshore to deeper depths where the settling velocity of this grade overrides the current velocity that maintains suspension (Rogers & Li 2002).

The sediment components of the study area comprise mainly gravel and sand (Pulfrich *et al.* 2004b). Sediment samples for the turbidity assessment of the study were taken in Chameis Bay and covered the full spectrum of the sediment types (Sediment A: sandy mud, Sediment B: fine sand) found in this region.

Turbidity Effects on Physical Factors

In the context of this study, turbidity was considered as the suspension of sediment in the water column. Laboratory assessments on the impacts of turbidity yielded no evidence that increases in turbidity influenced the parameters of physical factors such as pH, temperature and dissolved oxygen. However, having been carried out in a laboratory the general applicability of these findings is debatable, because many *in-situ* factors – such as sediment nutrient content, sediment chemical attributes and current attributes that are inherently linked to the suspension of sediment – were excluded from the assessments so that the results portray an unrealistic picture of the potential impacts of turbidity.

Considering the 'sediment chemical attributes' factor, dredging-induced turbidity in hydrogen sulphide prone areas will undoubtedly have significant influences on the pH and dissolved oxygen profiles of the impacted areas (Shooter 1999). Dredging-induced suspension of nutrient-rich sediment can result in unnatural phytoplankton blooms which will influence dissolved oxygen profiles in the water column (Shannon & O'Toole 1999).

If turbid conditions are prolonged as a result of continuous dredge-mining, the potential for anoxic conditions could be increased – which in turn may lead to the formation of hydrogen sulphide particularly in areas where the sediments are fine, or in deeper waters. Current regime plays an important role in that it dictates the suspension time of sediment as well as the spatial context of suspended sediment (Berger & Wefer 2002). It is thus important that all these factors – as well as those not defined yet – be incorporated in the assessments of the impacts of dredge-mining induced turbidity by way of conducting *in-situ* assessments.

Although no correlations between sediment suspension and pH, dissolved oxygen or temperature came to light through this study, it must be borne in mind that the duration and nature of the study were restricted. However, considering that average current speeds within the Chameis Bay area are around 6.61 cm/s for roughly nine months of the year, with maximum velocities of up to

20.87 cm/s confined to Summer (Pulfrich *et al.* 2006), sediment settlement rates there are likely to be fairly rapid, particularly in Chameis Bay where the sediments are relatively coarse. For similar reasons, turbidity plumes generated by the dredge-mining will be relatively confined to the vicinity (<50 m) around the mining or discharge areas, and suspension times are likely to be short.

Turbidity Impacts on Rock Lobsters

No negative impacts on rock lobsters were observed during the short-term exposure to sediment suspension. Behavioural responses varied between random movements and no movement during turbidity phases, with no obvious differences between controls and treatments. Feeding was suspended during turbidity events, while control lobsters exhibited normal feeding behaviour.

Settling rates for both sediment types were relatively fast compared to natural cycles (Carter 1995). Under natural conditions currents keep sediment (particularly fines) in suspension for long periods of time (Carter 1995). Turbidity profiles sometimes persist for three to four days during seasonal rough-sea spells (Carter 1995). Therefore the short-term laboratory simulations represented a calm-water scenario and the fast settlement of sediment meant that the lobsters were exposed to turbidity for a relatively short time for any negative effects to be detected. In addition, the levels of turbidity induced in the experiments were at the lower end of turbidities associated with dredge-mining and future experiments at more extreme levels will be required to assess the mining-related turbidity impacts on lobsters – as discussed in Chapter 5. Based on all these results, the first null hypothesis that there would be no short-term impacts of sediment suspension on lobster behaviour and feeding was not rejected.

Feeding rates during the medium-term turbidity experiments were within range and sometimes even above those of the controls. It was expected that feeding would be suppressed during high turbidity events, as turbidity greatly reduces visibility, which is likely to play an important part in food detection. However, the experiment was constructed in a confined space. In the natural environment reduced visibility may have an impact on both lobsters and lobster predators, and thus influence predator-prey relationships (Heydorn 1969). On the basis of these findings the second null hypothesis regarding the absence of impacts of medium-term turbidity on lobster feeding rate was also accepted.

It should be noted that, although the medium-term experiments yielded no significant effects of turbidity on the lobsters, these were simulated conditions. There are many factors to consider under *in-situ* conditions. Additional studies need to be conducted to accurately ascertain turbidity impacts on benthic communities in the mining areas. Lobsters may be able to survive dredging-induced sediment loads, but other species such as mussels might not be as resilient (Carter 1995). As mussels are an important food source for rock lobsters (Mayfield & Branch 2000, Mayfield *et al.* 2001) any loss of mussel biomass caused by siltation could have severe negative impacts on lobsters in mined areas. Thus a complete *in-situ* study of the impacts of dredging-induced sediment suspension is recommended.

In short, neither set of experiments revealed any negative impacts were evident regarding the effects of turbidity on juvenile rock lobsters. These findings are generally consistent with the study done by Pulfrich *et al.* (2003b) who found that *Jasus lalandii* is resilient to the coastal sedimentation caused by discharges from land-based diamond-processing plants in Elizabeth Bay. They could detect no differences in lobster densities, sizes or sex ratios between control areas and areas impacted by the release of fines into the sea. The laboratory experiments were in the lower range of turbidity since Zoutendyk (1995) found that natural background turbidity levels in the Benguela are generally <12 mg/ℓ and that during storms (and at certain areas along the coast – like near the Orange River mouth) turbidity levels can get as high as 7 400 mg/ℓ. However, Clarke and Wilber (2000) showed that crustaceans can survival turbidity values far exceeding storm turbidity levels (10 000 mg/ℓ) for relatively long periods of time (240 hours). Thus the "no-effect" outcome is not surprising for *J. lalandii*. However, Herrkind *et al.* (1988) did find that high siltation reduced post-larval pueruli settlement for the spiny lobster *Panilurus argus* thus negatively impacting recruitment for this species.

On the basis of present evidence turbidity induced by dredge-mining along the southern coast of Namibia is unlikely to have significant effects on the rock lobster populations. However, this finding cannot be considered in isolation. As outlined in Chapter 5 my experiments on turbidity need to be repeated at higher levels of turbidity, more appropriate to levels that encompass both natural variability and mining conditions. Moreover, longer-term laboratory experiments on the effects of turbidity on growth rates and fecundity, field data on natural variability in turbidity relative to that caused by mining, and the effects of turbidity on benthic communities all warrant investigation before definitive conclusions can be drawn.

CHAPTER 5: OVERVIEW, RECOMMENDATIONS AND THE WAY FORWARD.

Introduction

This project was proposed to address three main issues about the potential impacts of diamond dredge-mining along the Namibian south coast: (1) What are the natural conditions in the mining areas in terms of temperature, hydrogen sulphide, dissolved oxygen and turbidity? (2) To what extent does diamond dredge-mining alter these variables? (3) What are the behavioural responses of juveniles of the rock lobster *Jasus lalandii* to hypoxia, elevated hydrogen sulphide and turbidity? However, due to practical limitations in experimental design and the accessibility of existing data on natural variability and mining-related impacts, not all aspects of these questions could be addressed, and thus the outcomes of the project are only a step towards answering these questions. This chapter reviews the findings of my research, touches on some crucial issues not covered in this assessment and suggests ways of addressing these concerns.

Overview of Study

In marine environments physical and chemical regimes form the laws that define the characteristics of ecosystems, and thus alterations of these physico-chemical regimes may distort the nature of ecosystems. Being of the temporal and spatial grandeur that it is, the Namibian marine diamond-mining enterprise falls in the category of forces that could bring about such regime shifts, but its effects need to be gauged against the natural variability of the system, including sediment inputs from the Orange River and from Berg winds (Zoutendyk 1995, Pulfrich 2005). Four environmental factors that are potentially linked to dredge-mining were examined: temperature, dissolved oxygen, hydrogen sulphide and turbidity. Because of its ecological and economic significances the rock lobster *Jasus lalandii* was chosen as the biological indicator against which the impacts of any mining-induced alterations could be assessed. Fig. 5.1 reviews the key findings as outlined in the sections that follow.

Temperature

It was expected that dredge-mining would reduce temperature stratification through the introduction of cold bottom-water into surface layers. However, the results showed that dredge-mining did not significantly alter temperature profiles in the water column. There was no evidence of any consistent differences in water stratification between natural (control) and mined conditions. Water temperatures were always cooler at the bottom than at the top of the water column, and the range of temperatures declined with depth. Control profiles had a mean surface water temperature of 12.7°C while mined profiles had a mean of 12.4°C (Fig. 5.1). The laboratory studies also yielded no conclusive results on any correlations between turbidity and temperature. Thus, alteration of temperature by dredge-mining or as a secondary consequence of increased turbidity is not a problem, at least in the relatively shallow waters examined in this study. A different story may emerge if dredging is undertaken in deeper areas exceeding 80 m.

Dissolved Oxygen

The review of the natural and dredge-mining related variability in dissolved oxygen revealed subtle patterns. There was a natural decrease in dissolved oxygen with an increase in depth – in both control and mining-related cases (Fig. 5.1), which was clear-cut in instances where thermal stratification was marked but absent when the water column was mixed (Chapter 2). The surface strata of mined profiles had significantly lower dissolved oxygen concentrations than those of control profiles for 50% of the cases evaluated – all also cases in which the water column was stratified. Dissolved oxygen was inversely correlated with turbidity, although the evidence for this being a cause-and-effect relationship was weak. Again, these conclusions are valid only for the relatively shallow waters (< 80 m) examined in the dredging trial.

Hydrogen Sulphide

No data for hydrogen sulphide concentration in the water column were available for the mining areas. However, Bailey *et al.* (1985) recorded water-soluble sulphide concentrations of 10-30 mg/ℓ in the sediment of certain areas in the Northern Benguela. Mining-related influences on hydrogen sulphide remain unknown, but the prediction is that dredging in sulphide-prone areas will liberate hydrogen sulphide from the sediment into the water at concentrations within or greater than the range reported by Bailey *et al.* (1985). This will result in potentially catastrophic impacts on benthic fauna in affected areas. The laboratory experiments on juvenile rock lobsters revealed that critical evasive tail flips occurred at sulphide levels as low as 3.9 mg/ℓ and moribundity at values as low as 14.8 mg/ℓ. The release of this gas will not only negatively affect benthic species, but all marine fauna in the surrounding water and may even pose health risks to humans on board the mining vessel.

Turbidity

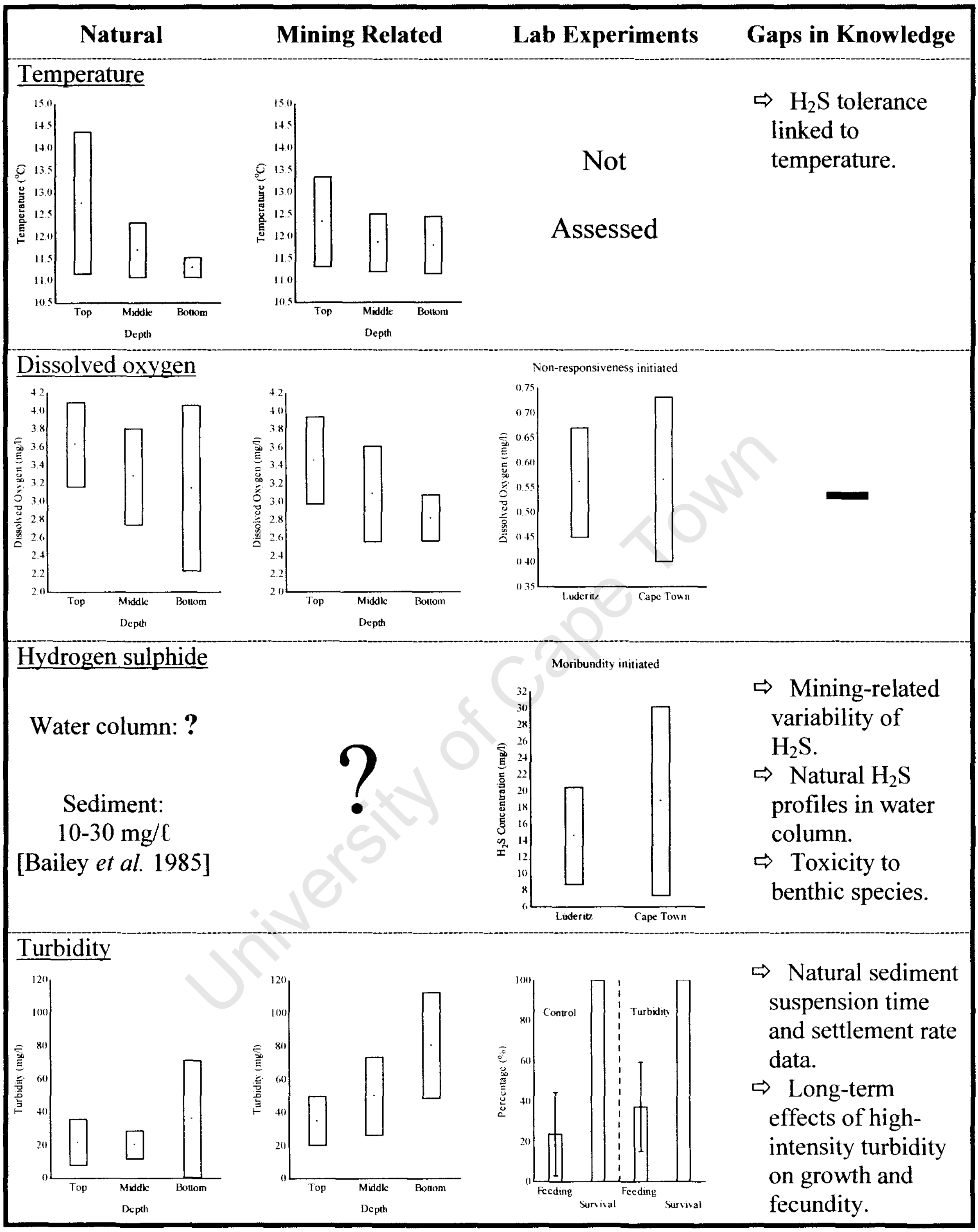
One perception with regards to turbidity was that it would increase with depth, with the highest levels being found close to the seafloor as a result of the natural suspension of sediment by currents. The data reviewed supported this view, since 75% of the cases examined had the highest natural turbidity in the bottom water strata. A second perception was that dredge-mining would increase turbidity. The profiling data revealed that controls had mean turbidity counts of 38 mg/ℓ in the bottom strata, 20 mg/ℓ in the middle strata and 21 mg/ℓ in the surface strata (Fig. 5.1). Mined profiles, on the other hand, had mean turbidity counts of 80 mg/ℓ in the bottom strata, 50 mg/ℓ in the middle strata and 37 mg/ℓ in the surface strata. The change in turbidity from the controls to mined profiles constituted a 2-fold increase in the bottom strata, 2.5-fold increase in the middle strata and a 1.8-fold increase in the surface strata.

Impacts on lobsters

The short and medium-term laboratory experiments yielded no negative impacts of turbidity on juvenile rock lobsters. Survival was 100% for the medium-term turbidity assessments and food consumption of lobsters subject to turbidity was similar to that of control animals (Fig. 5.1), although this outcome may simply reflect the fact that the turbidity induced in the experiment was relatively low, falling in the lower end of values recorded in the environment during both natural conditions and mining.

The dissolved oxygen experiments showed that juvenile lobsters survived hypoxic concentrations as low as 0.40 mg O_2/ℓ but many became 'non-responsive' at 0.56 mg/ℓ (Fig. 5.1). Although classified as 'non-responsive' lobsters were still capable of flight responses at these oxygen concentrations. No statistically significant gender or site-related differences were found for the Lüderitz versus Cape Town comparisons.

Hydrogen sulphide had the most dramatic impact on the lobsters out of the three laboratory treatments examined. Short-term survival results showed that Lüderitz and Cape Town lobsters were (on average) intolerant to hydrogen sulphide concentrations higher than 14.8 and 19 mg/ℓ, respectively, which were the levels at which moribundity set in (Fig. 5.1). The lowest hydrogen sulphide concentrations at which escape responses were elicited ranged between 0.7 and 1.2 mg/ℓ, indicating the range at which lobsters first experienced the sulphide stress.



	Natural	**Mining Related**	**Lab Experiments**	**Gaps in Knowledge**
Temperature			Not Assessed	⇨ H_2S tolerance linked to temperature.
Dissolved oxygen			Non-responsiveness initiated	—
Hydrogen sulphide	Water column: ? Sediment: 10-30 mg/ℓ [Bailey *et al.* 1985]	?	Moribundity initiated	⇨ Mining-related variability of H_2S. ⇨ Natural H_2S profiles in water column. ⇨ Toxicity to benthic species.
Turbidity				⇨ Natural sediment suspension time and settlement rate data. ⇨ Long-term effects of high-intensity turbidity on growth and fecundity.

Figure 5.1: Diagrammatic summary of the findings of the study. Dots on bar graphs represent means and bars (and whiskers) represent standard deviation.

The medium-term hydrogen sulphide behavioural experiments provided a good reference against which to gauge the progression of stress as the sulphide concentration increased. The critical level at which rock lobsters exhibited the strongest avoidance behaviour was between 3.9 and 5.7 mg/ℓ. The concentrations of hydrogen sulphide at which lobsters became completely incapacitated ranged between 14.8 and 33.4 mg/ℓ. However, all the lobsters used in the short-term survival and medium-term behavioural experiments recovered within 14 hours after the sulphide stress was removed.

In summary (Fig. 5.1): (1) Field data showed that under natural (control) conditions, temperature and oxygen usually declined with depth whereas turbidity increased. With the exceptions of temperature (11-11.5°C) and hydrogen sulphide (unknown), wide variations were the norm, with bottom values spanning 2.2-4.0 mg/ℓ for dissolved oxygen, and <17-70 mg/ℓ for turbidity. (2) There was no evidence that dredge-mining influenced water-column temperature, some suggestion that it did deplete surface oxygen, and strong evidence that it enhanced turbidity. (3) From the laboratory experiments, the rock lobsters displayed no adverse reactions to turbidity, but were stressed and became 'non-responsive' when oxygen dropped to 0.56 mg/ℓ, and moribund when hydrogen sulphide levels reached between 14 and 33 mg/ℓ.

Recommendations for Mitigation and Future Research

Several shortcomings and difficulties were experienced during the study. Some of these have been discussed in the relevant chapters, but here I revisit the limitations of the project and outline potential future studies that could expand this study to resolve whether offshore dredge-mining indeed poses a threat to rock lobster populations.

Holism over Species-specific Approach

One fundamental role of an Environmental Impact Assessment (EIA) is to safeguard the diversity and functionality of the ecosystem. This implies that caution should be practised when planning EIAs to ensure that economic motivations do not over-shadow ecological significance and thus bias the nature of the assessments in favour of economics. Economic significance is not a proxy for ecological significance. *Jasus lalandii* forms part of a complex biotic system in which the characteristics of the system are defined by the presence of not one, but a multitude of species, although it is true that the presence or absence of a single (keystone) species such as *J. lalandii* can greatly alter the nature of an ecosystem (Barkai & Branch 1988b). Adopting a species-specific approach over a holistic (ecosystem) approach is contrary to the fundamentals of an EIA. It can contribute one step but it cannot constitute an entire assessment.

Against this background, holistic ecosystem-wide impact assessments are encouraged for future studies on the effects of dredge-mining. If, however, for economic and/or practical reasons such a holistic approach cannot be implemented, at least a focused multi-species approach should be used – selecting species that are closely linked to the target species the assessment is intended for. In the current scenario the preferred prey of *J. lalandii* (e.g. mussels and sea urchins) would constitute desirable additions to the impact assessment since indirect effects on the target species can be measured from direct impacts on the prey species. The underlying concept of holism is to encompass as many species and environmental variables as possible, so as to obtain a multi-variate dataset that can be used to draw meaningful conclusions from.

In-situ versus Lab-based Assessments

A major component of this study was confined to the laboratory where physical conditions were simulated to represent potential negative conditions for assessing the impacts of gaseous and turbidity conditions on rock lobsters. The strength of this approach is that the variables of interest (low oxygen, hydrogen sulphide and turbidity) are controlled, so that their individual impacts can be assessed without the potentially compounding influence of other variables. However, results from lab-based assessments should always be considered with caution because many factors are either excluded from the assessment or kept constant, which gives an unrealistic account of what might happen if all the known and unknown factors were to be incorporated. *In-situ* assessments are thus a desirable complement to laboratory studies because they do incorporate natural variations in a multiplicity of factors, although they can be difficult to interpret for this very reason.

This study paves the way in combining *in-situ* and laboratory research. It began with an overview of the background information (Chapter 1), then reviewed existing work on *in-situ* conditions and the effects of mining (Chapter 2), and finally turned to laboratory studies to explore the possible effects of dissolved oxygen, hydrogen sulphide and turbidity as isolated variables on one key species, the rock lobster *Jasus lalandii*. One problem associated with these lab-based assessments was that they could not incorporate the complicated effects of water stratification and current dynamics, which are difficult to simulate in the laboratory. As discussed in Chapter 1, the Benguela ecosystem features very dynamic current regimes, as well as seasonal water stratification resulting from the combined effects of thermal and saline profiles within the water column. It is thus important to determine how these two factors (i.e. water currents and vertical stratification) affect gas attributes and sediment suspension (e.g. residence time and gas turn-over rates) since the combined effect will influence the intensity and duration of stresses imposed on lobsters condition, thus dictating lobster behaviour, and even survival rates. A second problem with the laboratory studies was that they could not always simulate the full range of likely conditions in the field. While the dissolved oxygen and hydrogen sulphide experiments did span the full range, the turbidity experiment did not. Far higher turbidity levels were recorded in the field, especially associated with mining, than I was able to simulate in the laboratory. An expansion of the turbidity studies is needed to (a) incorporate turbidity levels that realistically approach those recorded during mining, and (b) run for longer periods. For this, a different experimental design will be needed, as the magnetic stirrers I employed were run at their maximum potential and still failed to induce realistic turbidities.

Temporal and Spatial Aspects of Assessments

Environmental Impact Assessments (EIAs) need to be conducted on appropriately broad spatial and temporal scales, particularly in the case of monitoring assessments. It is thus important to effect long-term forecasts of mining strategies (perhaps two to three years) before feasibility studies are implemented, so that the necessary monitoring data are collected and analysed well in advance. This will ensure effective decision-making during the final stages of resource evaluation and serve as guidelines for the nature of feasibility studies. During and post-mining monitoring studies are also important, as these data are crucial to evaluate impacts and subsequent recovery. Necessary mitigation can be implemented as results emerge.

In the present context, the short-term laboratory experiments provide information on the immediate responses of lobsters, but would profitably have been supplemented with long-term studies spanning sufficient time periods to assess the sub-lethal effects of oxygen depletion, hydrogen sulphide elevation and turbidity on lobster condition, growth and reproduction.

Lobster-specific Assessments

This study was aimed at assessing the effects of potential dredging-induced impacts on the behaviour and survival of juvenile rock lobsters. Although this study yielded useful data, it is a starting point, not a final conclusive study. To fully understand the impacts of dredge-mining on lobsters, complementary studies need to be undertaken. Recommendations on key issues to be addressed in future research studies are briefly discussed below:

- Information is needed on natural variability in hydrogen sulphide and the effects of dredge-mining on this gas. Absence of this knowledge made it impossible to use my laboratory results on hydrogen sulphide to make predictions about the effects of any mining-induced elevation of hydrogen sulphide.
- Emphasis should be placed on *in-situ* assessments. This will ensure that crucial factors (current, vertico-horizontal temperature and gas profiles, water stratification etc.) are covered in the assessments and thus more realistic results are obtained from the studies.
- All life cycle stages of the lobsters (larval, juvenile & adult) should be considered. This would shed light on any differences in tolerance among the different stages. This is important as an impact-minimization indicator for selecting mining areas (for example, if juveniles are found to be the stage most impacted by mining activity then measures could be taken to avoid mining in and around recruitment areas).
- Mining impacts on fecundity (reproductive output) and recruitment should be assessed over a long-term (year-scale) period. These variables are considered as critical factors since reductions of recruitment of pueruli and fecundity would have severe implications for stock resilience and the fishing sector.
- Impacts on feeding and growth would be useful, particularly if combined with studies on the impacts of mining on the prominent prey species of *Jasus lalandii*, thus shedding light on possible effects on benthic communities in mining areas.
- Measurements of the effects of dredge-mining on nutrient dynamics of affected areas, and the rate of smothering of the benthos.

Mining-specific Recommendations

Control sites and monitoring

It is recommended that adequately replicated, clearly defined and permanent control sites be identified and set aside. These should represent previously non-mined areas and should be maintained on an absolute policy of no-mining. Such control sites should have benthic community structures comparable to those of mining areas. This will allow comparative monitoring of control and mined areas to assess immediate impacts and subsequent secondary succession. It will also identify trends in recovery rates for different benthic communities as well as the probabilities of recovery to a pre-mining state. Control and mined sites should be replicated to avoid the criticism that any differences between them are simply site-specific. Monitoring should be done before, during and after mining and over appropriate durations. Data from the benthic baseline and subsequent monitoring surveys conducted by DBMN would prove useful in this regard. Most epifaunal species found on sandy habitats are motile and thus the monitoring of infaunal communities should suffice in assessing mining-related impacts – which is covered by the benthic surveys conducted by DBMN.

Regular monitoring of lobster abundance and size composition (preferably by Fisheries Independent Monitoring Surveys) and of mining attributes for the entire duration of the mining process is essential. This is crucial for the assessments of long-term and inter-seasonal patterns, as well as building a dataset on natural and mining-induced variability. One possible approach would be to employ automated *in-situ* multi-parameter recorders for physical factors such as temperature, dissolved oxygen, hydrogen sulphide and turbidity to get high resolution time-correlated data for these physical factors. Key issues that this type of monitoring should address are the daily, monthly and annual variability of physico-chemical factors at appropriate scales alongshore, offshore and relative to depth. True, this may have significant financial implications, however, solutions can be found through strategic planning and approaching interested and affect organisations that are willing to co-operate on such projects.

Mining in hydrogen sulphide prone areas

The feasibility of mining needs to be carefully considered where diamond resources are found to coincide with sediment laden with hydrogen sulphide. My laboratory studies indicated that of the three variables (dissolved oxygen, hydrogen sulphide and turbidity) examined it is the increase in hydrogen sulphide concentration that constitutes the greatest threat. Not knowing how mining may alter hydrogen sulphide levels is thus a critical gap in knowledge. Against this background it is advised that detailed pre-mining studies be conducted to determine the spatial extent, concentration and ecological significance of hydrogen sulphide in the areas intended for mining. This could be achieved through studies where *in-situ* profiling with regards to hydrogen sulphide is conducted before, during and after mining in hydrogen sulphide prone areas. This can be supported by pre and post-mining benthic surveys to assess the impact from mining.

This information should then be used in conjunction with the results of my experiments and any other existing data to gauge the potential impacts of mining the area. This information can then be used, in conjunction with dredgers equipped with *in-situ* sulphide-measuring devices to measure sulphide concentrations in the turbidity plumes created by the drag-head, to set a critical toxicity level (in line with ecological limits) at which mining will be stopped until such time that sulphide concentrations return to acceptable levels.

Mitigation of mining impacts

The most prominent concern regarding dredge-mining relates to the high sediment load in the water column resulting from both the resuspension of sediment during actual dredging process as well as from vessel overspill – with overspill being more significant. This has two impacts: (1) the active displacement of large quantities of sediment away from the point of extraction, and (2) the remote effects (e.g. smothering of reefs) caused by the settling of sediment. A possible tactic to mitigate both impacts with a single application would be to return overspill to the original point of extraction (the OSD, Original-Site-Discharge, tactic). The current dredging procedure is to suck-up sediments from the seafloor and transfer it to the hopper on the mining vessel, where the heavier sand and gravel components settle and the suspended materials (mainly fines) are released, via the overflow sluices of the hopper, back into the ocean. Instead of allowing this to take place it could be possible to direct the overspill into pipelines extending to the seafloor, to areas that have just recently been mined, while mining is ongoing. A practical approach for the TSHDs would be to have a rigid pipe attached to the upper side of the suction pipe (connecting the dredge-head to the vessel) transfer the overspill directly from the hopper's overflow sluices to recently mined-out areas in the lee of the drag-head.

A similar concept could be applied to the discharge of fines from land-based paddocks. Current practice is to discharge unwanted sediment into the intertidal zone during the processing phases of the gravels from the paddocks – resulting in significant beach accretion that impacts the beach ecosystem (Pulfrich *et al.* 2003b, CSIR 2005b). Instead of discharging these sediments onto the beach a pipeline could be constructed to transport the fines offshore to a deep outfall – a process referred to as submarine tailings disposal (STD) (Ellis *et al.* 1995). This is considered to be one of the best impact mitigation measures to date, given that the necessary environmental safeguards are in place and STD protocols adhered to (see Ellis *et al.* 1995 and Ellis 1995).

True, there may be significant financial costs associated to both the OSD and STD approaches, but the ecological and ethical obligations of conservation and sustainable resource utilization should be weighed against these costs. Of these two approaches the OSD is probably more feasible, given the strong wave action on the coast and the costs of maintaining an STD (Ellis *et al.* 1995).

To further justify the OSD approach, its potential benefits are outlined below:

- Algal blooms: the discharge of potentially nutrient-rich sediment from deeper depths into warmer and better illuminated surface waters (as is the case with the current overspill scenario) may increase the frequency, intensity and duration of algal blooms, which could have powerful effects on immediate and remote pelagic and benthic marine species. The OSD tactic would greatly reduce the potential of unnatural blooms by returning sediment to its original point of extraction.
- Turbidity plumes: in nearshore areas strong currents keep fine sediment in suspension for long periods of time and, as a result, it may be advected great distances along and offshore. These turbidity plumes may reduce light penetration within the water column. Depending on the duration and amount of sediment being discharged, this could affect metabolic equilibria (i.e. photosynthetic versus respiration rates) and potentially inhibit productivity. Again, the OSD tactic would mitigate this by returning processed sediment to its point of origin and not in the surface waters.
- Remote sedimentation: the settling of suspended sediment in areas far away from the mining and processing areas because of its transport by strong currents can be mitigated with the OSD tactic. Heavier material (i.e. sand) will settle immediately at the point of discharge while lighter material (fines) will be relatively localised since currents near the seabed are generally slow. Studies by the CSIR (2004) show that plumes arising from dredge-mining remain confined to the vicinity of mining. The OSD tactic would be an improvement on the current approach where sediment is released in surface waters where

currents tend to be stronger. In addition to this, the OSD tactic would minimise reef scouring and smothering since sediment would be returned to the point of extraction which will have limited benthic fauna because of the disturbance caused by mining.

The other concern about dredging is the mechanical damage caused by the drag-head. This destroys epi- and infaunal species in the mined areas. Naturally the impact will be less severe for mobile species (e.g. lobsters and benthic fish), since they can avoid the impact by moving away from the mining areas. However, sedentary and sessile species are particularly threatened by mechanical damage. For this reason it is imperative to determine the benthic community composition in control and mining areas – before and after mining. This will allow an assessment of the magnitude of mining effects and the duration of recovery. Pre-mining surveys would also allow a pre-emptive shift of mining to another site if the benthic community has a high conservation status or commercial significance.

This study has clearly shown that dredge-mining has a noticeable affect on turbidity but not oxygen or temperature (Chapter 2). Mining will also disrupt benthic communities – a topic that needs separate investigation. It remains unknown whether dredge-mining will influence hydrogen sulphide levels, but my laboratory experiments suggest lobsters are more sensitive to hydrogen sulphide elevation than to oxygen depletion (Chapter 3) or an increase in turbidity (Chapter 4). However, necessary steps to fill gaps in knowledge about dredge-mining impacts are highlighted in this concluding chapter.

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